Appendix 4.A, Status of the Species and Critical Habitat Accounts

4.A Status of the Species and Critical Habitat Accounts

4.A.1 Chinook Salmon, Sacramento River Winter-Run (*Oncorhynchus tshawytscha*)

This section provides information on the basic biology, life history, status, and threats and stressors of Sacramento River winter-run Chinook salmon in the action area.

4.A.1.1 Status

The Sacramento River winter-run Chinook salmon evolutionarily significant unit (ESU) is listed as an endangered species under the ESA. The ESU includes all naturally spawned winter-run Chinook salmon in the Sacramento River and its tributaries (Figure 4.A.1-1), as well as winter-run Chinook salmon from one artificial propagation program: the Livingston Stone National Fish Hatchery (50 CFR 224.101(h)). The captive broodstock program at the Livingston Stone National Fish Hatchery was discontinued in 2007 and only natural origin brood stock are used for propagation in the hatchery (California HSRG 2012). In 2014 in response to emergency drought conditions and the loss of the majority of naturally produced winter-run Chinook in-river brood stock criteria were lessened allowing hatchery origin fish in response to ramping up production to buffer against drought affects.

The Sacramento River winter-run Chinook salmon ESU was initially listed as a threatened species in August 1989, under emergency provisions of the federal Endangered Species Act (ESA) (54 *Federal Register* [FR] 32085; August 4, 1989), and was listed as threatened in a final rule in November 1990 (55 FR 46515; November 5, 1990). The ESU consists of only one population confined to the mainstem of the upper Sacramento River in California's Central Valley below Keswick Dam. The ESU was reclassified as endangered under the ESA on January 4, 1994 (59 FR 440), because of increased variability of run sizes, expected weak returns as a result of two small year classes in 1991 and 1993, and a 99% decline between 1966 and 1991. The National Marine Fisheries Service (NMFS) reaffirmed the listing of the Sacramento River winter-run Chinook salmon ESU as endangered on June 28, 2005 (70 FR 37160), and included winter-run Chinook salmon in the Livingston Stone National Fish Hatchery artificial propagation program in the ESU. This ESU is not considered to be viable because there is only one extant population, which is spawning outside of its historical range, in artificially maintained habitat that is vulnerable to drought. The rising levels of hatchery-origin Feather River spring-run fish spawning areas among winter-run poses another concern for the viability of winter-run stocks.

In its latest 5-year status review, NMFS concluded that the extinction risk of this ESU has increased since the last status review and its classification as an endangered species was still appropriate (NMFS 2011). NMFS determined that the ESU had continued to decline since 2005, with a negative point cohort replacement rate for the 10-year trend. However, the current population size still falls within the low-risk criterion, and the 10-year average introgression rate of hatchery fish (about 8%) is below the low-risk threshold for hatchery influence (NMFS 2011).

CESA: Sacramento River winter-run Chinook salmon was listed as endangered under the California Endangered Species Act (CESA) on September 22, 1989.

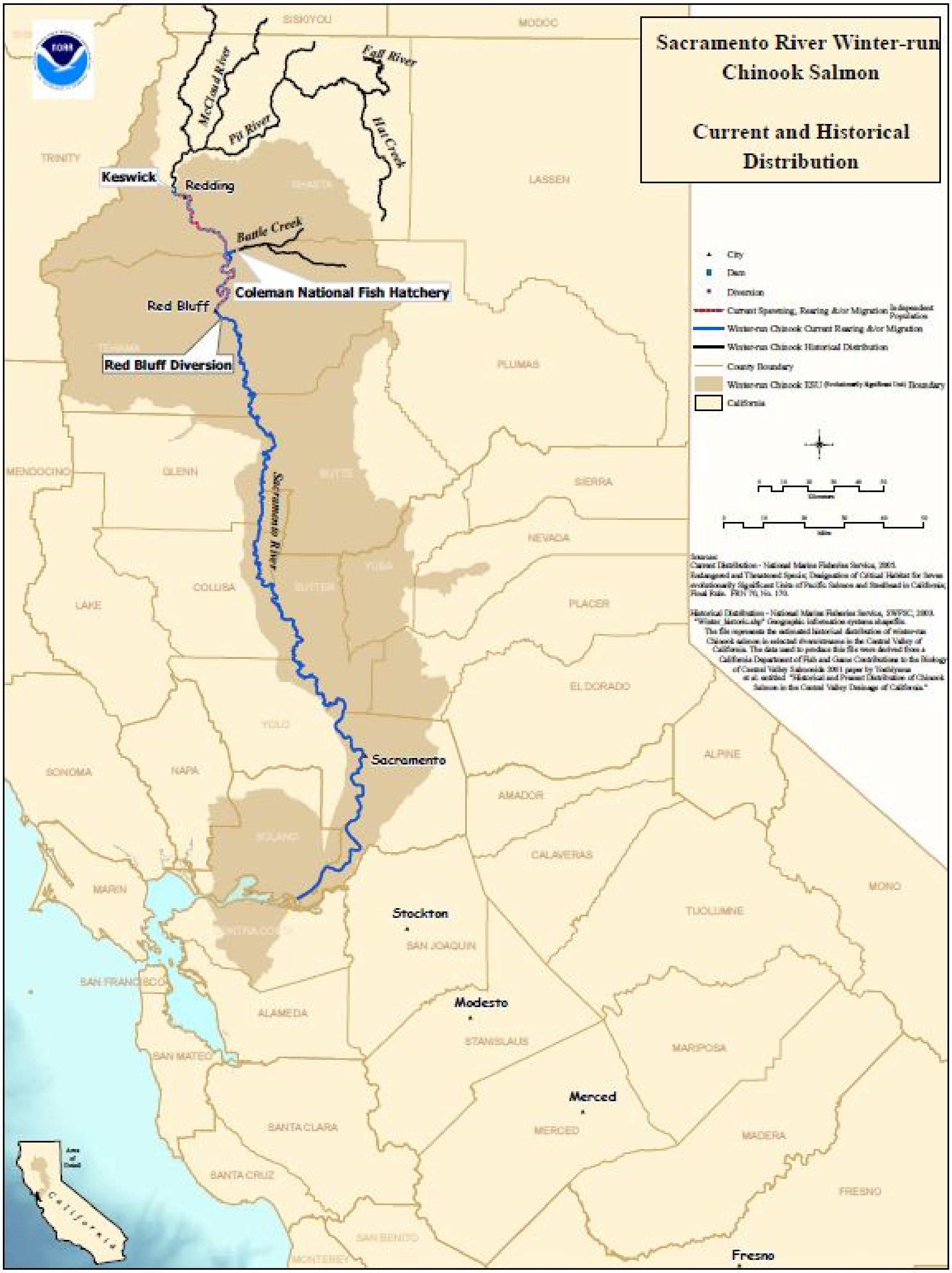

Figure 4.A.1-1. Sacramento River Winter-Run Chinook Salmon Evolutionarily Significant Unit Boundary, and Current and Historical Distribution (Source: NMFS 2014)

4.A.1.2 Critical Habitat

Critical habitat for the Sacramento River winter-run Chinook salmon ESU was designated under the ESA on June 16, 1993 (58 FR 33212). Designated critical habitat includes the Sacramento River from Keswick Dam (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Delta, all waters from Chipps Island westward to Carquinez Bridge, including Honker, Grizzly, and Suisun Bays, and 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) from San Pablo Bay to the Golden Gate Bridge (59 FR 440, January 4, 1994). In the Sacramento River, critical habitat includes the river water column, river bottom, and adjacent riparian zone used by fry and juveniles for rearing. In the areas westward of Chipps Island, critical habitat includes the estuarine water column and essential foraging habitat and food resources used by winter-run Chinook salmon as part of their juvenile emigration or adult spawning migration.

The designated critical habitat includes physical or biological features (PBFs)[1] that are essential for the conservation of winter-run Chinook salmon: (1) access from the Pacific Ocean to appropriate spawning areas (as outlined below) in the upper Sacramento River, (2) the availability of clean gravel for spawning substrate, (3) adequate river flows for successful spawning, incubation of eggs, fry development and emergence, and downstream transport of juveniles, (4) water temperatures between 42.5 and 57.5 °F (5.8 and 14.1°C) for successful spawning (NMFS 2014), egg incubation, and fry development, (5) habitat areas and adequate prey that are not contaminated, (6) riparian habitat that provides for successful juvenile development and survival, and (7) access downstream so that juveniles can migrate from the spawning grounds to San Francisco Bay and the Pacific Ocean (NMFS 2014). The condition of these features is described below.

Pacific salmon habitat (inclusive of winter-run Chinook salmon) is also protected under the Magnuson-Stevens Fishery Conservation and Management Act as essential fish habitat (EFH). Those waters and substrate necessary to support Chinook salmon spawning, breeding, feeding, or growth—including those for winter-run Chinook salmon—are included as EFH (Figure 4.A.1-2).

[1] The designations of critical habitat for listed species have generally used the term primary constituent elements (PCEs). NMFS and USFWS' recently issued a final rule amending the regulations for designating critical habitat (81 FR 7414; February 11, 2016), which replaced the term PCEs with physical or biological features (PBFs). In addition, NMFS and USFWS' recently issued a final rule revising the regulatory definition of "destruction or adverse modification" of critical habitat (81 FR 7214; February 11, 2016), which refers to PBFs, not PCEs. The shift in terminology does not change the approach used in conducting an analysis of the effects of the proposed action on critical habitat, which is the same regardless of whether the original designation identified PCEs or PBFs. In this biological assessment, we use the term PBFs to include PCEs, as appropriate for the specific critical habitat, for NMFS' species.

Figure 4.A.1-2. Chinook Salmon Freshwater Essential Fish Habitat (Source: PFMC 2014)

4.A.1.2.1 Spawning Habitat

According to (NMFS 2014), freshwater spawning sites should provide water quantity and quality conditions and substrate supporting spawning, incubation and larval development. Spawning habitat for winter-run Chinook salmon is restricted to the Sacramento River primarily between Red Bluff Diversion Dam (RBDD) and Keswick Dam. In-progress construction in Battle Creek is creating a section of suitable spawning and rearing habitat for winter-run Chinook salmon, although individuals have not yet used the restored habitat for spawning. Spawning sites include those stream reaches with clean, loose gravel, in swift, relatively shallow riffles, or along the margins of deeper river reaches where suitable water temperatures, depths, and velocities favor redd construction and oxygenation of incubating eggs (NMFS 2014).Water velocity and substrate conditions are more critical to the viability of spawning habitat than depth. Incubating eggs and embryos buried in gravel require sufficient water flow through the gravel to supply oxygen and remove metabolic wastes (Kondolf et al. 2008). Spawning occurs in gravel substrate in relatively fast moving, moderately shallow riffles or along banks with relatively high water velocities. The gravel must be clean and loose, yet stable for the duration of egg incubation and the larval development.

Substrate composition has other key implications to spawning success. The embryos and alevins (newly hatched fish with the yolk sac still attached) require adequate water movement through the substrate; however, this movement can be inhibited by the accumulation of fines and sand. Generally, a redd should contain less than 5% fines (Kondolf et al. 2008).

Water velocity in Chinook salmon spawning areas typically ranges from 1.0 to 3.5 feet per second and optimum velocity is 1.5 feet per second (Hampton 1988). Spawning occurs at depths between 1 to 5 feet with a maximum observed depth of 20 feet. A depth of less than 6 inches can be restrictive to Chinook salmon movement.

4.A.1.2.2 Freshwater Rearing Habitat

According to (NMFS 2009b), freshwater salmon rearing habitats contain sufficient water quantity and floodplain connectivity to form and maintain physical habitat conditions that support juvenile growth and mobility; suitable water quality; availability of suitable forage species that support juvenile salmon growth and development; and cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. In the Sacramento River, both spawning areas and migratory corridors also function as rearing habitat for juveniles, which feed and grow before and during their out-migration. Nonnatal, intermittent tributaries also may be used for juvenile rearing. Rearing habitat value is strongly affected by habitat diversity and complexity, food supply, and fish and avian predators. Channeled, leveed, and riprapped river reaches and sloughs are common along the Sacramento River and throughout the Delta; which typically have low habitat complexity, have low abundance of food organisms, and offer little protection from predation by fish and birds. However, some of these more complex and productive habitats with floodplains are still found in the system (e.g., reaches with setback levees [i.e., primarily located upstream of the City of Colusa]).

4.A.1.2.3 Freshwater Migration Corridors

According to (NMFS 2014), Freshwater migration corridors for winter-run Chinook salmon, including river channels, floodplains, channels through the Delta, and the Bay-Delta estuary should support mobility, survival, and food supplies for juveniles and adults. Migration corridors from the Pacific Ocean to the upper Sacramento River should be free from obstructions (passage barriers and impediments to migration), providing satisfactory water quality, water quantity, water temperature, water velocity, cover, shelter, and safe passage conditions in order for adults to reach spawning areas. Migratory corridors for winter-run Chinook salmon are located downstream of the spawning areas and include the lower Sacramento River, the Delta, and the San Francisco Bay complex extending to coastal marine waters. These corridors allow the upstream passage of adults and the downstream emigration of juvenile salmon. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, limit false attraction, provide low vulnerability to predation, and not contain impediments and delays in both upstream and downstream migration.

Results of mark-recapture studies conducted using juvenile Chinook salmon (typically hatchery-reared late fall-run Chinook salmon that are considered to be representative of juvenile winter-run Chinook salmon) released into the Sacramento River have shown high mortality during passage downstream through the rivers and Delta (Brandes and McLain 2001; Newman and Rice 2002; Hanson 2008; del Rosario et al. 2013). Mortality is typically greater in years when spring flows are reduced and water temperatures are increased. Results of survival studies have shown that closing the Delta Cross Channel gates to reduce the movement of juvenile salmon into the Central Delta, contributes to improved survival of emigrating juvenile Chinook salmon (Brandes and McLain 2001; Manly 2004). Results of estimating incidental take of juvenile winter-run Chinook salmon at the Central Valley Project (CVP) / State Water Project (SWP) fish salvage facilities based on comparison of the juvenile production estimates for winter-run Chinook salmon emigrating from the upper Sacramento River rearing areas (e.g., estimated based on results of spawning carcass surveys and environmental conditions and/or fishery monitoring at RBDD) generally show similar small direct losses of Sacramento River juvenile winter-run Chinook salmon at the fish salvage facilities. There are several factors affecting direct loss of Sacramento River juvenile salmon at salvage facilities including pumping rates, Sacramento River flow, run timing, species abundance, water year type, DCC gate operations and predator abundance (Larry Walker Associates 2010, Buchanon 2013, Cloern 2012, Harvey 2011, Perry & Brandes 2010, Perry & Skalski 2010, Perry 2012, Zeug & Cavallo 2013, Perry et al 2015).

4.A.1.2.4 Estuarine Habitat

Estuarine migration and juvenile rearing habitats should be free of obstructions (i.e., dams and other barriers) and provide suitable water quality, water quantity (river and tidal flows), and salinity conditions to support juvenile and adult physiological transitions between fresh and salt water. Natural cover, such as submerged and overhanging large wood, native aquatic vegetation, and side channels, provide juvenile foraging habitat and cover from predators. Tidal wetlands and seasonally inundated floodplains have also been identified as high-value foraging and rearing habitats for juvenile salmon migrating downstream through the estuary. Estuarine areas contain a high conservation value because they function to support juvenile Chinook salmon growth, smolting, and avoidance of predators, as well as provide a transition to the ocean environment (NMFS 2009b; NMFS 2014).

The current condition of the estuarine habitat in the action area has been substantially degraded from historic conditions. Over 90% of the fringing fresh, brackish, and salt marshes have been lost to human actions. This loss of the fringing marshes reduces the availability of forage species and eliminates the cycling of nutrients from the marsh vegetation into the water column of the adjoining waterways. The channels of the Delta have been modified by the raising of levees and armoring of the levee banks with stone riprap. This reduces habitat complexity by reducing the incorporation of woody debris and vegetative material into the nearshore area, minimizing and reducing local variations in water depth and velocities, and simplifying the community structure of the nearshore environment. Delta hydraulics has been modified as a result of CVP/SWP actions. Within the central and southern Delta, net water movement is towards the pumping facilities, altering the migratory cues for emigrating fish in these regions. Operations of upstream reservoir releases and diversion of water from the southern Delta have been manipulated to maintain a “static” salinity profile in the western Delta near Chipps Island (the X2 location). This area of salinity transition, the low salinity zone (LSZ), is an area of high productivity. Historically, this zone fluctuated in its location in relation to the outflow of water from the Delta and moved westwards with high Delta inflow (i.e., floods and spring runoff) and eastwards with reduced summer and fall flows. This variability in the salinity transition zone has been substantially reduced by the operations of the CVP/SWP projects. The CVP/SWP long-term water diversions also have contributed to reductions in the phytoplankton and zooplankton populations in the Delta itself as well as alterations in nutrient cycling within the Delta ecosystem. Heavy urbanization and industrial actions have lowered water quality and introduced persistent contaminants to the sediments surrounding points of discharge (i.e., refineries in Suisun and San Pablo bays, creosote factories in Stockton, etc.)

4.A.1.2.5 Marine Habitats

Although ocean habitats are not part of the designated critical habitat for Sacramento River winter-run Chinook salmon, biologically productive coastal waters are an important habitat component for the species. Juvenile Chinook salmon inhabit near-shore coastal marine waters for a period of typically 2 to 4 years before adults return to Central Valley rivers to spawn. During their marine residence, Chinook salmon forage on krill, squid, and other marine invertebrates and a variety of fish such as northern anchovy, sardines, and Pacific herring.

The variation in ocean productivity off the West Coast can be high both within and among years. Changes in ocean currents and upwelling have been identified as significant factors affecting nutrient availability, phytoplankton and zooplankton production, and the availability of other forage species in near-shore surface waters. Ocean conditions during a salmon’s ocean residency period can be important, as indicated by the effect of the 1983 El Niño on the size and fecundity of Central Valley fall-run Chinook salmon (Wells et al. 2006). Ocean conditions are thought to be one of the primary causes of Central Valley fall-run Chinook stock collapse in 2008 (Lindley et al. 2009). Although the effects of ocean conditions on Chinook salmon growth and survival have not been investigated extensively, recent observations since 2007 have shown a significant decline in the abundance of adult Chinook and coho salmon returning to California rivers and streams (Pacific Fishery Management Council 2008). The decline has been hypothesized to be the result of decreased ocean productivity and associated high mortality rates during the period when these fish were rearing in near-shore coastal waters (MacFarlane et al. 2008b; Pacific Fishery Management Council 2008). The importance of changes in ocean conditions on growth, survival, and population abundance of all races of California Chinook salmon is currently undergoing further investigation (*sensu* Peterson et al. 2012, Sharma et al. 2012).

4.A.1.3 Life History

Chinook salmon exhibit two generalized freshwater life history types (Healey 1991). Stream-type adults enter fresh water months before spawning and juveniles reside in fresh water for a year or more following emergence, whereas ocean-type adults spawn soon after entering fresh water and juveniles migrate to the ocean as fry or parr in their first year. Adequate instream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting a stream-type life history due to over-summering by adults and/or juveniles. Winter-run Chinook salmon are somewhat anomalous in that they have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter fresh water in winter or early spring, and delay spawning until spring or early summer (stream-type). However, juvenile winter-run Chinook salmon migrate to sea after only 5 to 9 months of river life (ocean-type). This life-history pattern differentiates the winter-run Chinook from other Sacramento River Chinook runs and from all other populations within the range of Chinook salmon (Hallock and Fisher 1985).

In addition to their unique life-history patterns, the behavior of winter-run Chinook adults as they return to spawn differentiates the population. Adults enter freshwater in an immature reproductive state, similar to spring-run Chinook, but winter-run Chinook move upstream much more quickly and then hold in the cool waters downstream of Keswick Dam for an extended period before spawning (Moyle et al. 1989).

4.A.1.3.1 Adult Migration and Holding

Sacramento River winter-run Chinook salmon adults enter the Sacramento River basin between December and July; the peak occurs in March (Table 4.A.1-1) (Yoshiyama et al. 1998, Moyle 2002). Because winter-run Chinook salmon use only the Sacramento River system for spawning, adults are likely to migrate upstream primarily along the western edge of the Delta through the Sacramento River corridor. Their migration past RBDD at river mile 242 begins in mid-December and continues into early August. The majority of the run passes RBDD between January and May, with the peak in mid-March (Hallock and Fisher 1985). The timing of migration may vary somewhat due to changes in river flows, dam operations, and water year type (Yoshiyama et al. 1998, Moyle 2002). Sacramento River winter-run Chinook salmon migrate into freshwater while still being immature and delay spawning for weeks or months upon reaching their spawning grounds (Healey 1991).

4.A.1.3.2 Spawning

In general, Sacramento River winter-run Chinook spawn in the area from Redding downstream to RBDD. However, the spawning distribution, as determined by aerial redd surveys is somewhat dependent on the operation of the gates at RBDD, river flow, and probably temperature. The permanent opening of RBDD gates may expand the timing and spatial distribution of winter-run Chinook salmon spawning.

Spawning occurs from mid-April to mid-August, peaking in May and June, in the Sacramento River reach between Keswick Dam and Red Bluff Diversion Dam (Vogel and Marine 1991). The majority of winter-run Chinook salmon spawners are 3 years old. Prespawning activity requires an area of 200 to 650 square feet. The female digs a nest, called a redd, with an average size of 165 square feet, in which she buries her eggs after they are fertilized by the male (Healey 1991).

Winter-run Chinook salmon use only the upper Sacramento River as spawning habitat, although occasional strays have been reported in Battle Creek and Clear Creek. Since fish passage improvements were completed at the ACID Dam in 2001, winter-run Chinook salmon spawning has shifted upstream. Since 2007, over half of winter-run Chinook salmon spawning has occurred in the area from Keswick Dam to the ACID Dam (approximately 5 miles) (NMFS 2009a).

4.A.1.3.3 Egg to Parr

Winter-run Chinook salmon fry begin to emerge from the gravel in late June to early July and continue through October (Table 4.A.1-1); Fisher 1994), with emergence generally occurring at night. Fry then seek lower velocity nearshore habitats with riparian vegetation and associated substrates important for providing aquatic and terrestrial invertebrates, predator avoidance, and slower velocities for resting (NMFS 1996).

Table 4.A.1-1. Temporal Occurrence of Adult and Juvenile Sacramento River Winter-Run Chinook Salmon in the Sacramento River and Delta.

Winter run relative abundance	High				Medium				Low			
a) Adults freshwater												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Immigration RBDD[a,b]												
Holding, Keswick, Bend Bridge[c]												
Spawning,egg incubation,alevins[d]												
Juvenile rearing, Keswick, RBDD[e]												
b) Juvenile emigration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Sacramento River at RBDD [d,f]												

Sources: [a] (NMFS 1997); [b] (Hallock and Fisher 1985); [c](Inferred from immigration and spawning timing) ; [d] (Vogel and Marine 1991) ; [e] (Gaines and Martin 2002); [f] (Poytress et al 2014)
Abbreviations: RBDD refers to Red Bluff Diversion Dam

4.A.1.3.4 Juvenile Outmigration

Emigrating juvenile winter-run Chinook salmon pass the Red Bluff Diversion Dam beginning as early as mid-July, typically peaking in September, and can continue through March in dry years (Vogel and Marine 1991; NMFS 1997). Many juveniles apparently rear in the Sacramento River below Red Bluff Diversion Dam for several months before they reach the Delta (Williams 2006). From 1995 to 1999, all winter-run Chinook salmon outmigrating as fry passed the Red Bluff Diversion Dam by October, and all outmigrating presmolts and smolts passed the Red Bluff Diversion Dam by March (Martin et al. 2001).

Juvenile winter-run Chinook salmon are present in the Delta primarily from November through early May based on data collected from trawls in the Sacramento River at West Sacramento (river mile 55) (Table 4.A.1-1; USFWS 2006), although the overall timing may extend from September to early May (NMFS 2012). The timing of migration varies somewhat because of changes in river flows, dam operations, seasonal water temperatures, and hydrologic conditions (water year type). Winter-run Chinook salmon juveniles remain in the Delta until they reach a fork length of approximately 118 millimeters and are between 5 and 10 months of age. Distinct emigration pulses from the Delta appear to coincide with high precipitation and increased turbidity (Hood 1990, as cited in U.S.Bureau of Reclamation 2008).

The entire population of the winter-run Chinook salmon must pass through the Delta as emigrating juveniles. Because juvenile winter-run Chinook salmon have been collected at various locations in the Delta (including the CVP/SWP south Delta export facilities), it appears that juveniles likely use a wider range of the Delta for migration and rearing than adults do. Studies using acoustically tagged juvenile and adult Chinook salmon are ongoing to further investigate the migration routes, migration rates, reach-specific mortality rates, and the effects of hydrologic conditions (including the effects of CVP/SWP export operations) on salmon migration through the Delta (Lindley et al. 2008; MacFarlane et al. 2008a; Michel et al. 2008; Perry et al. 2008; Perry et al. 2012; Michel et al. 2013). Juvenile winter-run Chinook salmon likely inhabit Suisun Marsh for rearing, and a recent acoustic tagging study indicates winter-run Chinook may also rear in the Napa River or other bay tributaries (Hearn et al. 2014). Winter-run Chinook salmon juveniles also inhabit the Yolo Bypass, when flooded, using it as rearing and a migratory pathway (Del Rosario et al 2013).

It has been hypothesized that changes in habitat conditions in the Delta over the past century have resulted in reduced juvenile salmon rearing opportunities in the Delta compared to historic conditions when habitat for juvenile salmon rearing was more suitable. Shallow water habitat occurring in floodplains provides for higher abundances of food and warmer temperatures, which promotes rapid growth, presumably resulting in larger out-migrants, which have higher survival rates in the ocean (Sommer et al. 2001). The reduction of floodplain habitat may have significant negative impacts on winter-run Chinook salmon, assuming that survival rates are lower when floodplain habitat is reduced. Emigration to the ocean begins as early as November and continues through May (Fisher 1994; Myers et al. 1998). The importance of the Delta in the life history of winter-run Chinook salmon is not well understood. However, several studies are actively examining the life history patterns of winter-run Chinook salmon using acoustic telemetry and otolith studies.

Additionally, a study of winter-run Chinook salmon juvenile migration showed differences in timing of catch at Knights Landing and subsequent catch at Chipps Island indicating the apparent use of the Delta or habitats just north of the Delta for extended periods (41-117 days) prior to ocean entry (Del Rosario et al. 2013).

4.A.1.3.5 Ocean Behavior

Data from the Pacific States Marine Fisheries Commission Regional Mark Information System database indicate that winter-run Chinook salmon adults are not as broadly distributed along the Pacific Coast as other Central Valley Chinook salmon runs and concentrate in the region between San Francisco and Monterey (NMFS 2010). Winter-run Chinook salmon remain in the ocean environment for 2 to 4 years prior to returning to fresh water to spawn. In the Ocean, they are exposed to many stressors including recreational and commercial harvest, and prey availability due to changes in ocean currents, winds, and climate (Orsi and Davis 2013). Impacts from predators may be variable due to the availability of other prey (Orsi and Davis 2013). Low ocean sea temperatures may delay migration and reduce growth, thereby contributing to higher mortality (Orsi and Davis 2013).

4.A.1.3.6 Status and Trends

Results of fishery monitoring using adult counts at the Red Bluff Diversion Dam (RBDD) fish ladder (1967 through 2000) and carcass surveys (2001 to 2014) have been used to estimate annual adult escapement of winter-run Chinook salmon on the mainstem Sacramento River (Figure 4.A.1-3). Estimates of the winter-run Chinook salmon population (including both male and female salmon) reached nearly 120,000 adult fish in the late 1960s before declining to under 200 fish in the 1990s (Fisher 1994; CDFW 2014). Population abundance remained very low through the mid-1990s, with adult abundance in some years less than 500 fish (CDFW 2014). Beginning in the mid-1990s and continuing through 2006, adult escapement showed a trend of increasing abundance, approaching 20,000 fish in 2005 and 2006. However, recent population estimates of winter-run Chinook salmon spawning upstream of the RBDD have declined since the 2006 peak. The escapement estimate for 2007 through 2014 has ranged from a low of 738 adults in 2011 to a high of 5,959 adults in 2013. The escapement estimate of 738 adults in 2011 was the lowest total escapement estimate since the all-time low escapement estimate of 144 adults in 1994. Poor ocean productivity (Lindley et al. 2009), drought conditions during 2007–2009, and low in-river survival (National Marine Fisheries Service 2011) are suspected to have contributed to the recent decline in escapement of adult winter-run Chinook salmon.

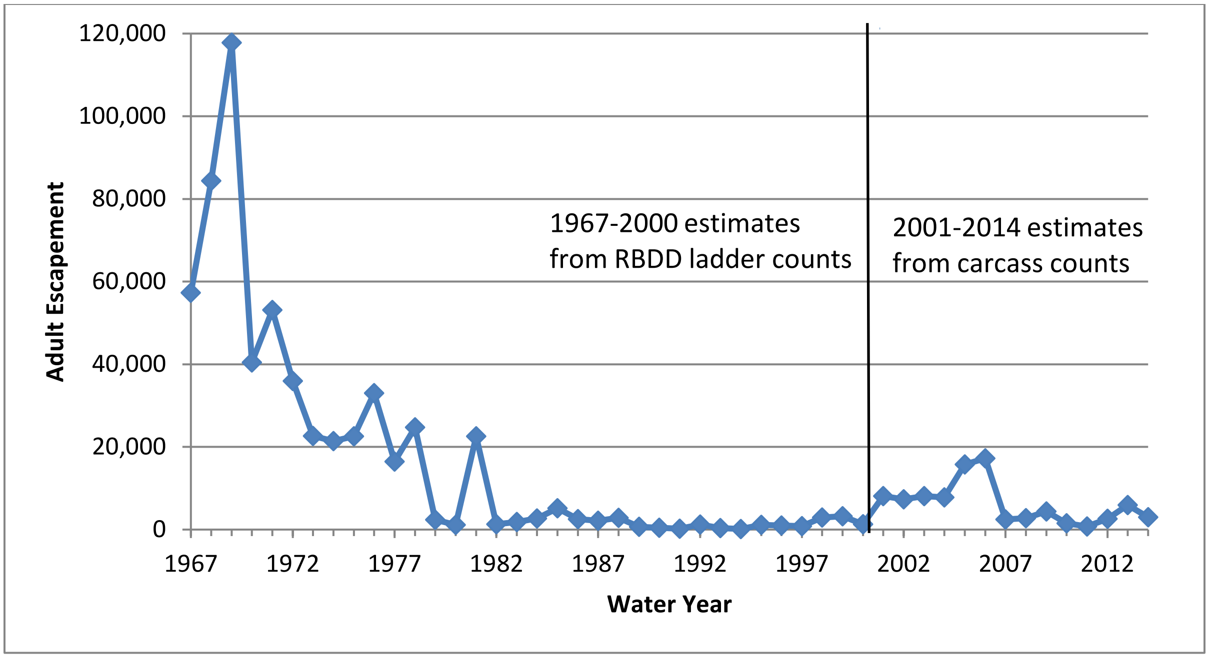

Figure 4.A.1-3. Historical Spawner Escapement of Escapement of Sacramento River Winter-Run Chinook Salmon (1967–2014) (Source: 1967–1969 data from Fisher [1994]; 2009–2014 data are preliminary; from CDFW 2014 and Lehr pers. comm.)

The following factors likely contributed to the increasing trend in adult abundance from the mid-1990s until 2006.

- Improved water temperatures and temperature management in the Shasta Reservoir and the mainstem river downstream of Keswick Dam.
- Improvements in the operations of the Red Bluff Diversion Dam (keeping holding gates open for a longer period).
- Favorable hydrological and ocean rearing conditions.
- Habitat enhancements, reductions in loading of toxic chemicals.
- Improved fish screens on major water diversions.
- Changes in ocean commercial and recreational angling to reduce harvest mortality.

The substantial declines in adult winter-run Chinook salmon escapement since 2006 likely were the result of reduced productivity of near-shore coastal waters and reduced prey availability resulting in poor juvenile salmon growth and high mortality during the juvenile ocean rearing phase (MacFarlane et al. 2008b). In response to the low numbers of adult fall-run Chinook salmon returning to the Central Valley beginning in 2006, commercial and recreational fishing for salmon was curtailed between 2007 and 2009. In 2010, NMFS issued a biological opinion (BiOp) for ocean salmon fisheries effects on Sacramento River winter Chinook salmon which concluded harvest was contributing to a truncated age-distribution (90% of winter Chinook return at age-3) and that “the salmon ocean fishery reduced the reproductive capability of this population, and subsequently the entire ESU, by 10–25% per brood…" (NMFS 2010). NMFS reasoned that if the status of winter-run Chinook salmon remains generally positive that impacts from the salmon ocean fishery would not be expected to negatively affect the abundance and population growth capability of this ESU at a level that would appreciably increase the risk of extinction. However, during times of generally negative patterns in spawner returns or other indications that the status of winter-run Chinook salmon is deteriorating, fishing impacts are likely to increase the probability of extinction of the ESU through losses in population abundance, impacts on diversity, and reductions in population growth rate (NMFS 2010).

Although NMFS proposed that this ESU be downgraded from endangered to threatened status in 2004 (69 FR 33102; June 14, 2004), NMFS decided in its final rule to continue to list the Sacramento River winter-run Chinook salmon ESU as endangered, noting that a key concern of the BRT was the lack of diversity within this ESU and the fact that it is represented by a single extant population at present (70 FR 37160; June 28, 2005). NMFS reconfirmed that the classification as an endangered species was appropriate in its latest 5-year status review in 2011. NMFS concluded that the most recent biological information suggests that the extinction risk of this ESU has increased since the last status review and that several of the listing factors have contributed to the decline, including recent years of drought and poor ocean conditions (NMFS 2011).

4.A.1.4 Threats and Stressors

NMFS issued a final rule on June 28, 2005, concluding that the ESU was still "in danger of extinction throughout all or a significant portion of its range and ... the ESU continues to warrant listing as an endangered species under the ESA" (70 FR 37160). NMFS noted risks associated with the ESU's lack of diversity and spatial structure. In addition, NMFS noted concerns that there is only one extant population, and it is spawning outside of its historical range, in artificially maintained habitat that is vulnerable to drought, climate change, and other catastrophes. There was also a concern over the increasing number of Livingston Stone National Fish Hatchery fish spawning in natural areas, although the duration and extent of this possible introgression was still consistent with a low extinction risk as of 2004 (NMFS 2011). Since 2000, the proportion of hatchery-origin fish spawning in the Sacramento River has generally ranged between 5–10% of the total population, except for in 2005 when it reached approximately 20% of the population, which is consistent with the goals of the hatchery program (NMFS 2011). In addition, recent analyses indicate ocean harvest is curtailing diversity in age-at-maturity (NMFS 2010) and substantially reducing abundance of the ESU (Winship et al. 2014).

The following conditions have been identified as important threats and stressors to winter-run Chinook salmon.

4.A.1.4.1 Reduced Access to and Quantity and Quality of Staging, Spawning, and Egg Incubation Habitat

Access to much of the historical upstream spawning habitat for winter-run Chinook salmon (Table 4.A.1-1) has been eliminated or degraded by artificial structures (e.g., dams and weirs) associated with water storage and conveyance, flood control, and diversions and exports for municipal, industrial, agricultural, and hydropower purposes (Yoshiyama et al. 1998). The construction and operation of Shasta Dam reduced the Sacramento River winter-run Chinook salmon ESU from four independent populations to just one. The remaining available habitat for natural spawners is currently maintained with cool water releases from Shasta and Keswick dams, thereby significantly limiting spatial distribution of this ESU in the reach of the mainstem Sacramento River immediately downstream of the dam. In-progress construction in Battle Creek is creating a 42-mile section of suitable spawning and rearing habitat for winter-run Chinook salmon, although individuals have not yet used the restored habitat for spawning.

Upstream diversions and dams have decreased downstream flows and altered seasonal hydrologic patterns, which have been identified as factors resulting in delayed upstream migration by adults and increased mortality of out-migrating juveniles (Yoshiyama et al. 1998; DWR 2005). Dams and reservoir impoundments and associated reductions in peak flows have blocked gravel recruitment and reduced the flushing of sediments from existing gravel beds, reducing and degrading natal spawning grounds. Furthermore, reduced flows can lower attraction cues for adult spawners, causing straying and delays in spawning (DWR 2005) although there is no evidence to suggest that a reduction in attraction cues is currently a problem. Adult salmon migration delays can reduce fecundity and increase susceptibility to disease and harvest (McCullough 1999).

High flows, such as those released from dams to draw down storage for flood control during heavy runoff periods, have the potential to scour winter-run Chinook salmon redds down to the depth of the eggs and injure eggs or sac-fry in the gravel, or to pile more gravel and fines on top of redds so that alevins are unable to emerge or are suffocated. These same flows are important for maintaining rearing habitat and high-quality spawning gravel. River-specific geomorphic studies evaluated the bedload mobilization flow for the affected rivers. The future probability of occurrence of flow releases exceeding the bedload mobilization flow is based on the historic hydrograph since the respective dam was constructed. This is because scouring flows are generally a result of flood control operations during high runoff periods, which will not likely change in the near future.

Buer (1980) conducted bedload movement experiments by burying a 50-gallon drum in a riffle below Redding. Gravel up to 3 inches in diameter began to accumulate in the barrel at about 25,000 cfs, indicating initiation of surface transport. Painted rocks moved 200 to 300 feet down the riffle at 25,000 cfs. Flows of 40,000 to 50,000 cfs would likely be required to move enough bedload to scour redds (Koll Buer, pers. comm. 2003, as cited in U.S.Bureau of Reclamation 2008).

Probability of occurrence for a release exceeding 25,000 cfs at Keswick Dam is approximately 50% of years and flows in the 40,000 to 50,000 cfs range occur in about 30 to 40% of years (Figure 4.A.1-4). Redds could potentially be scoured in up to 30% of years when flows over 50,000 cfs occur while eggs are in the gravel. The significance to the population is difficult to determine, but based on observations of the amount of scouring that occurs on unregulated rivers (with large salmon runs) versus Central Valley regulated rivers, it seems plausible that long-term negative population effects from redd scouring are probably not significant (U.S. Bureau of Reclamation 2008). The statistical probability of scour in the lower Sacramento River downstream of Shasta and Keswick dams has decreased with dam operation; the 2-year return interval flood has been reduced from 119,000 cfs pre-Shasta Dam to 79,000 cfs post-Shasta Dam (as measured at Red Bluff, Figure 4.A.1-5).

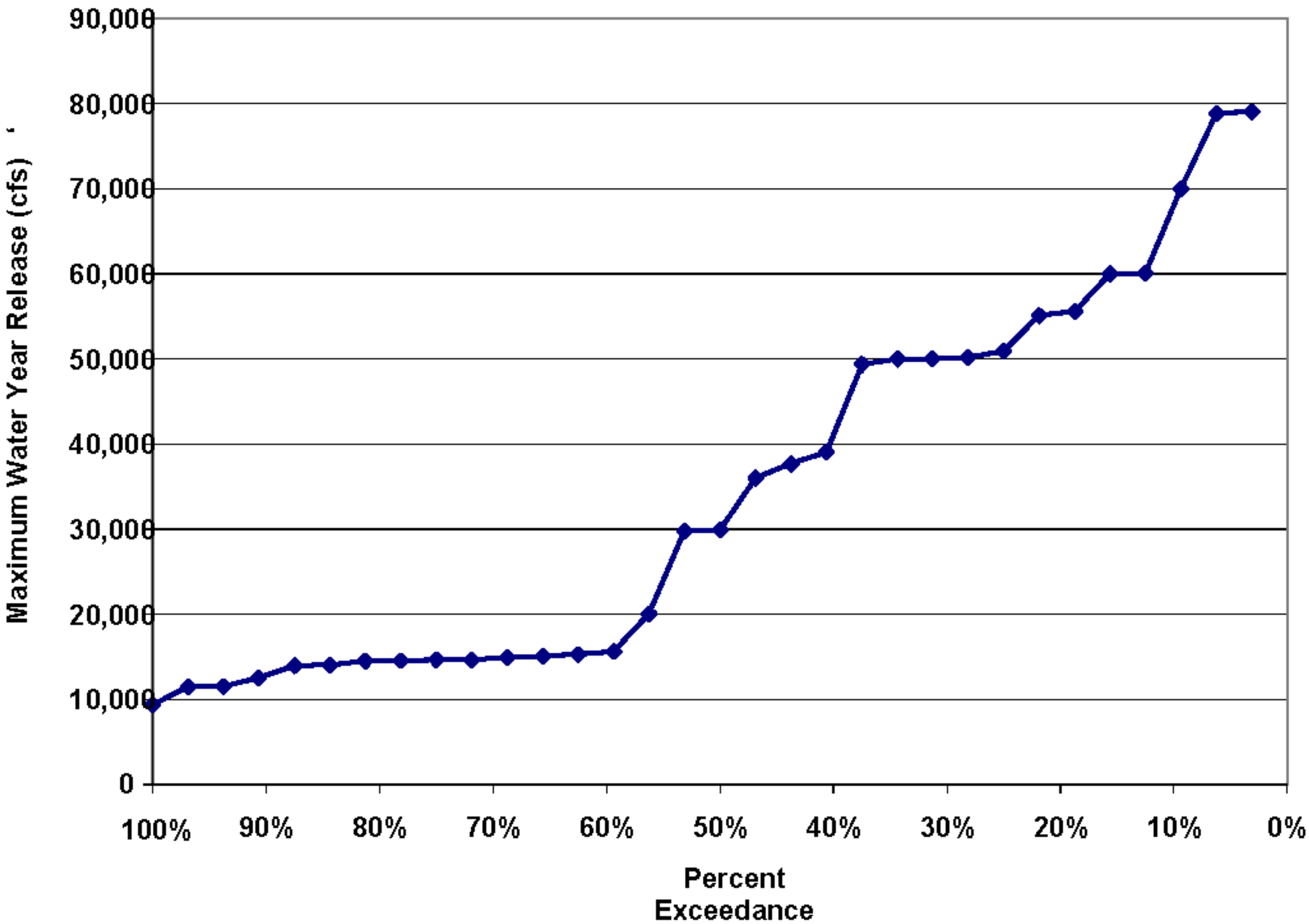

Figure 4.A.1-4. Yearly Probability of Exceedance for Maximum Releases (cfs) from Keswick Dam on the Sacramento River from <u>Historical</u> Dam Operations Records, [water year]-[water year].

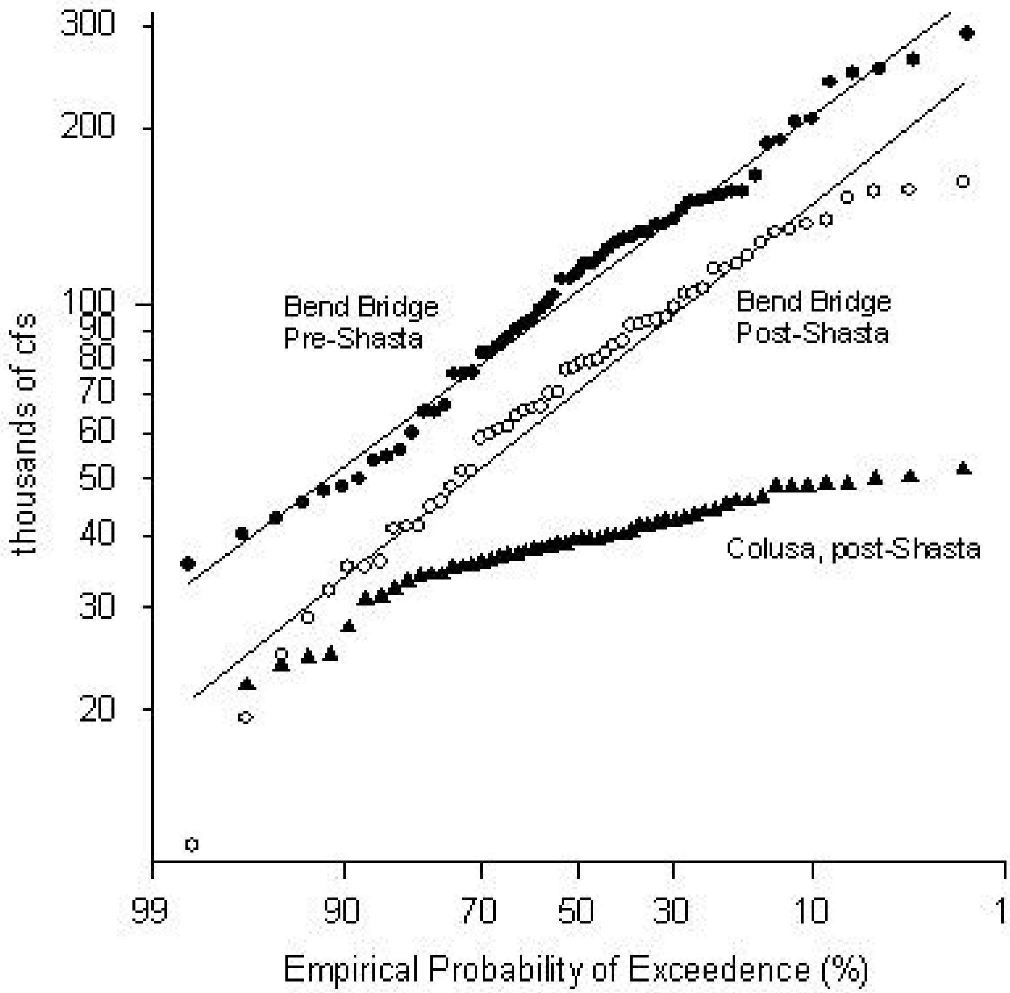

Figure 4.A.1-5. Empirical Flood Frequency Plots for the Sacramento River at Red Bluff (Bend Bridge gauge) for Pre- and Post-Shasta Periods (1879-1943 and 1943-1998, respectively), and Downstream at Colusa for the Post-Shasta Period.
The reduced peak flows at Colusa reflect diversions into the Butte Basin between the two gauges. Data from U.S. Geological Survey internet site (www.usgs.gov), Red Bluff (Bend Bridge) and Colusa gauges. Chart from Calfed 1999.

Flow fluctuations have the potential to dewater salmon redds downstream project reservoirs. Dewatering of winter-run Chinook salmon redds can occur when flows are suddenly reduced back to baseline after water has been released to make room in Shasta Reservoir for floodwater storage. Based on stage discharge relationships at Sacramento River at Bend Bridge gauge (Figure 4.A.1-2), drops in flow of approximately 800 cfs in the low end of the flow range up to about 20,000 cfs have the potential to start drying the shallowest redds 5 inches below the streambed. Most eggs are buried at least five inches below the streambed so an additional 800 cfs (1,600 cfs total) or more reduction could deteriorate hyporheic conditions and begin to impair the potential for successful emergence. Areas of the river away from stream gauges where there is not as much confinement and more spawning activity probably experience less change in stage for a given flow change but the data were not available to evaluate other locations.

Survival of eggs and fry is not strictly a function of water temperatures and flow fluctuations. For example, larger females generally have larger and more numerous eggs (Moyle 2002; Healey 1991), both of which provide reproductive advantages. Larger eggs produce larger juveniles, which tend to have higher survival rates (Quinn 2005) and are more resistant to temperature extremes. Differences in body size may also influence spawning habitat use as larger fish occupy areas with coarser substrate that smaller fish may not be able to use (Healey 1991). Thus, advantages of diversity in age-at-maturity could be especially important in degraded and thermally stressed habitats typical of Central Valley tributaries. As is described elsewhere, harvest pressure can alter diversity in age and size at-maturity (Kendall and Quinn 2011: Lewis et al. 2015). Thus, adverse effects from habitat degradation, flows, and water temperatures can be exacerbated by harvest practices, which select for early maturity.

Table 4.A.1-2. Stage Discharge Relationship in the Sacramento River at Bend Bridge, Gauge 11377100.

Stage, inches	Discharge, cfs
8	4,190
10	4,500
12	5,020
15	5,490
18	5,990
21	6,490
24	6,990
27	7,490
31	7,990
34	8,500
38	9,000
41	9,510
45	10,000
48	10,500
52	11,000
55	11,500
59	12,000
62	12,500
65	13,000
68	13,500
71	14,000
74	14,500
78	15,000
81	15,500
84	16,000
87	16,500
90	17,000
92	17,500
95	18,000
98	18,500
101	19,000
103	19,500
106	20,000
110	21,000
114	22,000
118	23,000
122	24,000
126	25,000
129	26,000
133	27,000
137	28,000
140	29,000
144	30,000

The construction and operation of the Red Bluff Diversion Dam has been identified as one of the primary factors that contributed to the decline in winter-run Chinook salmon abundance that led to listing of the species under the ESA. However, the dam gates were placed in a permanent open position in September 2011, and a new pump facility with a state-of-the-art fish screen was subsequently constructed. The project is expected to benefit both upstream and downstream migration and contribute to a reduction in juvenile predation mortality.

4.A.1.4.2 Reduced Rearing and Out-Migration Habitat

Juvenile winter-run Chinook salmon prefer natural stream banks, floodplains, marshes, and shallow water habitats for rearing and during out-migration. Channel margins throughout the Sacramento River and Delta have been leveed, channelized, and fortified with riprap for flood protection and island reclamation, reducing and degrading the value of natural habitat available for juvenile Chinook salmon rearing (Brandes and McLain 2001). Artificial barriers further reduce and degrade rearing and migration habitat and delay juvenile out-migration. Juvenile out-migration delays can reduce fitness and increase susceptibility to diversion screen impingement, entrainment, disease, and predation. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening and altering the seasonal timing of the hydrograph, reducing the extent and duration of seasonal floodplain inundation and other flow-dependent habitat used by migrating juvenile Chinook salmon (70 FR 52488, September 2, 2005; Sommer et al. 2001; California Department of Water Resources 2005).

Recovery of floodplain habitat in the Central Valley has been found to contribute to increased production in fall-run Chinook salmon (Sommer et al. 2001), but little is known about the potential benefits of recovered floodplains during the migration period for winter-run Chinook salmon, Nonetheless, Sommer et al. (2001) noted that the reduction of floodplain habitat might have significant negative impacts on winter-run Chinook salmon.

Reductions in flow rates have resulted in increased seasonal water temperatures. The potential adverse effects of dam operations and reductions in seasonal river flows, such as delays in juvenile emigration and exposure to a higher proportion of agricultural return flows, have all been identified as factors that could affect the survival and success of winter-run Chinook salmon inhabiting the Sacramento River in the future.

Tidal areas form important rearing habitat for foraging juvenile salmonids. Studies have shown that foraging winter-run Chinook salmon may spend 2 to 3 months in the Delta (Del Rosario et al. 2013). Loss of tidal habitat because of land reclamation facilitated by levee construction is considered a major stressor on juvenile salmonids in the Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) conceptual model (Williams 2010).

Channel margins habitats have been considerably degraded because of the construction of levees and the armoring of their banks with riprap (Williams 2010). Functional shallow-water habitat areas provide refuge from unfavorable hydraulic conditions and predation, as well as foraging habitat for out-migrating juvenile salmonids. Recent research has focused on the use of channel margin habitat by Chinook salmon fry (McLain and Castillo 2010; H. T. Harvey & Associates with PRBO Conservation Science 2011). Benefits for larger Chinook salmon migrant juveniles and steelhead may be somewhat less than for foraging Chinook salmon fry, although the habitat may serve an important function as holding areas during downstream migration (Burau et al. 2007), thereby improving connectivity along the migration route.

Releases of hatchery-origin Chinook salmon from Coleman Fish hatchery into Battle Creek and then into the Sacramento River as recommended by the California HSRG (2012), may indirectly increase competition and reduce the amount of habitat and food available to support rearing winter-run Chinook salmon through increased competition for space and prey items.

4.A.1.4.3 Predation

Predation is an important ecosystem process that helps to structure and maintain fish communities. Predation effects are very difficult to discern in nature because they are typically nonlinear and density-dependent (Bax 1999). Even without human intervention, natural predation rates are affected by spatio-temporal overlap of predators and prey, activity and metabolic needs of predators and prey at different temperatures, efficiency of different types of predators at capturing different prey, and the relative availability of appropriate prey types. Anthropogenic changes to ecosystems can alter these predator-prey dynamics, resulting in artificially elevated predation rates (Pickard et al. 1982; Gingras 1997, as cited in Beamesderfer et al. 2007). Aside from direct human harvest, three factors could affect predation dynamics on juvenile salmon in the action area. These are changes in the species composition and diversity of potential salmon predators through nonnative species introductions, changes in the abundance of potential salmon predators (both of these may or may not be coupled to habitat alteration), and the placement of large structures in migratory pathways of the salmon.

There have been substantial changes in the abundance of several potential Chinook salmon predators over the past 20 to 30 years. These changes could have altered the predation pressure on salmon, but the data needed to determine this have not been collected. A few examples of changes in potential predator abundance are discussed below.

The striped bass is the largest piscivorous fish in the Bay-Delta. Its abundance has declined considerably since at least the early 1970s (Kimmerer et al. 2000). Both striped bass and spring-run and winter-run Chinook were much more abundant during the 1960s (DFG 1998) when comprehensive diet studies of striped bass in the Delta were last reported. During fall and winter 1963–1964, when spring-run Chinook salmon yearlings and juvenile winter-run Chinook salmon would have been migrating through the Delta, Chinook salmon only accounted for 0%, 1%, and 0% of the stomach content volume of juvenile, subadult, and adult striped bass respectively (Stevens 1966). During spring and summer 1964, Chinook salmon accounted for up to 25% of the stomach content volume of subadult striped bass in the lower San Joaquin River, although most values were less than 10%. Presumably most of these spring and summer prey were fall-run since they dominate the juvenile salmon catch during that time of year. Despite lower population levels, striped bass are suspected of having significant predation effects on Chinook salmon near diversion structures (see below).

Although striped bass abundance has decreased considerably, the abundance of other potential Chinook salmon predators may have increased. Nobriga and Chotkowski (2000) reported that the abundance of virtually all centrarchid fishes in the Delta, including juvenile salmon predators like largemouth bass and crappies, had increased since the latter 1970s, possibly because of the proliferation of Brazilian waterweed, *Egeria densa*. The increase in largemouth bass abundance is further corroborated by DFG fishing tournament data (Lee 2000). Predation by centrarchids such as largemouth bass and bluegill on salmon is probably minor because centrarchids are active at higher temperatures than those preferred by salmon so the two species are not likely present in the same areas at the same time. Recent acoustic telemetry studies indicate that mortality rates of juvenile salmonids in the Delta are 80% to 99% (Michel et al. 2013), although the exact cause of mortality is not known.

Predatory fish are known to aggregate around structures placed in the water, where they maximize their foraging efficiency by using shadows, turbulence, and boundary edges. Examples include dams, bridges, diversions, piers, and wharves (Stevens 1966, Vogel et al. 1988, Garcia 1989, Decoto 1978, all as cited in DFG 1998). In addition, predatory fish are common in deeper scour holes, such as at the entry of Clifton Court Forebay and downstream of the Head of Old River (HOR) in the San Joaquin River.

Predation in Clifton Court Forebay (CCF) has also been identified as a substantial problem for juvenile Chinook. Between October 1976 and November 1993, DFG conducted 10 mark and recapture experiments in CCF to estimate prescreen loss (which includes predation) of fishes entrained to the forebay (Gingras 1997, as cited in Beamesderfer et al. 2007). Eight of these experiments involved hatchery-reared juvenile Chinook salmon. Prescreen loss (PSL) rates for juvenile fall-run Chinook ranged from 63% to 99%, and for late-fall-run smolts they ranged from 78% to 99%. These studies were used to establish the standard prescreen loss figures used today. PSL of juvenile Chinook was inversely proportional to export rate, and striped bass predation was implicated as the primary cause of the losses. Although a variety of potential sampling biases confounds the PSL estimates, the results suggest salmon losses are indeed high at the times of year when the studies were conducted.

Predation studies have also been conducted at release sites for fish salvaged from the CVP/SWP Delta pumping facilities (Miranda et al. 2010). Common piscine predators observed at the Horseshoe Bend release site include (in order of abundance): largemouth bass, Sacramento pikeminnow, and striped bass. Avian predators included gulls and cormorants. Overall, results indicate that, although highly variable, predation by fish and birds could have a substantial effect on the number of fish surviving the release.

4.A.1.4.4 Harvest

Central Valley origin Chinook salmon of all races are harvested in commercial and recreational fisheries off the coast of California. Central Valley origin fall-run Chinook salmon are the primary target of this harvest. Harvested Chinook between Point Conception and Bodega Bay were found to be composed of 89–95% Central Valley fall-run Chinook salmon (Winans et al. 2001). More recent studies have shown most Central Valley fall-run Chinook salmon are produced by hatcheries, and are not of natural origin. Barnett-Johnson et al. (2007) analyzed otolith microstructure from harvested Chinook salmon and estimated 90% were of hatchery origin. Palmer-Zwhalen and Kormos (2013) reported data indicating spawning-escapement for Central Valley fall-run Chinook salmon was composed of 75% hatchery origin fish.

Commercial and recreational harvest of winter-run Chinook salmon in the ocean and inland fisheries has been a subject of management actions by the California Fish and Game Commission and the Pacific Fishery Management Council. The primary concern is the incidental harvest of winter-run Chinook salmon as part of fisheries primarily targeting hatchery produced fall- and late fall-run salmon. Natural-origin Chinook salmon stocks are less able to withstand high harvest rates which may be sustainable for hatchery-based stocks (California HSRG 2012).

Commercial fishing for salmon in west coast ocean waters is managed by the Pacific Fishery Management Council and is constrained by time, size, species, and area closures to help provide protections for several ESA-listed stocks, including the winter-run Chinook salmon ESA. Ocean harvest restrictions since 1995 have led to reduced ocean harvest of winter-run Chinook salmon (i.e., CV Chinook salmon ocean harvest index, ranged from 0.55 to nearly 0.80 from 1970 to 1995, and was reduced to 0.27 in 2001). Major restrictions in the commercial fishing industry in California and Oregon were enforced to protect Klamath River coho salmon stocks. Because the fishery is mixed, these restrictions have likely reduced harvest of winter-run Chinook salmon as well. Previous harvest practices are the likely cause of the predominance of 3-year-old spawners, with few (if any) 4- and 5-year-old fish surviving the additional years in the ocean to return as spawners (NMFS April 30, 2012 Memo).

Since 2005, NMFS has issued a new BiOp (NMFS 2010) addressing the ocean harvest impacts on this ESU from commercial and recreational ocean salmon fisheries managed under the Pacific Coast Salmon Fishery Management Plan. The BiOp concluded the fisheries were likely to jeopardize the continued existence of the ESU, and therefore, included a reasonable and prudent alternative (RPA) that required NMFS to develop and implement a new management framework for the ocean fishery addressing impacts to winter-run Chinook salmon before the 2012 ocean salmon fishery season. In the interim, the RPA required implementation of either an increase in size limits or reduction in fishery effort (seasonal closures) in the recreational fishery in 2010 and 2011. NMFS determined that impacts from the fishery needed to be constrained from reaching the levels estimated for brood years 1998 through 2005 (age-3 impacts rates up to 0.21; total spawner reduction rates up to 0.25), due to the significant decline in abundance of winter-run Chinook salmon spawning returns since 2006. A description and evaluation of the related harvest management strategy is provided in (NMFS April 30, 2012 Memo). In summary, the available information indicates that the level of ocean fishery impacts on the age-3 component of this ESU is expected to have been reduced since 2005, but the ocean fishery is still constraining abundance (Winship et al. 2013) and adversely affecting diversity in age-at-maturity, a key factor for viability (NMFS 2010).

The ocean fishery is thought to select against fish that mature later because fish that would do so are vulnerable to harvest for more years (Ricker 1981; Hankin and Healey 1986; Franks and Lackey 2015), and age-at-maturity has moderate heritability (Hankin et al. 1993). As such, reduced ocean harvest would contribute substantially to age-at-maturity diversity and thereby enhance population viability. It is also important to recognize that a downward shift in size and age at maturity also affects fitness by reducing fecundity and reproductive rates (Calduch-Verdiell et al. 2014). Since size and age-at-maturity are heritable, selection for earlier adult maturity leads to a feedback loop in which younger and smaller adults produce offspring that mature earlier at smaller sizes.

Because adult winter-run Chinook salmon hold in the mainstem Sacramento River until spawning during the summer months, they are particularly vulnerable to illegal (poaching) harvest. Various watershed groups have established public outreach and educational programs in an effort to reduce poaching. In addition, CDFW wardens have increased enforcement against illegal harvest of winter-run Chinook salmon. The level and effect of illegal harvest on adult winter-run Chinook salmon abundance and population reproduction is unknown, although the upper Sacramento River was closed to all fishing in 2015.

4.A.1.4.5 Reduced Genetic Diversity and Integrity

Artificial propagation programs conducted for winter-run Chinook salmon conservation purposes (i.e., Livingston Stone National Fish Hatchery) were developed to increase the abundance and diversity of winter-run Chinook salmon and to protect the species from extinction in the event of a catastrophic failure of the wild population. It is unclear what the effects of the hatchery propagation program are on the productivity and spatial structure of the Sacramento River winter-run Chinook salmon ESU (i.e., genetic fitness and productivity). One of the primary concerns with hatchery operations is the genetic introgression by hatchery origin fish that spawn naturally and interbreed with local natural populations (USFWS 2001; Bureau of Reclamation 2004; Goodman 2005). It is now recognized that Central Valley hatcheries are a significant and persistent threat to wild Chinook salmon and steelhead populations and fisheries (NMFS 2009a, California HSRG 2012). Such introgression introduces maladaptive genetic changes to the wild winter-run Chinook salmon stocks and may reduce overall fitness (Myers et al. 2004; Araki et al. 2007). Taking egg and sperm from a large number of individuals is one method to ameliorate genetic introgression, but artificial selection for traits that assure individual success in a hatchery setting (e.g., rapid growth and tolerance to crowding) are unavoidable (Bureau of Reclamation 2004).

Hatchery-origin winter-run Chinook salmon from Livingston Stone National Fish Hatchery represent more than 5% of the natural spawning run in recent years and as high as 18% in 2005 (NMFS April 30, 2012 Memo). Lindley et al. (2007) recommended reclassifying the winter-run Chinook population extinction risk as moderate, rather than low, if hatchery introgression exceeds about 15% over multiple generations of spawners. Since 2005, however, the percentage of hatchery fish has been consistently below 15% of the spawning run (NMFS April 30, 2012 Memo).

Investigations are continuing to evaluate the genetic characteristics of winter-run Chinook salmon, improve genetic management of the artificial propagation program, evaluate the minimum viable population size that would maintain genetic integrity in the population, and explore methods for establishing additional independent winter-run Chinook salmon populations as part of recovery planning and conservation of the species.

4.A.1.4.6 Entrainment

The vulnerability of juvenile winter-run Chinook salmon to entrainment and salvage at CVP/SWP export facilities varies in response to multiple factors, including the seasonal and geographic distribution of juvenile salmon in the Delta, operation of Delta Cross Channel gates, through-Delta survival hydrodynamic conditions i.e. instantaneous velocities and instantaneous velocity fields occurring in both the north Delta (i.e., Sacramento River flows and tidal stage) and the central and southern regions of the Delta (e.g., Old and Middle Rivers), and export rates at project and nonproject facilities. The loss of fish to entrainment mortality has been hypothesized as an impact on Chinook salmon populations (Kjelson and Brandes 1989).

Between February and April, juvenile winter-run Chinook salmon may be distributed in the central Delta where they have an increased risk of entrainment and salvage. Nearly half of the average annual salvage occurs in March (NMFS April 30, 2012 Memo).

The number of Sacramento River winter-run Chinook lost at the Delta pumping facilities each year is found to be proportional to the amount of Sacramento River flow diverted into the Central Delta during the time that juvenile winter-run Chinook are emigrating through the lower Sacramento River. The proportion of flow diverted into the interior Delta during December and January is significantly influenced by the position of the DCC gates and is correlated to subsequent loss of winter-run Chinook juveniles at the Delta pumping facilities in subsequent months (Low and White 2006).

Tidally averaged flow (or net flow) in Old and Middle rivers (OMR flows) are often negative as a result of export through the Federal and state export facilities. The hydrodynamic conditions associated with negative OMR flows have been hypothesized by NMFS (2009b) to be associated with increased southward movement of emigrating juveniles in those channels, delayed emigration through the Delta, and directly or indirectly increasing vulnerability to the many stressors within the central and south Delta. However, published science and expert panels completed since 2009 have not supported this view (e.g., Monismith et al. 2014, Anderson et al. 2013). Nevertheless, previous studies have observed increased entrainment of tagged salmonids at the CVP/SWP facilities when exports (NMFS 2009b). Recent independent science reviews have observed numerous parameters that influence juvenile salmonid movement. These include instantaneous velocities, which are perceived by the fish in its immediate surrounding environment, detection of chemical constituents in the water by chemo-sensory organs that elicit migratory behavioral responses, and spatial distribution of the migrating fish across the river channel in the vicinity of junctions that affect ultimate route selection (Anderson et al. 2013; Monismith et al. 2014). In addition, Cavallo et al. (2015) showed that exports exerted little influence on routing at junctions leading to the South Delta, with the exception of the HOR at lower San Joaquin River inflows.

CVP/SWP exports have been shown to affect water velocities and direction at locations nearer to the export facilities. Farther away from the export facilities, there is considerably smaller influence on instantaneous velocities within the San Joaquin River channel (Cavallo et al. 2015).

Chinook salmon interact with complex velocity fields during both upstream adult and downstream juvenile migration through the Delta. Where velocity fields change as a result of CVP/SWP export operations during the period that salmon are migrating through Delta channels it may contribute to the use of false migration pathways, delays in migration, or increased movement of migrating salmon toward the export facilities leading to an increase in entrainment risk. During the past several years, additional investigations have been designed using radio or acoustically tagged juvenile Chinook salmon to monitor migration behavior through the Delta channels and to assess the effects of changes in hydraulic cues and CVP/SWP export operations on migration (San Joaquin River Group Authority 2010; SJRG 2011; SJRG 2013; Delaney et al. 2014; Cavallo et al. 2015). These studies are ongoing.

Incidental take of juvenile winter-run Chinook salmon at the CVP/SWP export fish salvage facilities is routinely monitored and reported as part of export operations. Salvage monitoring and the protocol for identifying juvenile winter-run Chinook salmon from other Central Valley Chinook salmon have been refined over the past decade. Run identification was originally determined based on the length of each fish and the date it was collected. Subsequent genetic testing has been used to refine species identification. Methods for estimating juvenile winter-run

Chinook salmon production each year (year class strength) have been developed that take into account the number of adults spawning in the river from carcass surveys, hatching success based on a consideration of water temperatures and other factors, and estimated juvenile survival. Authorized incidental take can then be adjusted each year (1% to 2% of juvenile production) to reflect the relative effect of take at a population level rather than based on a predetermined level that does not reflect year-to-year variation in juvenile production in the Sacramento River.

In addition to CVP/SWP exports, there are more than 2,200 small water diversions (mostly unscreened) throughout the Delta, including unscreened diversions located on the tributary rivers (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978; Schneeberger and Jude 1981; Zeitoun et al. 1981; Weisberg et al. 1987). Many juvenile winter-run Chinook salmon migrate downstream through the Delta during the late winter or early spring when many of the agricultural irrigation diversions are not operating or are only operating at low levels. Juvenile winter-run Chinook salmon also migrate primarily in the upper part of the water column, reducing their vulnerability to unscreened diversions located near the channel bottom. In a study by Vogel (2013) that spanned 4 years and 12 agriculture diversions relatively few Chinook salmon were entrained into the irrigation canals monitored. The study confirmed that important determinants of salmon entrainment likely include initial timing of irrigation diversions in the spring, hydrologic conditions preceding irrigation diversions, and the natural emigration timing of salmon in relation to springtime diversion of water. Irrigation diversions in the middle-lower Sacramento River occur mostly in the late-spring and summer months when this area is not heavily used for rearing, thus accounting for the low numbers of juveniles salmon entrained. The effect of entrainment mortality on the population dynamics and overall adult abundance of winter-run Chinook salmon is not well understood. While the above study (Vogel 2013) doesn’t give definitive answers, it does support a hypothesis that during different water year types (dry, below normal and wet) the overlap between normal diversion timing (Apr/May thru September) and out-migration patterns in general lead to low numbers of juvenile salmon (Fall/Late-fall-run) entrained. During very dry (drought) years there could be more overlap with winter-run out-migration if diversions were started much earlier (Feb/Mar), but even this would be on the tail end of winter-run out-migration.

Power plants in the Delta have the ability to impinge and entrain juvenile Chinook salmon on the existing cooling water system intake screens. However, use of cooling water is currently low with the retirement of older units. Furthermore, newer units are being equipped with a closed-cycle cooling system that virtually eliminates the risk of impingement of juvenile salmon.

Winter-run Chinook salmon may move into the Colusa Drain via Yolo Bypass into the Knights Landing Ridge-cut or up the Sacramento River, then moving through the Knights Landing outfall gates. Once in the canal fish migrate upstream until barriers are reached that prevent further migration. Unless rescued at these points they die and are lost to the population. In 2015 a pickett weir was installed in front of the Knights Landing Outfall Gates that should prevent most fish from moving through the radial gates.

Besides mortality, salmon fitness may be affected by delays in out-migration of smolts caused by altered instantaneous velocities... Delays in migration resulting from water management related to CVP/SWP operations can make juvenile salmonids more susceptible to many of the threats and stressors discussed in this section, such as predation, entrainment, angling, exposure to poor water quality, and disease. The quantitative relationships among changes in Delta hydrodynamics, the behavioral and physiological response of juvenile salmon, and the increase or decrease in risk associated with other threats is increasingly better understood, but is currently the subject of ongoing investigations and analyses.

4.A.1.4.7 Exposure to Toxins

Inputs of toxins into the Delta watershed include agricultural drainage and return flows, municipal wastewater treatment facilities, and other point and nonpoint discharges (Moyle 2002). These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to affect fish health and condition, and adversely affect salmon distribution and abundance. Toxic chemicals have the potential to be widespread throughout the Sacramento River and Delta, or may occur on a more localized scale in response to episodic events (e.g., stormwater runoff and point source discharges). Agricultural return flows are widely distributed throughout the Sacramento River and the Delta, although dilution flows from the rivers may reduce chemical concentrations to sublethal levels. Toxic algae (e.g., Microcystis) have also been identified as a potential factor adversely affecting salmon and other fish. Exposure to these toxic materials has the potential to directly and indirectly adversely affect salmon distribution and abundance.

Concern regarding exposure to toxic substances for Chinook salmon includes both waterborne chronic and acute exposure, but also bioaccumulation and chronic dietary exposure. For example, selenium is a naturally occurring constituent in agricultural drainage water return flows from the San Joaquin River that is then dispersed downstream into the Delta (Nichols et al. 1986). Exposure to selenium in the diet of juvenile Chinook salmon has been shown to result in toxic effects (Saiki and Lowe 1987; Hamilton et al. 1986, 1990; Hamilton and Buhl 1990). Selenium exposure has been associated with agricultural and natural drainage in the San Joaquin River basin and petroleum refining operations adjacent to San Pablo and San Francisco Bays.

Other contaminants of concern for Chinook salmon include, but are not limited to, mercury, copper, oil and grease, pesticides, herbicides, ammonia, and localized areas of depressed dissolved oxygen (e.g., Stockton Deep Water Ship Channel and return flows from managed freshwater wetlands). As a result of the extensive agricultural development in the Central Valley, exposure to pesticides and herbicides has been identified as a significant concern for salmon and other fish species in the Delta (Bennett et al. 2001). In recent years, changes have been made in the composition of herbicides and pesticides used on agricultural crops in an effort to reduce potential toxicity to aquatic and terrestrial species. Modifications have also been made to water system operations and discharges related to agricultural wastewater discharges (e.g., agricultural drainage water system lock-up and holding prior to discharge) and municipal wastewater treatment and discharges. Ammonia released from the City of Stockton Wastewater Treatment Plant contributes to the low dissolved oxygen conditions in the adjacent Stockton Deep Water Ship Channel. In addition to the adverse effects of the lowered dissolved oxygen on salmonid physiology, ammonia is toxic to salmonids at low concentrations. Actions have been implemented to remedy this source of ammonia, by modifying the treatment train at the wastewater facility (NMFS 2012). Concerns remain, however, regarding the toxicity of contaminants such as pyrethroids that adsorb to sediments and other chemicals (e.g., including selenium and mercury, as well as other contaminants) on salmon.

Mercury and other metals such as copper have also been identified as contaminants of concern for salmon and other fish, as a result of direct toxicity and impacts related to acid mine runoff from sites such as Iron Mountain Mine (U.S. Environmental Protection Agency 2006). The potential problems include tissue bioaccumulation that may adversely affect the fish, but also represent a human health concern (Gassel et al. 2008). These materials originate from a variety of sources including mining operations, municipal wastewater treatment, agricultural drainage in the tributary rivers and Delta, nonpoint runoff, natural runoff and drainage in the Central Valley, agricultural spraying, and a number of other sources.

The State Water Resources Control Board (State Water Board), Central Valley Regional Water Quality Control Board (CVRWQCB), U.S. Environmental Protection Agency (EPA), U.S. Geological Survey (USGS), California Department of Water Resources (DWR), and others have ongoing monitoring programs designed to characterize water quality conditions and identify potential toxins and contaminant exposure to Chinook salmon and other aquatic resources in the Delta. Programs are in place to regulate point source discharges as part of the National Pollutant Discharge Elimination System (NPDES) program, as well as programs to establish and reduce total daily maximum loads of various constituents entering the Delta. Changes in regulations have also been made to help reduce chemical exposure and reduce the adverse impacts on aquatic resources and habitat conditions in the Delta. These monitoring and regulatory programs are ongoing. Regulations and changes in monitoring and management of agricultural pesticide and herbicide chemicals and their application, education on the effects of urban runoff and chemical discharges, and refined treatment processes have been adopted over the past several decades in an effort to reduce the adverse effects of chemical pollutants on salmon and other aquatic species.

In the final listing determination of the ESU, acid mine runoff from Iron Mountain Mine, located adjacent to the upper Sacramento River, was identified as one of the main threats to winter-run Chinook salmon (70 FR 37160, June 28, 2005),). Acid mine drainage, including elevated concentrations of metals, produced from the abandoned mine degraded spawning habitat of winter-run Chinook salmon and resulted in high mortality. Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (Bureau of Reclamation 2004). EPA's Iron Mountain Mine remediation program and 2002 restoration plan has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s. Pollution from Iron Mountain Mine is no longer considered a main factor threatening the Sacramento River winter-run Chinook salmon ESU.

Concern has been expressed regarding the potential to resuspend toxic materials into the water column where they may adversely affect salmon through seasonal floodplain inundation, habitat construction projects, channel and harbor maintenance dredging, and other means. For example, mercury deposits exist at a number of locations in the Central Valley and Delta, including the Yolo Bypass. Seasonal inundation of floodplain areas, such as in the Yolo Bypass, has the potential to create anaerobic conditions that contribute to the methylation of mercury, which increases toxicity. Additionally, there are problems with scour and erosion of these mercury deposits by increased seasonal flows. Similar concerns exist regarding creating aquatic habitat by flooding Delta islands or disturbance created by levee setback construction or other habitat enhancement measures. The potential to increase toxicity as a result of habitat modifications designed to benefit aquatic species is one of the factors that needs to be considered when evaluating the feasibility of habitat enhancement projects in the Central Valley.

Sublethal concentrations of toxics may interact with other stressors on salmonids, such as increasing their vulnerability to mortality as a result of exposure to seasonally elevated water temperatures, predation, or disease (Werner 2007). For example, Clifford et al. (2005) found in a laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common pyrethroid, esfenvalerate, were more susceptible to the infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Although not tested on winter-run Chinook salmon, a similar response is likely.

4.A.1.4.8 Increased Water Temperature

Water temperature is among the physical factors that affect the value of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive life stages, such as during incubation or rearing. The Central Valley is the southern limit of Chinook salmon geographic distribution and increased water temperatures are often recognized as an important stressor to California populations.

The tolerance of winter-run Chinook salmon to water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors, such as predator avoidance (Myrick and Cech 2004; Reclamation 2004). Higher water temperatures can lead to physiological stress, reduced growth rates, prespawning mortality, reduced spawning success, and increased mortality of salmon (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2008). Exposure to seasonally elevated water temperatures may occur as a result of reductions in flow, as a result of upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate, and solar radiation.

The recommendations included in this Biological Assessment (BA) were developed by Boles et al. (1988) based on previous temperature studies of Chinook salmon and other salmonids (Table 4.A.1-3). An overview of temperature effects on Chinook salmon follows.

Table 4.A.1-3. Recommended Water Temperatures for All Life Stages of Chinook Salmon in Central Valley Streams as Presented in Boles et al. (1988).[a]

Life Stage	Temperature (°F)
Migrating adult	<65
Holding adult	<60
Spawning	53 to 57.5[b]
Egg incubation	<55
Juvenile rearing	53 to 57.5[c]
Smoltification	<64[d]

[a] The lower thermal limit for most life stages was about 38°F.
[b] Can have high survival when spawned at up to 60°F, provided temperatures drop quickly to less than 55°F.
[c] Temperature range for maximum growth rate based on Brett (1952, as cited in Boles et al. 1988).
[d] Marine and Cech 2004
Note: °F = degrees Fahrenheit.

The temperature recommendations for adult holding and spawning, and for egg incubation were based on laboratory studies of Central Valley fall-run Chinook egg survival (Seymour 1959). Egg mortality was high at constant temperature of 60°F, but was considerably reduced at temperatures between 55°F and 57.5°F. However, sac-fry mortality remained very high (greater than 50%) at temperatures above 56°F, presumably due to “aberrations in sequential physiological development.” These were long-duration experiences that are not representative of river conditions. Table 4.A.1-4 shows the relationship between water temperature and mortality of Chinook eggs and pre-emergent fry compiled from a variety of studies. This is the relationship used for comparing egg mortality between scenarios. USFWS (1998) conducted studies to determine Sacramento River winter-run and fall-run Chinook salmon early life temperature tolerances. They found that higher alevin mortality can be expected for winter-run Chinook salmon between 56°F and 58°F. Mortality at 56°F was low and similar to fall-run Chinook salmon mortality at 50°F. Their relationships between egg and pre-emergent fry mortality and water temperature were about the same as that used in the mortality model in this BA (Table 4.A.1-4).

Table 4.A.1-4. Relationship between Water Temperature and Mortality of Chinook Salmon Eggs and Pre-emergent Fry used in the Reclamation Egg Mortality Model.

Water Temperature (EF)[a]	Egg Mortality[b]	Instantaneous Daily Mortality Rate (%)	Pre-Emergent Fry Mortality[b]	Instantaneous Daily Mortality Rate (%)
41–56	Thermal optimum	0	Thermal optimum	0
57	8% @ 24 days	0.35	Thermal optimum	0
58	15% @ 22 days	0.74	Thermal optimum	0
59	25% @ 20 days	1.40	10% @ 14 days	0.75
60	50% @ 12 days	5.80	25% @ 14 days	2.05
61	80% @ 15 days	10.70	50% @ 14 days	4.95
62	100% @12 days	38.40	75% @ 14 days	9.90
63	100% @11 days	41.90	100% @ 14 days	32.89
64	100% @ 7 days	65.80	100% @10 days[c]	46.05

[a] This mortality schedule was compiled from a variety of studies each using different levels of precision in temperature measurement, the lowest of which was whole degrees Fahrenheit ($\pm$0.5°F). Therefore, the level of precision for temperature inputs to this model is limited to whole degrees Fahrenheit.

[b] These mortality schedules were developed by the USFWS and DFG for use in evaluation of Shasta Dam temperature control alternatives in June 1990 (Richardson et al. 1990, as cited in U.S. Bureau of Reclamation 2008)

[c] This value was estimated similarly to the preceding values but was not included in the biological assumptions for Shasta outflow temperature control FES (Reclamation 1991, as cited in U.S. Bureau of Reclamation 2008).

Temperature compliance points (Bend Bridge and Balls Ferry) vary by water year type and date between April 15 and October 31 for winter-run Chinook salmon spawning, incubation, and rearing. The objective is to meet a daily average temperature of 56°F for incubation and 60°F for rearing. After October 31, natural cooling generally provides suitable water temperatures for all Chinook life cycles.

The theoretical upper lethal temperature that Sacramento River Chinook salmon can tolerate has been reported as 78.5°F (Orsi 1971, as cited in Boles et al. 1988). However, this result must be interpreted with several things in mind.

First, the theoretical maximum corresponds to the most temperature-tolerant individuals. It is not a generality that can be applied to an entire stock. Second, it is only a 48-hour LT 50 (lethal time for 50% mortality). This means it is a temperature that can only be tolerated for a short period. It does not indicate a temperature at which a Chinook could feed and grow. Third, indirect mortality factors (for example, disease and predation) would likely lead to increases in total mortality at temperatures well below this theoretical laboratory-derived maximum. For example, Banks et al. (1971, as cited in Boles et al. 1988) found Chinook growth rates were not much higher at 65°F than at 60°F, but the fish had higher susceptibility to disease at 65°F. Subacute and sublethal temperature thresholds have been identified for Central Valley Chinook salmon by Marine and Cech (2004). Sublethal impairment of predator avoidance, smoltification, and disease begins in the range of about 64° to 68°F.

Myrick and Cech (2001) show that Chinook salmon that complete juvenile and smolt phases in the 50 to 62°F range are optimally prepared for saltwater survival. Marine and Cech (2004) identified a smoltification threshold of <64 F for Central Valley Chinook salmon.

It is also important to note that operation of CVP/SWP facilities cannot influence (1) the water temperatures on many of the tributaries to the Sacramento and San Joaquin Rivers or (2) those other factors that affect water temperatures that are unrelated to the appropriation of water for use by the CVP/SWP.

The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool in Shasta Reservoir, has reduced many of the temperature issues on the Sacramento River and has been specified in Reclamation's water right (SWRCB WR Order 90-05 and 91-01).. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows further downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. As a result of the longitudinal gradient of seasonal water temperatures, the coldest temperatures and best areas for winter-run Chinook salmon spawning and rearing are typically located immediately downstream of Keswick Dam.

The effects of climate change and global warming patterns, in combination with changes in precipitation and seasonal hydrology in the future, have been identified as important factors that may adversely affect the health and long-term viability of winter-run Chinook salmon (Crozier et al. 2008). The rate and magnitude of these potential future environmental changes, and their effect of habitat value and availability for winter-run Chinook salmon, however, are subject to a high degree of uncertainty.

4.A.1.5 Description of Viable Salmonid Population (VSP) Parameters

NMFS measures the conservation status of salmonids, with the viable salmonid population (VSP) framework and uses it to identify the attributes needed to assess the effects of management and conservation actions. The framework is known as the VSP concept (McElhany et al. 2000). The VSP concept measures population performance in term of four key parameters: abundance, population growth rate, spatial structure, and diversity.

4.A.1.5.1 Abundance

The NMFS (2014) Recovery Plan set out criteria for moderate and low risk of extinction. Moderate risk criteria put census population size at 250 to 2,500 adults, or an effective population size of 50 to 500 adults. Low risk would need to have a census population size of greater than 2,500 adults, or an effective population size that is greater than 500 adults.

The historical abundance of winter-run Chinook salmon prior to commercial harvesting is difficult to quantify, in part, because the distinct nature of the run was not recognized by early workers (Yoshiyama et al. 1998). The earliest estimates of run-size are based on monthly commercial catch data for 1916–1957. During this period, the annual run-size of winter-run Chinook salmon is estimated to have numbered between 200 and 91,840 adults, with abundance exceeding 20,000 adults in 30 of the 42 years in this period (Yoshiyama et al. 1998). Although these abundance estimates are based on a number of assumptions, it is believed that these estimates are conservative because they excluded the catch data for some months when winter-run Chinook salmon were mixed with the catches of other runs , primarily late-fall and spring-run Chinook salmon (Yoshiyama et al. 1998). Generally, the historical annual run-size of winter-run Chinook salmon is thought to have numbered in the high tens of thousands at a minimum and perhaps occasionally exceeding 100,000 adults in some years (Yoshiyama et al. 1998), although Fisher (1994) estimated that historical maximum spawner abundance may have approached 200,000 fish.

As discussed previously under "*Status and Trends*", estimates of the winter-run Chinook salmon population reached nearly 120,000 adult fish in the late 1960s before declining to less than 200 fish in the 1990s (Fisher 1994; CDFW 2014) (Table 4.A.1-3). Abundance remained very low through the mid-1990s, with adult abundance in some years less than 500 fish, but then showed a trend of increasing abundance with a peak approaching 20,000 fish in 2005 and 2006 (CDFW 2014). However, population estimates since the 2006 peak (17,296) have declined, with a low of 827 in 2011 and average escapement of 2,013 over eight years. Reasons for decline include less favorable ocean and freshwater conditions for early life history stages, which when coupled with consistent harvest rates leads to less abundance (NMFS 2010).

Hatchery winter-run Chinook salmon have been released by Coleman and Livingston Stone National Fish Hatcheries since 1959 (Table 4.A.1-5). Livingston Stone National Fish Hatchery was constructed at the base of Shasta Dam in 1997 to help restore natural production of winter-run Chinook salmon in the upper Sacramento River through an integrated-recovery type program (i.e., hatchery winter-run Chinook salmon are managed to be integrated into the natural population of winter-run Chinook salmon in the upper Sacramento River). As a conservation hatchery, Livingston Stone National Fish Hatchery was designed to overcome problems at Coleman National Fish Hatchery, including summer water quality issues and lack of a natal water source. In addition, because of concerns over genetic introgression within the winter-run Chinook salmon population, the following best management practices are employed by the hatchery:

1. Each adult used as broodstock is genotyped prior to spawning to confirm that it is a winter-run Chinook salmon;
2. Only a limited number of spawners are used (based on the effective population size); and
3. Only adults of natural-origin are used as broodstock (since 2009).

All hatchery produced winter-run Chinook salmon have originated from Livingston Stone National Fish Hatchery since 1998 (USFWS 2011). Generally, the Livingston Stone National Fish Hatchery produces approximately 200,000 winter-run Chinook salmon each year; however, production in 2015 (brood year 2014) was increased to over 600,000 fish to compensate for expected losses in natural production as a result of current drought conditions.

Table 4.A.1-5. Number of Juvenile Sacramento River Winter-Run Chinook Salmon Released by Coleman and Livingston Stone National Fish Hatcheries, 1959–2015.

Release Year	Brood Year	Number Released	Release Location
1959	1958	3,117	Sacramento River
1963	1962	34,516	Sacramento River
1964	1963	73,000	Sacramento River
1966	1965	53,000	Sacramento River
1967	1966	4,300	Sacramento River
1968	1967	16,176	Sacramento River
1979	1978	9,942	Battle Creek
1983	1982	11,548	Battle Creek
1990	1989	3,203	Sacramento River
1991	1990	1,286	Sacramento River
1992	1991	11,153	Sacramento River
1993	1992	26,433	Sacramento River
1994	1993	18,723	Sacramento River
1995	1994	43,346	Sacramento River
1995	1995	51,267	Sacramento River
1997	1996	4,718	Sacramento River
1998	1997	21,271	Sacramento River
1999	1998	153,908	Sacramento River
2000	1999	30,840	Sacramento River
2001	2000	166,207	Sacramento River
2002	2001	61,952	Sacramento River
2003	2002	233,612	Sacramento River
2004	2003	218,517	Sacramento River
2005	2004	168,260	Sacramento River
2006	2005	173,343	Sacramento River
2007	2006	196,268	Sacramento River
2008	2007	71,883	Sacramento River
2009	2008	146,211	Sacramento River
2010	2009	198,582	Sacramento River
2011	2010	123,857	Sacramento River
2012	2011	194,000	Sacramento River
2013	2012	182,662	Sacramento River
2014	2013	193,000*	Sacramento River
2015	2014	640,000*	Sacramento River

Notes:
* Estimated
1959–2009 release data from USFWS (2011).
2010–2011 release data from California Hatchery Review Project, Livingston Stone National Fish Hatchery, June 2012.
2012–2013 release data from Data Assessment Team (DAT) conference notes for 2012 and 2013, respectively.
2014–2015 release data from NMFS winter-run Chinook salmon juvenile production estimate letter to Reclamation for 2014 and 2015, respectively.

4.A.1.5.2 Productivity

Two methods are used to estimate the natural-origin juvenile production of winter-run Chinook salmon in the Sacramento River upstream of the RBDD: USFWS's juvenile production index (JPI) method (using rotary screw traps) and NMFS's juvenile production estimate (JPE) method (using carcass surveys) (Table 4.A.1-6). The two methods produce statistically similar results (Poytress and Carillo 2012). Based on the JPI and JPE methods, the production of juvenile winter-run Chinook salmon in the Sacramento River upstream of RBDD is estimated to have averaged 3,890,442 juveniles from 1995 through 2010 (excluding 2000 and 2001 when rotary screw trapping was not conducted) and 4,476,633 juveniles from 1996 through 2010 (carcass surveys began in 1996), respectively (Poytress and Carillo 2012). Both JPI and JPE methods indicate that natural-origin juvenile production in the upper Sacramento River peaked during the mid-2000s and has generally been below average since 2006.

Table 4.A.1-6. Estimates of Natural-Origin Juvenile Production as Calculated using Juvenile Production Index and Juvenile Production Estimate Methods (Poytress and Carrillo 2012).

Brood-year	Fry-equivalent JPI	Fry-equivalent JPE
1995	1,816,984	ND
1996	469,183	550,872
1997	2,205,163	1,386,346
1998	5,000,416	4,676,143
1999	1,366,161	1,490,249
2000	ND	4,946,418
2001	ND	5,643,635
2002	8,205,609	6,964,626
2003	5,826,672	6,181,925
2004	3,758,790	2,786,832
2005	8,941,241	12,109,474
2006	7,301,362	11,818,006
2007	1,642,575	1,864,521
2008	1,371,735	1,952,614
2009	4,993,787	3,728,444
2010	1,566,507	1,049,385
Mean	3,890,442	4,476,633
ND=No data		

Long-term population growth and temporal variation are important in analyzing a population's extinction subtlety (Lande 1993, 1998, Middleton and Nisbet 1997). Using this assumption past performance provides a useful predictor of future population dynamics. Populations should exhibit future tendencies that are consistent with those observed in the past in terms of the mean trajectory and variation exhibited over time. Cohort replacement rates (CRR) are indications of whether a cohort is replacing itself in the next generation (Table 4.A.1-7)

Table 4.A.1-7. Sacramento River Winter-run Chinook Salmon Population Estimates from CDFW Grand Tab (2014) with Corresponding Cohort Replacement Rates for Years since 1986.

Return Year	Adult Population Estimate[a]	Cohort Replacement Rate[b]	Juvenile Production Estimate (JPE)[c]
1986	2,596		
1987	2,185		
1988	2,878		
1989	696	0.27	
1990	430	0.20	
1991	211	0.07	
1992	1,240	1.78	40,100
1993	387	0.90	273,100
1994	186	0.88	90,500
1995	1,297	1.05	74,500
1996	1,337	3.45	338,107
1997	880	4.73	165,069
1998	2,992	2.31	138,316
1999	3,288	2.46	454,792
2000	1,352	1.54	289,724
2001	8,224	2.75	370,221
2002	7,441	2.26	1,864,802
2003	8,218	6.08	2,136,747
2004	7,869	0.96	1,896,649
2005	15,839	2.13	881,719
2006	17,296	2.10	3,556,995
2007	2,542	0.32	3,890,534
2008	2,830	0.18	1,100,067
2009	4,537	0.26	1,152,043
2010	1,596	0.63	1,144,860
2011	827	0.29	332,012
2012	2,674	0.59	162,051
2013	6,075	3.88	1,196,387
2014	3,015	4.13	124,521
median	3,709	0.95	874,931

[a] Population estimates include hatchery returns based on RBDD ladder counts until 2001, after which the methodology changed to carcass surveys.

[b] Assumes all adults return after three years. CRR is calculated using the adult spawning population, divided by the spawning population three years prior. Two year old returns were not used.

[c] Includes survival estimates from spawning to Delta (*i.e.*, Sacramento at I St Bridge) entrance, but does not include through-Delta survival.

The cohort replacement rate (CRR), which is a measure of the population's growth rate, is shown in Figure 4.A.1-6 for brood years 1999 through 2014. The CRR was positive (i.e., greater than 1.00) for the last two years and indicates an increasing trend in the population following low abundance during 2007–2012. Although the CRR for the last two years was greater than the

CRR in all previous years dating back to 1999, juvenile winter-run Chinook salmon productivity is still much lower than other Chinook salmon runs in the Central Valley.

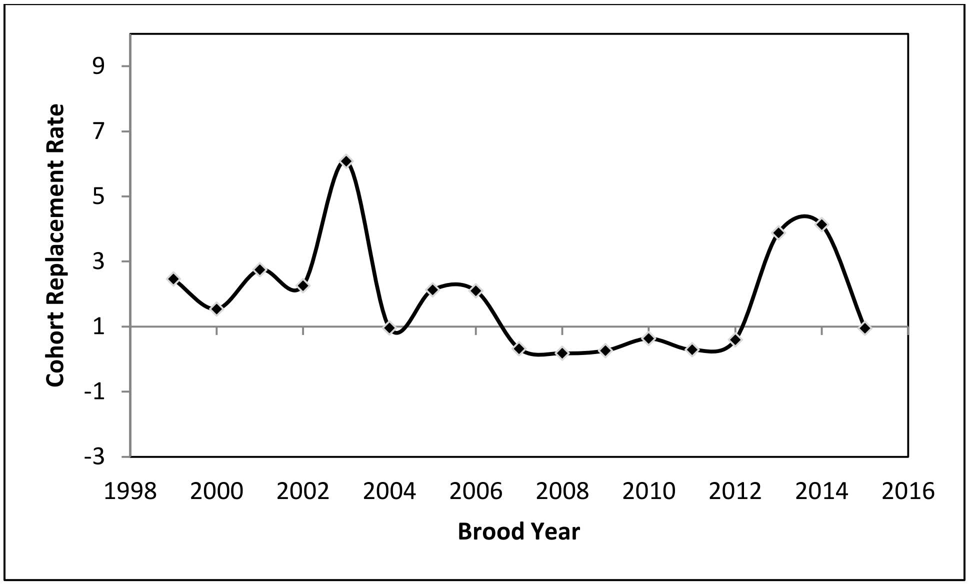

Figure 4.A.1-6. Winter-run Chinook Salmon Adult Cohort Replacement Rate, 1999–2014. (Source: Rea pers. comm.).

Spatial Structure

The distribution of winter-run Chinook salmon spawning and rearing was limited historically to the upper Sacramento River and tributaries, where cool spring-fed streams supported successful adult holding, spawning, egg incubation, and juvenile rearing (Slater 1963; Yoshiyama et al. 1998). The headwaters of the McCloud, Pit, and Little Sacramento Rivers and Hat and Battle Creeks, provided clean, loose gravel, cold, well-oxygenated water, and year-round flow in riffle habitats for spawning and incubation (Figure 4.A.1-1). These areas also provided the cold, productive waters necessary for egg and fry survival and juvenile rearing over summer.

Construction of Shasta Dam in 1943 and Keswick Dam in 1950 blocked access to all of these upstream waters except Battle Creek, which is blocked by a weir at the Coleman National Fish Hatchery and other small hydroelectric facilities (Moyle et al. 1989; NMFS 1997). Approximately 299 miles of tributary spawning habitat in the upper Sacramento River are inaccessible to winter-run Chinook salmon (NMFS 2012). Yoshiyama et al. (2001) estimated that in 1938, the Upper Sacramento River had a “potential spawning capacity” of 14,303 redds.

The greatest risk factor for winter-run Chinook salmon lies within its spatial structure (NMFS 2011). The remnant and remaining population cannot access 95% of their historical spawning habitat, and must therefore be artificially maintained in the Sacramento River outside of its historical range through spawning gravel augmentation, hatchery supplementation, and regulation of the finite cold-water pool behind Shasta Dam to reduce water temperatures. Winter-run Chinook salmon require cold-water temperatures in the 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. Presently, Battle Creek is the most feasible opportunity to expand the spatial structure of the ESU. Once completed, the Battle Creek Salmon and Steelhead Restoration

Project will reestablish approximately 48 miles of prime salmon and steelhead habitat in Battle Creek and its tributaries; however, the restoration project is not scheduled to be completed until 2020. The final Central Valley Salmon and Steelhead Recovery Plan (NMFS 2014) lists the McCloud River and Battle Creek as being a top priority for reintroduction, and the Little Sacramento River as a possible area for reintroduction, while the Pit River is classified as a non-candidate area for reintroduction (i.e., reintroduction should not be attempted here).

4.A.1.5.3 Diversity

Genetic: The genetic integrity of winter-run Chinook salmon has been compromised. Construction of Shasta Dam merged at least three independent winter-run Chinook salmon populations into a single population in the upper Sacramento River, resulting in the substantial loss of genetic diversity, life history variability, and local adaptation (NMFS 2014). Finally, multiple years of critically low adult escapement, especially in the early 1990s, have imposed further genetic “bottlenecks” that have reduced further the genetic diversity of the existing population (Good et al. 2005).

Although Livingston Stone National Fish Hatchery is operated to maximize genetic diversity and minimize domestication of the offspring produced in the hatchery, there is still concern that the hatchery may compromise the long-term viability and extinction risk of winter-run Chinook salmon through reduced genetic diversity and integrity because of the increasing number of hatchery-origin fish (sourced from the Livingston Stone National Fish Hatchery) spawning in habitat. Since 2000, the proportion of hatchery-origin fish spawning in the Sacramento River has generally ranged between 5–10% of the total population, except for in 2005 when it reached approximately 20% of the population. These rates are consistent with the goals of the hatchery program and the 10-year average introgression rate of hatchery fish (about 8%) is below the low-risk threshold (15%) for hatchery influence (National Marine Fisheries Service 2011).

Life-History Diversity: The habitat characteristics in areas where winter-run Chinook salmon adults historically spawned suggest unique adaptations by the population. Before the construction of Shasta Dam, winter-run Chinook salmon spawned in the headwaters of the McCloud, Pit, and Little Sacramento Rivers and Hat Creek as did spring-run Chinook salmon. Schofield (1900) reported that salmon arriving “earlier” than spring-run (presumably winter-run Chinook salmon) ascended Pit River Falls and entered the Fall River while the succeeding spring-run Chinook remained to spawn in the waters below. This indicates that winter-run Chinook salmon, unlike the other runs, ascended to the highest portions of the headwaters, and into streams fed mainly by the flow of constant-temperature springs arising from the lavas around Mount Shasta and Mount Lassen. These headwater areas probably provided winter-run Chinook salmon with the only available cool, stable temperatures for successful egg incubation over the summer (Slater 1963). Harvest pressure of the intensity experienced by the Sacramento River winter-run Chinook ESU can also alter diversity in age at-maturity; a critical factor for population viability (NMFS 2010). The ocean fishery is thought to select against fish that mature later because fish that would do so are vulnerable to harvest for more years (Ricker 1981; Hankin and Healey 1986; Franks and Lackey 2015), and age at maturity has moderate heritability (Hankin et al. 1993). As such, reduced ocean harvest would contribute substantially to age at-maturity diversity and thereby enhance population viability. Although important factors such as age at-maturity diversity that effect population dynamics and diversity are not explicitly incorporated into the Management Strategy Evaluation, further NMFS recognizes that it would be desirable to link specific influences across the entire life history into an ecosystem approach to manage impacts encompassing the complete life history (NMFS April 30, 2012 Memo).

As stated above, Shasta Dam merged all of the independent winter-run Chinook salmon populations that historically existed above Shasta Dam, resulting in the substantial loss of the life history variability and local adaptation that undoubtedly resulted from the diverse habitat characteristics found in these respective upstream areas.

4.A.1.5.4 ESU Viability

There is only one population of winter-run Chinook salmon and it depends heavily on coldwater releases from Shasta Dam (Good et al. 2005). Lindley et al. (2007) consider the winter-run Chinook salmon population at a moderate risk of extinction primarily because of the risks associated with only one existing population. The viability of an ESU that is represented by a single population is vulnerable to changes in the environment through a lack of spatial geographic and genetic diversity. A single catastrophic event with effects persisting for 4 or more years could extirpate the entire Sacramento River winter-run Chinook salmon ESU, which puts the population at a high risk of extinction over the long term (Lindley et al. 2007). Such potential catastrophes include volcanic eruption of Mount Lassen; prolonged drought, which depletes the coldwater pool in Lake Shasta or some related failure to manage coldwater storage; a spill of toxic materials with effects that persist for 4 years; regional declines in upwelling and productivity of near-shore coastal marine waters resulting in reduced food supplies for juvenile and sub-adult salmon, reduced growth, and/or increased mortality; or a disease outbreak. Another vulnerability to an ESU that is represented by a single population is the limitation in life history and genetic diversity that would otherwise increase the ability of individuals in the population to withstand environmental variation.

The most recent biological information suggests that the Sacramento River winter-run Chinook salmon ESU is at a high risk of extinction, and that several listing factors have contributed to the recent decline, including drought and poor ocean conditions (NMFS 2011). Long-term recovery of the Sacramento River winter-run Chinook salmon ESU will require improved freshwater habitat conditions and abatement of a wide range of threats throughout the entire ESU, and the establishment of populations in Battle Creek and possibly in the McCloud and/or Little Sacramento Rivers.

4.A.1.6 Relevant Conservation Efforts

Since the listing of winter-run Chinook salmon, several habitat and harvest-related problems that were identified as factors contributing to the decline of the species have been addressed and improved through restoration and conservation actions. The impetus for initiating restoration actions stems primarily from the following actions.

- ESA Section 7 consultation Reasonable and Prudent Alternative Actions that address water operations related management of water temperature, flow, and operations of the CVP/SWP (NMFS 2009b).

- Regional Water Quality Control Board decisions requiring compliance with Sacramento River water temperature objectives, which resulted in the installation of the Shasta Temperature Control Device in 1998.

- A 1992 amendment to the authority of the CVP through the Central Valley Improvement Act to give fish and wildlife equal priority with other CVP objectives.

- Fiscal support of habitat improvement projects from the CALFED Bay-Delta Program (CALFED) (e.g., installation of a fish screen on the Glenn-Colusa Irrigation District diversion, Battle Creek Restoration Project).

- EPA actions to control acid mine runoff from Iron Mountain Mine.

- Ocean harvest restrictions implemented in 1995.

Results of monitoring at the CVP/SWP fish salvage facilities and extensive experimentation over the past several decades have led to the identification of a number of management actions designed to reduce or avoid the potentially adverse effects of CVP/SWP export operations on salmon. Many of these actions have been implemented through State Water Board water rights decisions (D-1485, D-1641), BiOps issued on project export operations by NMFS and U.S. Fish and Wildlife Service (USFWS), CALFED programs (e.g., Environmental Water Account), and Central Valley Project Improvement Act actions. These requirements support multiple conservation efforts to enhance habitat and reduce entrainment of Chinook salmon by the CVP/SWP export facilities.

The artificial propagation program for winter-run Chinook salmon at Livingston Stone National Fish Hatchery, located on the mainstem of the Sacramento River, has operated for conservation hatchery fish, and only wild (not fin-clipped) fish are currently being spawned in the hatchery to reduce genetic introgression of the population (NMFS 2011).

BiOp s for CVP/SWP operations (NMFS 2009b) and other federal projects involving irrigation and water diversion and fish passage have improved or minimized adverse impacts on salmon in the Central Valley. In 1992, an amendment to the authority of the CVP through the Central Valley Project Improvement Act gave protection of fish and wildlife equal priority with other CVP objectives. From this act arose several programs that have benefited listed salmonids. The Anadromous Fish Restoration Program is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the Anadromous Fish Restoration Program include fish passage, fish screening, riparian easement, and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The Anadromous Fish Screen Program combines federal funding with state and private funds to prioritize and construct fish screens on major water diversions mainly in the upper Sacramento River. Despite these and other conservation efforts, the program has fallen short of the goal of doubling the natural production of winter-run Chinook salmon (NMFS 2011).

The goal of the Water Acquisition Program, Central Valley Project Improvement Act Section 3406 (b)(3), is to acquire water supplies to meet the habitat restoration and enhancement goals of the Central Valley Project Improvement Act, and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for Central Valley salmon, with the primary focus on listed Chinook salmon and steelhead, including winter-run Chinook salmon, by maintaining or increasing instream flows (e.g., improved water management and conservation (Section 3406 (b)(2)) on the Sacramento River at critical times, and to reduce salmonid entrainment at the CVP/SWP export facilities through reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses. However, impacts from factors such as drought, climate change, and poor survival conditions have increased in recent years and are likely to be substantial contributing factors to the declining abundance of the ESU (NMFS 2011).

A major restoration action currently under way is the Battle Creek Salmon and Steelhead Restoration Project, which is modifying facilities at Battle Creek Hydroelectric Project diversion dam sites located on the North and South Forks of Battle Creek and Baldwin Creek. Although winter-run Chinook salmon do not currently inhabit Battle Creek, they occurred there historically. CALFED is funding the establishment of a second independent population of winter-run Chinook salmon in the upper Battle Creek watershed using the artificial propagation program as a source of fish. The project is restoring 77 kilometers (48 miles) of habitat in Battle Creek and its tributaries to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $90 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. This restoration effort is thought to be the largest coldwater restoration project to date in North America. Other than the potential benefits of the Battle Creek restoration effort, there has been very limited habitat expansion, but no substantial changes in habitat condition or availability since the ESU was listed (NMFS 2011).

As part of CALFED and Central Valley Project Improvement Act programs, many of the largest water diversions located on the Sacramento River and Delta (e.g., Glenn Colusa Irrigation District, Reclamation District (RD) 1001 Princeton diversion, RD 108 Wilkins Slough Pumping Plant, Sutter Mutual Water Company Tisdale Pumping Plant, Contra Costa Water District’s Old River and Alternative Intake Project intake, and others) have been equipped with positive barrier fish screens, although the majority of smaller water diversions located on the Sacramento River and Delta remain unscreened. Reclamation District 108 has also designed and constructed a new fish screen and pumping plant (Poundstone Pumping Plant) located on the Sacramento River that consolidates and eliminates three existing unscreened water diversions. These fish-screening projects are specifically intended to reduce and avoid entrainment losses of juvenile winter-run Chinook salmon and other fish inhabiting the river.

The DRERIP was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements in the Delta (Williams 2010). The DRERIP team has created a suite of ecosystem and species conceptual models, including winter-run Chinook salmon, that document existing scientific knowledge of Delta ecosystems. The DRERIP team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation. DRERIP conceptual models were used in the analysis of proposed conservation measures.

The Central Valley Salmonid Project Work Team, an interagency technical working group led by CDFW, drafted a proposal to develop a Chinook salmon escapement monitoring plan that was selected by the CALFED Ecosystem Restoration Program Implementing Agency Managers for directed action funding.

Recent habitat restoration initiatives sponsored primarily by the CALFED Ecosystem Restoration Program have funded 29 projects (approximately $24 million) designed to restore ecological function to 9,543 acres (8,091 acres in the Bay Area and the remaining acres located in the Delta and Eastside Tributaries Regions of the CALFED action area) of shallow-water tidal and marsh habitats in the Delta. Over the last 11 years, the CALFED Ecosystem Restoration Program has provided funding for about 580 projects, totaling over $700 million, and is currently managing 74 previously funded projects and 18 newly funded projects totaling about $24 million (DFG et al. 2011). The majority of the funding has been spent on projects focusing on riparian habitat restoration, fish screen installations, water and sediment quality improvements, and stream hydrodynamic enhancements.

EPA's Iron Mountain Mine remediation involves removing toxic metals in acidic mine drainage from the Spring Creek Watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine, and other mining operations, has shown measurable reductions since the early 1990s. 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 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.

In 2001, a new fish screen was constructed and fish ladder was installed at the Anderson Cottonwood Irrigation District Diversion Dam to address the threats caused by the dam. As described in the final rule in which NMFS determined that the ESU should remain listed as endangered (70 FR 37160; June 28, 2005), the flashboard gates and inadequate fish ladders at the diversion dam blocked passage for winter-run Chinook salmon migrating upstream. Seasonal operation of the dam created unsuitable habitat upstream of the dam by reducing flow velocity over the incubating eggs, reducing egg survival. Since the new fish ladders was installed (2001) approximately half of the returning annual spawning winter-run Chinook salmon have been spawning upstream of the dam, predominantly on gravel augmented under the CVPIA program.

To eliminate an impediment to migration of adult and juvenile winter-run Chinook salmon and other species, operation of the Red Bluff Diversion Dam ceased in 2011 and dam gates were placed in a permanent open position. A new pumping facility was built that includes a state-of-the-art fish screen.

Since 1986, DWR's Delta Fish Agreement Program has approved approximately $49 million for projects that benefit salmon and steelhead production in the Sacramento and San Joaquin River basins and Delta. Delta Fish Agreement projects that benefit winter-run Chinook salmon include enhanced law enforcement efforts from San Francisco Estuary upstream into the Sacramento River, spawning gravel augmentations, and habitat enhancement projects. Through the Delta-Bay Enhanced Enforcement Program initiated in 1994, a team of 10 wardens focuses their enforcement efforts on salmon, steelhead, and other species of concern from the San Francisco Estuary upstream into the Sacramento and San Joaquin River basins. Enhanced enforcement programs attributed to CDFW are believed to have had significant benefits on Chinook salmon, although results have not been quantified.

Protective measures for winter-run Chinook salmon include seasonal constraints on sport and commercial fisheries south of Point Arena in an effort to reduce harvest of winter-run Chinook salmon. Ocean harvest restrictions since 1995 have led to reduced ocean harvest of winter-run Chinook salmon (i.e., Central Valley Chinook salmon ocean harvest index ranged from 0.55 to nearly 0.80 from 1970 to 1995, and was reduced to 0.27 in 2001). The average 2000 to 2007 harvest index was reduced to 0.17, and the closure of the primary ocean fishery on this stock in 2008 and 2009 is expected to reduce the harvest index to approximately zero during these two years (NMFS 2011). In an effort to identify reasonable and prudent alternatives (RPA's) NMFS set out to identify a threshold or set of thresholds, based on the status of winter-run Chinook salmon, which would trigger additional measures to reduce the impacts of the ocean salmon fishery on the species. The new fisheries management framework for managing winter-run Chinook salmon impacts in the ocean salmon fishery consists of two components. The first specifies that the previous consultation standards for winter-run Chinook salmon regarding minimum size limits and seasonal windows to south of Point Arena for both the commercial and recreational fisheries will continue to remain in effect at all times regardless of abundance estimates or impact rate limit. The second component is an abundance-based framework where, during periods of relatively low abundance, preseason fishery impact rate projections south of Point Arena for winter-run Chinook salmon based on the proposed structure of fishing management measures each year must be equal to or less than the maximum allowable impact rate (impact rate cap) specified annually, based on the population status of winter-run Chinook salmon. These impact rate caps will be determined annually based on the geometric mean of the most recent 3 years of spawning return estimates for winter-run Chinook salmon generated by carcass surveys conducted on the Sacramento River, including the fish collected at the Keswick trap (NMFS April 30, 2012 Memo).

The state of California has also established specific in-river fishing regulations and no-retention prohibitions designed to protect winter-run Chinook salmon. CDFW has implemented enhanced enforcement efforts to reduce illegal harvests.

4.A.1.7 Recovery Goals

The recovery plan for Central Valley salmonids, including Sacramento River winter-run Chinook salmon, was released by NMFS on July 22, 2014 (NMFS 2014). The overarching goal is the removal of, among other listed salmonids, Sacramento River winter-run Chinook salmon from the federal list of Endangered and Threatened Wildlife (NMFS 2014). Recovery goals usually can be subdivided into discrete component objectives that, collectively, describe the conditions (criteria) necessary for achieving the goal. Recovery objectives are the parameters of the goal, and criteria are the values for those parameters. For the ESU to achieve recovery, each of the Diversity Groups should support both viable and dependent populations and meet goals for redundancy and distribution. According to NMFS (2014), the Sacramento River winter-run Chinook salmon ESU should display the following characteristics to achieve recovery:

- Three populations in the Basalt and Porous Lava Diversity Group at low risk of extinction

Criteria for low risk of extinction include; a census population size that is >2,500 adults, or has an effective population size that is >500, has no productivity decline that is apparent, has had no catastrophic event that has occurred within the last 10 years, and hatchery influence is at low levels.

4.A.1.8 References

4.A.1.8.1 Written References

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4.A.2 Chinook Salmon, Central Valley Spring-Run (*Oncorhynchus tshawytscha*)

This section provides information on the basic biology, life history, status, and stressors of Central Valley (CV) spring-run Chinook salmon in the action area.

4.A.2.1 Status

The Central Valley (CV) spring-run Chinook salmon ESU is listed as a threatened species under ESA. The ESU includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California, and the Feather River Hatchery spring-run Chinook program (Figure 4.A.2-1). Key sub-populations in Mill, Deer, and Butte creeks are outside of the action area; however, all migratory life stages must pass through the action area. The ESU was originally listed as threatened on September 16, 1999 (64 *Federal Register* [FR] 50394) for the following reasons:

- The species occurred in only a small portion of its historical range.
- From 70 to 90% of spawning and rearing habitats had been lost.
- Abundance declined to low levels (5-year average of 8,500 fish, compared with 40,000 fish in 1940s).
- There is a potential for hybridization between spring- and fall-run fish in hatcheries and in the wild, including the mainstem Sacramento River and tributaries that support Central Valley spring-run Chinook.

In June 2004, the National Marine Fisheries Service (NMFS) proposed that CV spring-run Chinook salmon remain listed as threatened (69 FR 33102; June 14, 2004). This proposal was based on the recognition that, although CV spring-run Chinook (CVSC) salmon productivity trends were positive, the ESU continued to face risks from having a limited number of remaining populations (i.e., three existing populations from an estimated 17 historical populations), a limited geographic distribution, and potential for hybridization with fall-run Chinook salmon.

On June 28, 2005, NMFS issued its final decision to retain the status of CV spring-run Chinook salmon as threatened (70 FR 37160). This decision also included the Feather River Hatchery spring-run Chinook salmon population as part of the CV spring-run Chinook salmon ESU. Until this final decision, Feather River Hatchery spring-run Chinook salmon had not been included in the ESU, yet these fish are genetically distinct from other populations in Mill, Deer, and Butte Creeks.

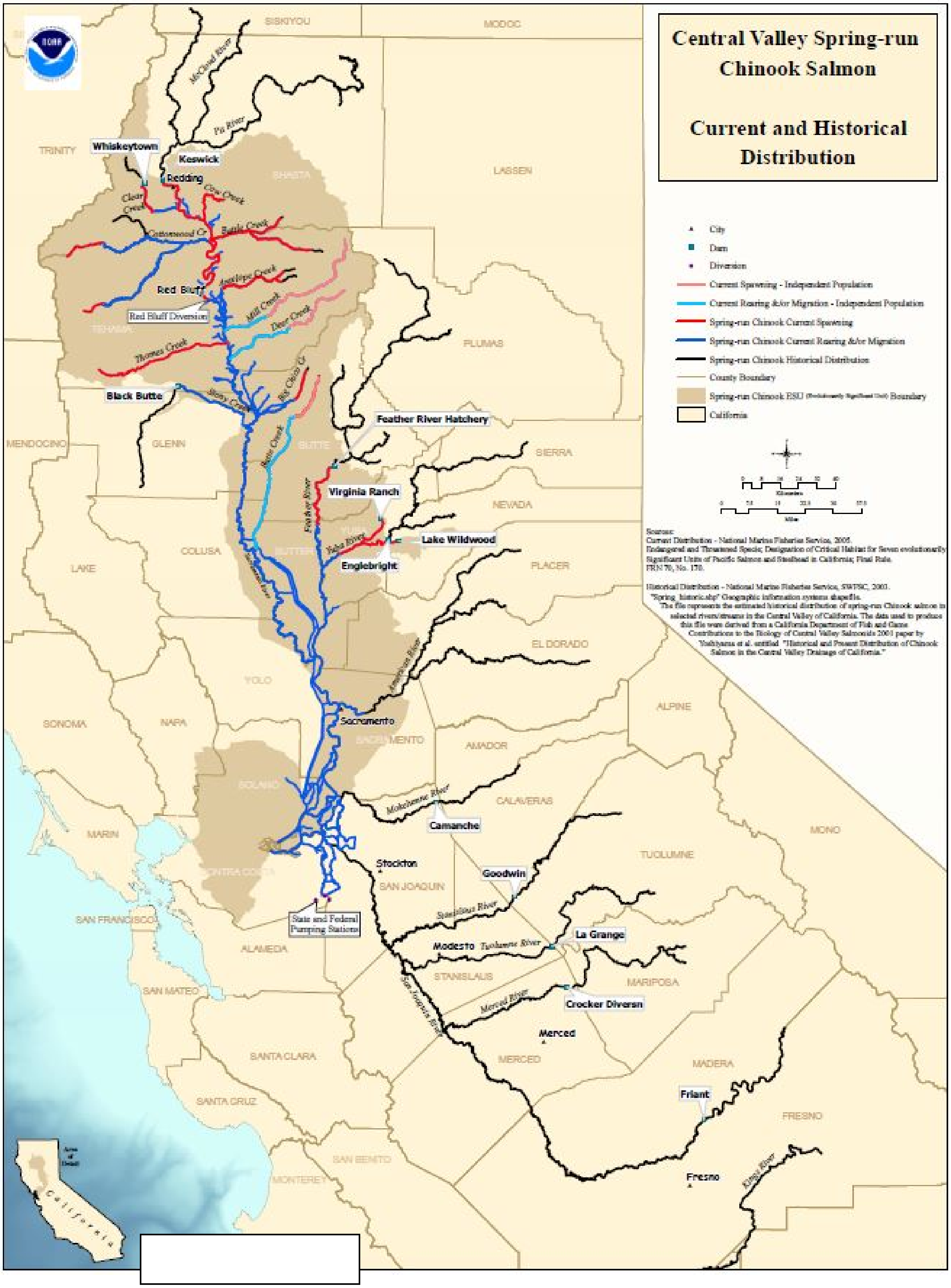

Figure 4.A.2-1. Current and Historical Distribution of Central Valley Spring-run Chinook Salmon

In its latest 5-year status review, NMFS determined that the ESU should remain classified as a threatened species. However, NMFS determined that the biological status had worsened since the last status review, and the ESU had an increased extinction risk (NMFS 2011). With a few exceptions, escapements had declined over the previous 10 years, particularly since 2006, placing the Mill and Deer Creek populations at high risk of extinction because of their rate of decline (NMFS 2011). While the Butte Creek population abundance continues to meet the low extinction risk criteria, the rate of decline is close to triggering the population decline criterion for high risk. Overall, the recent declines have been significant but not severe enough to qualify as a catastrophe under the criteria of Lindley et al. (2007). In addition, spring-run Chinook salmon appear to be repopulating Battle Creek, home to a historical independent population (NMFS 2011).

The San Joaquin River spring-run Chinook salmon population has been designated an experimental population under ESA Section 10(j) only when, and at such times as, they are found in the San Joaquin River from Friant Dam downstream to its confluence with the Merced River, which is outside of the action area (78 FR 79622; December 31, 2013). However, individuals of this population are given the same consideration as the listed ESU when they pass through the action area.

CESA: Spring-run Chinook salmon was listed as a threatened species under the California Endangered Species Act (CESA) on February 5, 1999.

4.A.2.2 Critical Habitat

Critical habitat for CV spring-run Chinook salmon ESU was updated on September 2, 2005 (70 FR 52488). Designated critical habitat includes 1,158 miles of stream habitat in the Sacramento River basin and 254 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex (70 FR 52488, Figure 4.A.2-2). Critical habitat includes stream reaches such as those of the Feather and Yuba Rivers, Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear Creeks, and the Sacramento River and Delta.

The critical habitat designation identified the following physical or biological features (PBFs) considered essential for the conservation of the ESU.

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

3. Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.
4. Estuarine areas free of obstruction and excessive predation with:
    i. Water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater;
    ii. Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and
    iii. Juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

CV spring-run Chinook salmon habitats are also protected under the Magnuson-Stevens Fishery Conservation and Management Act as essential fish habitat (EFH) identified in the Fisheries Management Plan (FMP) for Pacific Salmon. The FMP identified five Habitat Areas of Particular Concern (HAPC), a number of which are present in the action area. Those waters and substrate that are necessary to Chinook salmon for spawning, breeding, feeding, or growth to maturity are included as EFH (Figure 4.A.1-2). Critical habitat (Figure 4.A.2-2) and EFH are managed differently from a regulatory standpoint, but are biologically equal for the conservation of CV spring-run Chinook salmon.

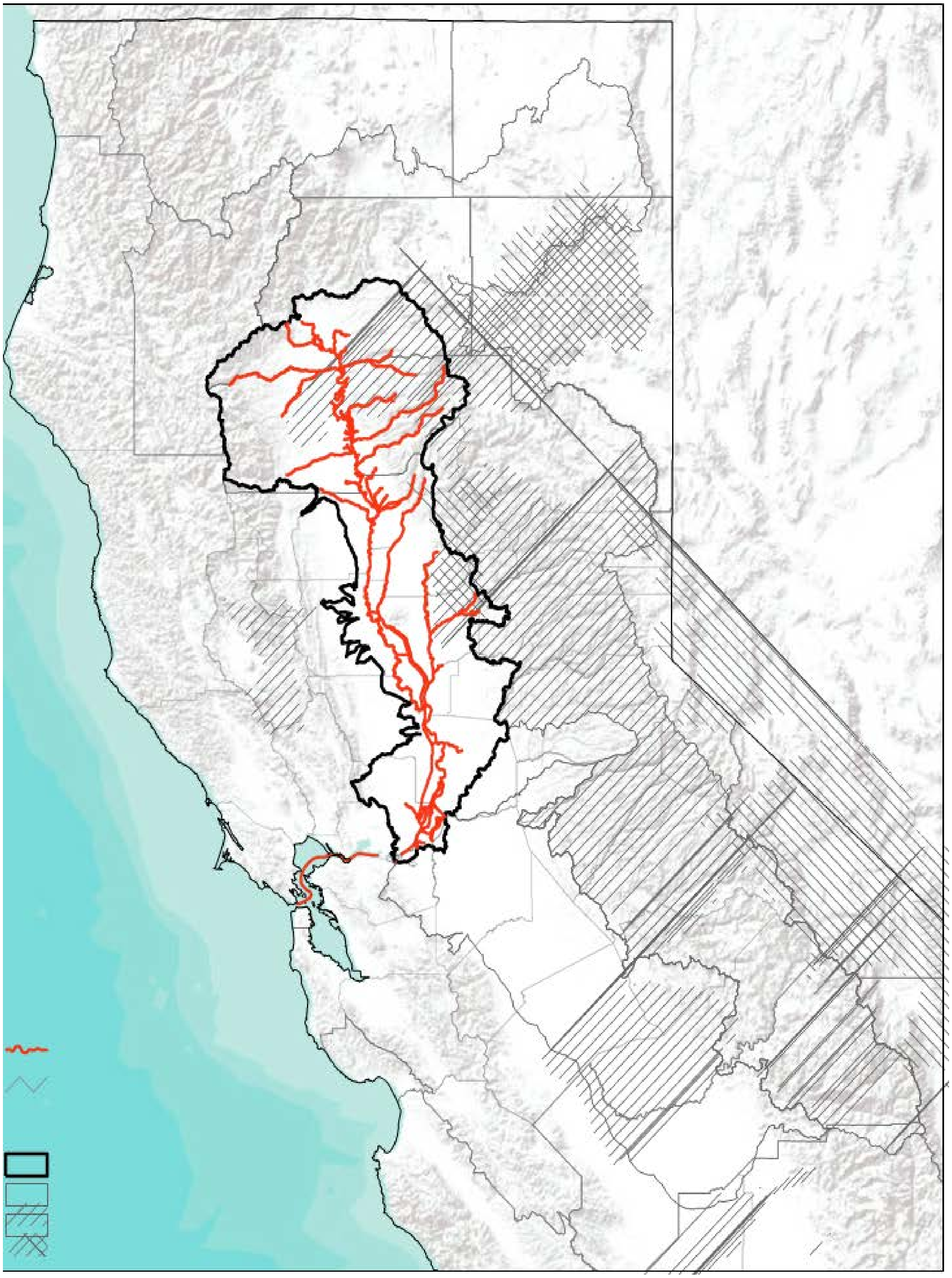

Figure 4.A.2-2. Central Valley Spring-Run Chinook Salmon Inland Designated Critical Habitat (Source: NMFS 2015)

4.A.2.2.1 Freshwater Spawning Habitat

According to NMFS (70 FR 52536, Sept. 2, 2005), freshwater spawning sites should provide water quantity (instream flows) and quality conditions (e.g., water temperature and dissolved oxygen) and substrate supporting spawning, egg incubation, and larval development. Spawning habitat has a high conservation value as its function directly affects the spawning success and reproductive potential of listed salmonids. Most spawning habitat in the Central Valley for spring-run Chinook salmon is located in areas directly downstream of dams containing suitable environmental conditions for spawning and incubation. Historically, spring-run Chinook salmon migrated upstream into high-elevation steep gradient reaches of the rivers and tributaries for spawning. Access to the majority of these historical spawning areas has been blocked by construction of major Central Valley dams and reservoirs. Currently, CV spring-run Chinook salmon spawn on the mainstem Sacramento River between the Red Bluff Diversion Dam and Keswick Dam, and in tributaries such as the Feather River, Mill, Deer, Clear, Battle and Butte Creeks. In the future, spawning is expected to occur in the San Joaquin River downstream of Friant Dam once the 10(j) experimental San Joaquin population becomes reestablished and self-sustaining.

4.A.2.2.2 Freshwater Rearing Habitat

According to NMFS (70 FR 52536, Sept. 2, 2005), freshwater rearing sites should have sufficient water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; suitable water quality; and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Freshwater rearing habitat has a high conservation value, as the juvenile life stage of salmonids is dependent on the function of this habitat for successful survival and recruitment to the adult population. Both spawning areas and migratory corridors comprise rearing habitat for juveniles, which feed and grow before and during their out-migration.

Juveniles also rear in nonnatal, intermittent tributaries. Rearing habitat condition is strongly affected by habitat diversity and complexity, food supply, and presence of predators. Channeled, leveed, and riprapped river reaches and sloughs that are common along the Sacramento and San Joaquin Rivers and throughout the Delta typically have low habitat complexity, low abundance of food organisms, and offer little protection from predatory fish and birds. However, some of these more complex, productive habitats with floodplain connectivity are still present in limited amounts in the Central Valley (e.g., Sacramento River reaches with setback levees [primarily located upstream of the City of Colusa]).

4.A.2.2.3 Freshwater Migration Corridors

Freshwater migration corridors for spring-run Chinook salmon, including river channels, channels through the Delta, and the Bay-Delta estuary support mobility, survival, and food supplies for juveniles and adults. According to NMFS (70 FR 52536, Sept. 2, 2005), migration corridors should be free from obstruction (passage barriers and impediments to migration) and predation, with water quantity (instream flows) and quality conditions (seasonal water temperatures) and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. Migratory corridors for spring-run Chinook salmon are located downstream of the spawning areas and include the lower Sacramento River, lower Feather River, tributaries providing suitable adult holding and spawning habitat, the Delta, and the San Francisco Bay complex extending to coastal marine waters. Efforts are currently under way to reestablish a spring-run Chinook salmon population on the San Joaquin River downstream of Friant Dam that would use the lower river and Delta as part of the migration corridor. These corridors allow the upstream passage of adults and the downstream emigration of juvenile salmon. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, reduce false attraction, avoid areas where vulnerability to predation is increased, and avoid impediments and delays in both upstream and downstream migration. For this reason, freshwater migration corridors are considered to have a high conservation value.

Results of mark-recapture studies conducted using juvenile Chinook salmon (typically fall-run or late fall-run Chinook salmon, which are considered to be representative of juvenile spring-run Chinook salmon) released into both the Sacramento and San Joaquin Rivers have shown high mortality during passage downstream through the rivers and Delta (Newman and Rice 2002; Newman 2008; Perry et al. 2010; Michel (2010); San Joaquin River Group Authority 2013). Mortality for juvenile salmon is typically greater in the San Joaquin River than in the Sacramento River (Brandes and McLain 2001). Results of survival studies have shown that closing the Delta Cross Channel gates to reduce the movement of juvenile salmon into the interior Delta contribute to improved survival of emigrating juvenile Chinook salmon (Newman and Brandes 2010; Perry et al. 2010). Although the factors contributing to high juvenile mortality have not been quantified, results of acoustic tagging experiments and anecdotal observations suggest predation, mediated by poor habitat or water quality conditions, is likely a primary factor (Grossman et al. 2013) There are several factors affecting direct loss of Sacramento River salmon at salvage facilities including pumping rates, Sacramento River flow, run timing, species abundance, water year type, DCC gate operations and predator abundance (Larry Walker Associates 2010, Buchanon 2013, Cloern 2012, Harvey 2011, Perry et al. 2010, Perry & Skalski 2008, Perry 2012, Zeug and Cavallo 2013, Perry et al 2015).

4.A.2.2.4 Estuarine Habitats

According to NMFS (70 FR 52536, Sept. 2, 2005), estuarine migration and juvenile rearing habitats should be free of obstruction (i.e., dams and other barriers) and excessive predation with suitable water quality, water quantity (river and tidal flows), and salinity conditions supporting juvenile and adult physiological transitions between fresh and salt water; natural cover, such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturity. Tidal wetlands and seasonally inundated floodplains are identified as high-value foraging and rearing habitats for juvenile salmon migrating downstream through the estuary. Estuarine areas have a high conservation value as they support juvenile Chinook salmon growth, smolting, avoidance of predators, and the transition to the ocean environment.

4.A.2.2.5 Marine Habitats

Although ocean habitats are not part of the designated critical habitat for CV spring-run Chinook salmon, biologically productive coastal waters are an important habitat component for the ESU. Juvenile Chinook salmon inhabit near-shore coastal marine waters for a period of typically 2 to 4 years before adults return to Central Valley Rivers to spawn. During their marine residence, Chinook salmon forage on krill, squid, and other marine invertebrates as well as a variety of fish such as northern anchovy and Pacific herring.

Results of oceanographic studies have shown the variation in ocean productivity off the West Coast within and among years. Changes in ocean currents and upwelling are significant factors affecting nutrient availability, phytoplankton and zooplankton production, and the availability of other forage species in nearshore surface waters. Ocean conditions during the salmon's ocean residency period can be important, as indicated by the effect of the 1983 El Niño on the size and fecundity of Central Valley fall-run Chinook salmon (Wells et al. 2006). Although the effects of ocean conditions on Chinook salmon growth and survival have not been investigated extensively, observations since 2006 to 2009 showed a significant decline in the abundance of adult Chinook salmon and coho salmon returning to California rivers and streams that was attributable primarily to ocean conditions (Lindley et al. 2009). These declines are believed to have been exacerbated by long-term degradation of freshwater, estuarine, and marine habitats, coupled with hatchery practices that limit life history diversity. The importance of changes in ocean conditions on growth, survival, and population abundance of Central Valley Chinook salmon is currently undergoing further investigation (Kilduff et al. 2014; Sharma et al. 2013).

4.A.2.3 Life History

Chinook salmon exhibit two generalized freshwater life history types (Healey 1991). Stream-type adults enter fresh water months before spawning and juveniles reside in fresh water for a year or more following emergence, whereas ocean-type adults spawn soon after entering fresh water and juveniles migrate to the ocean as fry or parr in their first year. Adequate instream flows and cool water temperatures are more critical for the survival of Chinook salmon exhibiting a stream-type life history due to over-summering by adults and/or juveniles. Spring-run Chinook salmon are somewhat anomalous in that they have characteristics of both stream- and ocean-type races (Healey 1991). Adults enter fresh water in early-late spring, and delay spawning until late summer or early fall (stream-type). However, most juvenile spring-run Chinook salmon migrate out of their natal stream after only a few months of river life (ocean-type), or they may remain for up to 15 months within their natal stream. This life-history pattern differentiates the spring-run Chinook from other Sacramento River Chinook runs and from all other populations within the range of Chinook salmon (Hallock and Fisher 1985)

In addition to their unique life-history patterns, the behavior of spring-run Chinook adults as they return to spawn differentiates the population. Adults enter freshwater in an immature reproductive state, similar to winter-run Chinook, but spring-run Chinook move into higher elevations and then hold in the cool water pools for an extended period before spawning (Moyle 2002).

4.A.2.3.1 Immigration and Holding

Freshwater entry and spawning timing generally are thought to be related to local water temperature and flow regimes. Runs of Chinook salmon are designated based on 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).

Adult CV spring-run Chinook salmon begin their upstream migration in late January and early February (DFG 1998) and enter the Sacramento River between February and September, primarily in May and June (Table 4.A.2-1) (Yoshiyama et al. 1998; Moyle 2002). Lindley et al. (2006) reported that adult CV spring-run Chinook salmon enter native tributaries from the Sacramento River primarily between mid-April and mid-June. Typically, spring-run Chinook salmon use mid- to high-elevation streams that provide appropriate seasonal water temperatures and sufficient flow, cover, and pool depth to allow over-summering while conserving energy and allowing their gonadal tissue to mature (Yoshiyama et al. 1998).

Spring-run Chinook salmon tend to enter fresh water as immature fish, migrate far upriver, hold in cool-water pools for a period of months during the spring and summer, and delay spawning for several months until the early fall. Pools in the holding areas need to be sufficiently deep, cool (about 64° F or less), and oxygenated to allow over-summer survival. Adults tend to hold in pools near quality spawning gravel. DFG (1998) characterized these holding pools as having moderate water velocities (0.5 to 1.3 feet per second) and cover, such as bubble curtains.

4.A.2.3.2 Spawning

Chinook salmon typically mature between 2 and 6 years of age, although more commonly from 2 to 4 years (Myers et al. 1998). Chinook salmon spawn in clean, loose gravel in swift, relatively shallow riffles or along the margins of deeper reaches where suitable water temperature, depth, and velocity favor redd construction and adequate oxygenation of incubating eggs. Chinook salmon spawning typically occurs in gravel beds located at the tails of holding pools (USFWS 1995). Adult Chinook have been observed spawning in water greater than 0.8 foot deep and in water velocities of 1.2 to 3.5 feet per second (Puckett and Hinton 1974, as cited in DFG 1998). Montgomery et al. (1999) reported adult Chinook tend to spawn in stream reaches characterized as low-gradient pool-riffle or forced pool-riffle reaches. Like steelhead, Chinook dig a redd and deposit their eggs within the stream sediment where incubation, hatching, and subsequent emergence take place. Optimum substrate for embryos is a gravel/cobble mixture with a mean diameter of 1 to 4 inches and a composition including less than 5% fines (particles less than 0.3 inch in diameter) (Platts et al. 1979; Reiser and Bjornn 1979 both as cited in DFG 1998).

Currently, adult Chinook that CDFW consider spring-run, spawn from mid to late August through early October, with peak spawning times varying among locations. For instance, in Deer Creek, spawning begins first at higher elevations, which are the coolest reaches. Spawning occurs progressively later in the season at lower elevations as temperatures cool (DFG 1998). Water temperatures between 42 °F and 58 °F are considered most suitable for spawning.

Fisher (1994) reported that 2% of female CV spring-run Chinook salmon adults are age 2, 87% are age 3, and 11% are 4-year olds based on observations of adult Chinook salmon trapped and examined at RBDD between 1985 and 1991.

DFG (1998) developed a regression model to predict Sacramento River Chinook fecundity from fork length. Using this model, they estimated CV spring-run Chinook salmon fecundity ranged from 1,350 to 7,193 eggs per female, with a weighted average of 4,161 eggs per female.

4.A.2.3.3 Egg to Parr

Egg survival rates are dependent, in part, on water temperature. At an incubation temperature of 56°F, eggs would be in the gravel approximately 70 days. Chinook eggs and alevins are in the gravel (spawning to emergence) for 900 to 1,000 accumulated temperature units. One accumulated temperature unit is equal to a temperature of 1°C for 1 day. Expressed in degrees Fahrenheit, the range is 1,652 to 1,832 accumulated temperature units.

Fry emergence from the gravel generally occurs at night from November to April (Moyle 2002; Harvey 1995; Bilski and Kindopp 2009). Upon emergence, fry swim or are displaced downstream (Healey 1991). The daily migration of juvenile spring-run Chinook salmon passing Red Bluff Diversion Dam is highest in the 4-hour period prior to sunrise (Martin et al. 2001).

Once fry emerge from the gravel, they initially seek areas of shallow water and low velocities while they finish absorbing the yolk sac (Moyle 2002). Many also disperse downstream during high-flow events. Fry may continue downstream to the estuary and rear, or may take up residence in the stream for a period from weeks to a year (Healey 1991). Fry seek streamside habitats containing beneficial characteristics such as riparian vegetation and associated substrates that provide aquatic and terrestrial invertebrates, predator avoidance cover, and slower water velocities for resting (NMFS 1996).

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 (Healey 1991). As is the case with other salmonids, there is a shift in microhabitat use by juveniles to deeper, faster water as they grow. Microhabitat use can be influenced by the presence of predators, which can force juvenile salmon to select areas of heavy cover and suppress foraging in open areas (Moyle 2002). Catches of juvenile salmon in the Sacramento River near West Sacramento by the U.S. Fish and Wildlife Service (USFWS) (1997) showed that larger juvenile salmon were captured in the main channel and smaller fry were typically captured along the channel margins. When the channel of the river is greater than 9 to 10 feet in depth, juvenile salmon tend to inhabit surface waters (Healey 1980). Stream flow changes and/or turbidity increases in the upper Sacramento River watershed are thought to stimulate juvenile emigration (Kjelson et al. 1982; Brandes and McLain 2001).

Juvenile CV spring-run Chinook salmon rear in natal tributaries, the Sacramento River mainstem, nonnatal tributaries to the Sacramento River, and the Delta (DFG 1998). Juvenile CV spring-run Chinook salmon from Mill and Deer creeks are thought to emigrate as yearlings in greater proportions than spring-run Chinook salmon from other tributaries (DFG 1998).

Within the Delta, juvenile Chinook salmon forage in shallow areas with protective cover, such as tidally influenced sandy beaches and shallow water areas with emergent aquatic vegetation (Meyer 1979; Healey 1980). Cladocerans, copepods, amphipods, and larval dipterans, as well as small arachnids and ants are common prey items (Kjelson et al. 1982; Sommer et al. 2001a; MacFarlane and Norton 2002). Although the bulk of production in Butte and Big Chico Creeks emigrate as fry, yearlings can enter the Delta as early as February and as late as June (DFG 1998). Yearling-sized spring-run Chinook salmon migrants appear at Chipps Island (entrance to Suisun Bay) between October and December (Brandes and McLain 2001; USFWS 2001).

While there have been few studies of estuarine habitat use by juvenile spring-run Chinook, the low numbers of juveniles encountered throughout the bays and lower tidal marshes, and the lack of growth observed in those reaches reflect the immense changes and habitat alteration that have taken place in those areas over the last century (MacFarlane and Norton 2002). Over this period, the bulk of the tidal marsh and creek habitats had been leveed, channelized, and dredged, for navigation and other anthropogenic purposes. In addition, water diversions at Delta pump facilities have altered hydrology, salinity, and turbidity in the lower Delta. These changes in habitat conditions in the Delta over the past century may have resulted in a reduction in extended juvenile salmon rearing when compared to periods when habitat for juvenile salmon rearing was more suitable.

4.A.2.3.4 Smolt and Pre-smolt Downstream Migration

Spring-run Chinook salmon emigration timing is highly variable, as they may migrate downstream as young-of-the-year or as juveniles or yearlings. The modal size of fry migrants at approximately 40 millimeters between December and April in Mill, Butte, and Deer Creeks reflects a prolonged emergence of fry from the gravel (Lindley et al. 2004). Studies in Butte Creek found that the majority of CV spring-run Chinook salmon migrants are fry occurring primarily during December, January, and February, and that fry movements appeared to be influenced by flow (Ward et al. 2002, 2003; McReynolds et al. 2005). Small numbers of CV spring-run Chinook salmon remained in Butte Creek to rear and migrated as yearlings later in the spring. Juvenile emigration patterns in Mill and Deer Creeks are very similar to patterns observed in Butte Creek, with the exception that juveniles from Mill and Deer creeks typically exhibit a later young-of-the-year migration and an earlier yearling migration (Lindley et al. 2006; Figure 4.A.2-3).

Peak movement of yearling CV spring-run Chinook salmon in the Sacramento River at Knights Landing occurs in December, and is high in January, tapering off through the middle of February; however, juveniles were also observed between November and the end of February (Snider and Titus 2000).

Table 4.A.2-1. Temporal Occurrence of Adult and Juvenile Central Valley Spring-Run Chinook Salmon in the Sacramento River.

Spring-run relative abundance	High				Medium				Low			
(a) Adults												
Location Sacramento River	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
[a,b] Immigration, RBDD												
[c] Holding, Keswick, RBDD												
[b,d] Spawning, eggs, alevins												
(b) Juveniles												
[e,f] Juvenile rearing, Keswick, RBDD												
[g,e,f] Emigration, RBDD												
(a) Adults												
Location Feather River	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
[g] Immigration, Sac. confluence												
[b] Holding, HF channel												
[h] Spawning,eggs, alevins, LF channel												
(b) Juveniles												
[h] Juvenile rearing, LF and HF channel												
[i]Juvenile emigration, Sac. confluence												

Sources: [a]Yoshiyama et al. (1998); [b]Moyle (2002); [c]inferred based on immigration and spawning timing; [d]CDFW aerial redd surveys; [e]Snider and Titus 2000; [f]Poytress et al 2014; [g]Seesholtz et al. (2004); [h]Bilski and Kindopp 2009; [i]inferred based on juv rearing timing

Abbreviations: RBDD = Red Bluff Diversion Dam, Keswick = Keswick Dam, Sac.= Sacramento River, HF = high-flow channel, LF = low-flow channel

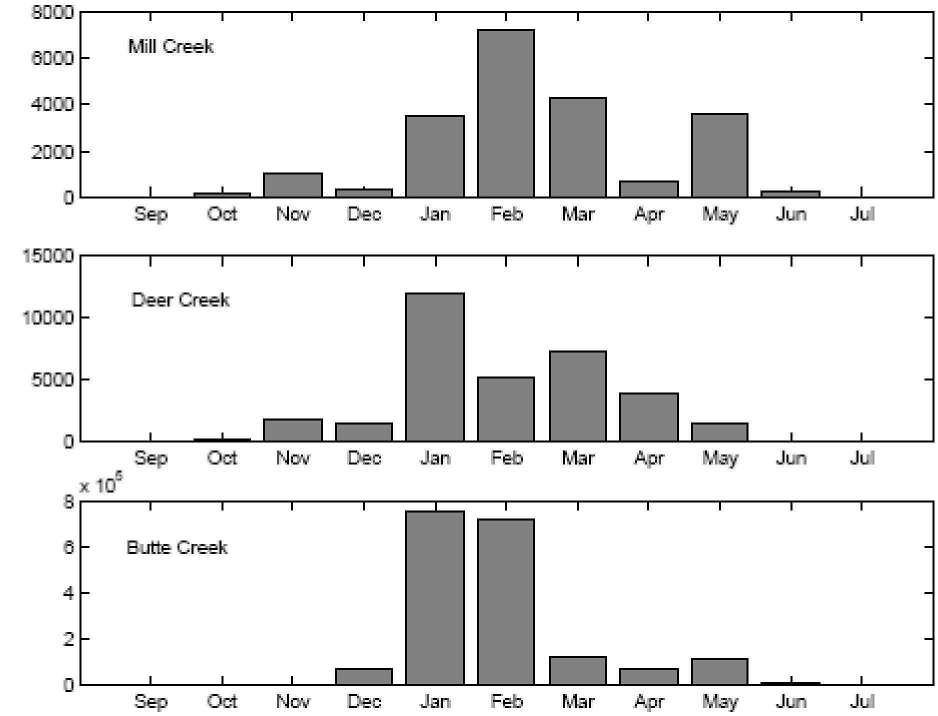

Figure 4.A.2-3. Mean monthly catches of juvenile CV spring-run Chinook salmon in rotary screw traps in Mill, Deer, and Butte Creeks (From Lindley et al. 2004)

4.A.2.3.5 Ocean Behavior

Spring-run Chinook salmon remain in the ocean environment for 2 to 4 years prior to returning to fresh water to spawn. CV spring-run Chinook salmon begin their ocean life in the coastal marine waters of the Gulf of the Farallones. Upon reaching the ocean, juveniles feed on larval and juvenile forage fishes, plankton, and other marine invertebrates (Healey 1991; MacFarlane and Norton 2002). Juveniles grow rapidly in the ocean environment with growth rates dependent on water temperatures and food availability (Healey 1991). In the Ocean, they are exposed to many stressors, including recreational and commercial harvest, prey availability due to changes in ocean currents, winds, and climate (Orsi and Davis 2013). The first year of ocean life is considered a critical period of high mortality for Chinook salmon that largely determines survival to harvest or spawning (Beamish and Mahnken 2001; Quinn 2005). Impacts from predators may be variable due to the availability of other prey (Orsi and Davis 2013). Low ocean sea temperatures may delay migration and reduce growth, thereby contributing to higher mortality (Orsi and Davis 2013).

4.A.2.4 Status and Trends

The CV spring-run Chinook salmon ESU has displayed broad fluctuations in adult abundance between 1960 and 2014 (Figure 4.A.2-4). The preliminary total spring-run Chinook salmon escapement (including all tributaries considered part of the ESU) for 2013 was 23,696 adults, which was the highest count since 2003 (30,697 adults) and over three times that of 2011 (7,774 adults) (CDFW 2014). Sacramento River tributary populations in Mill, Deer, and Butte Creeks are often considered the best indicators for the CV spring-run Chinook salmon ESU because these streams best represent historic populations of the ESU (as opposed to new populations emergent in Clear Creek, Battle Creek and the Feather River, for example). Generally, there was a positive trend in escapement in Deer, Mill, and Butte creeks between 1992 and 2015, (Figure 4.A.2-5). Adult spring-run Chinook salmon escapement to Mill, Deer, and Butte Creeks in 2013 was estimated to be 18,135 fish. Escapement numbers are dominated by Butte Creek returns, which typically represent approximately 70% of fish returning to these three creeks (CDFW 2014). In 2012, Battle Creek saw the highest number of returns in recent history (799 fish) (CDFW 2014). Individuals have only recently (1995) begun spawning in Battle Creek, where they spawned historically.

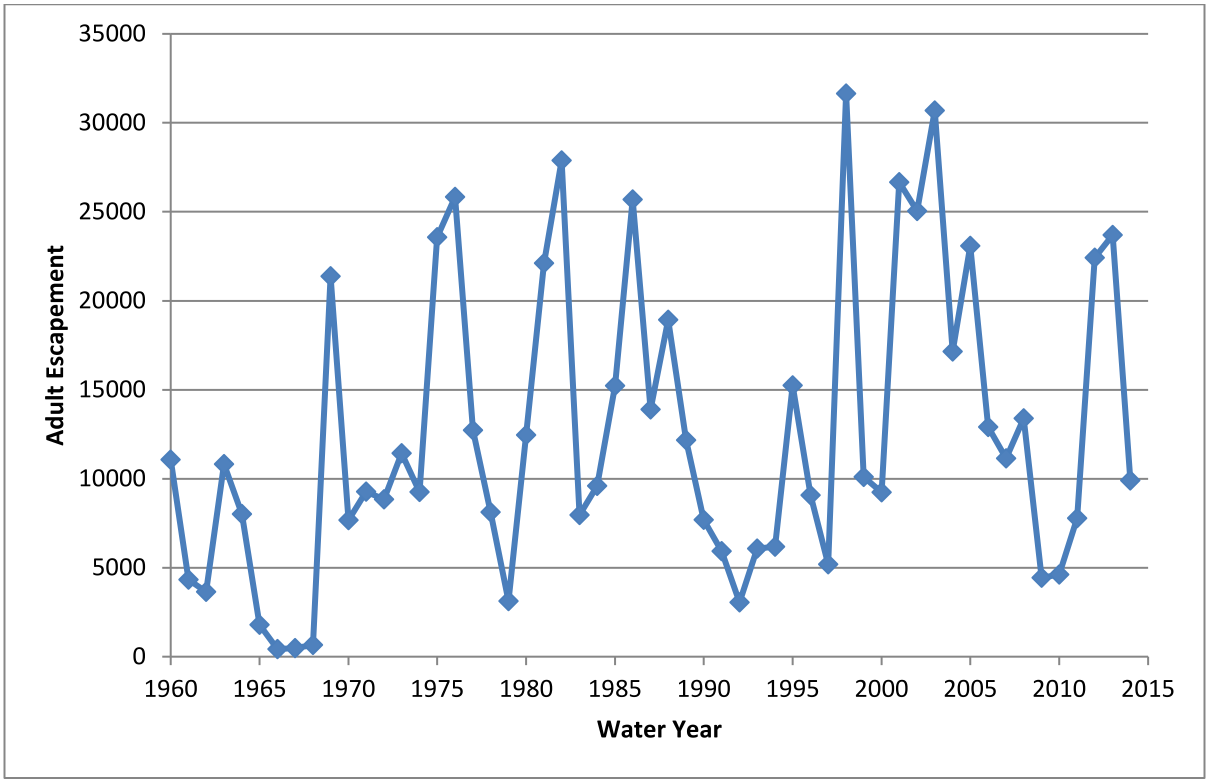

Figure 4.A.2-4. Historical Spawner Escapement of Central Valley Spring-Run Chinook Salmon throughout the Central Valley, including Returns to Feather River Fish Hatchery (1960–2013) (2009–2013 data are preliminary) (Source: GrandTab [Department of Fish and Wildlife 2014]).
Note: See GrandTab spring-run Chinook salmon table for notes concerning inclusion of fall-run Chinook salmon in some estimates.

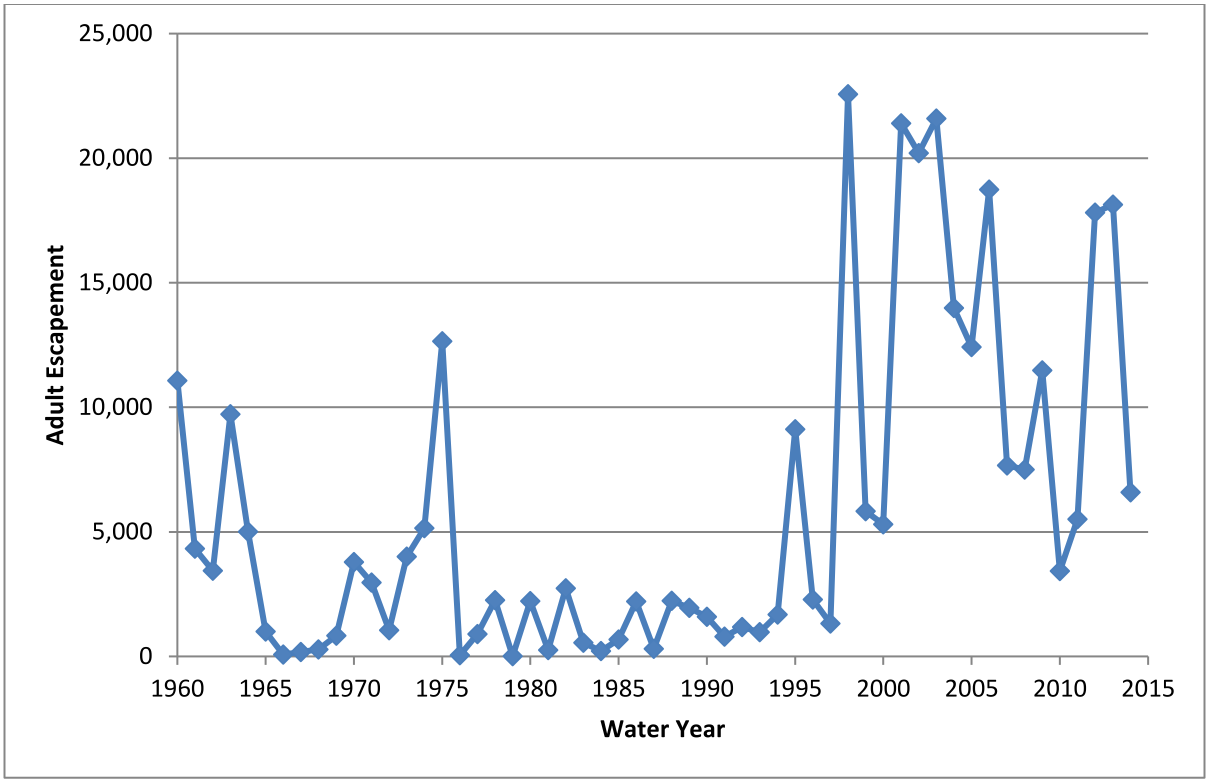

Figure 4.A.2-5. Historical Spawner Escapement of Central Valley Spring-Run Chinook Salmon in Mill, Deer, and Butte Creeks (1960–2013) (2009–2014 data are preliminary) (Source: Department of Fish and Wildlife 2014).
Note: Data from 2001 to 2014 use Butte Creek carcass survey estimates instead of snorkel survey estimates.

Between 1992 and 2012 there were significant habitat improvements in these watersheds, including the removal of several small dams, increases in summer flows, reduced ocean salmon harvest, and a favorable terrestrial and marine climate. The significant declines in adult fall-run Chinook salmon escapement during the late-2000s resulted in significant curtailment of the commercial and recreational salmon fisheries, which increased the level of protection for and benefit the CV spring-run Chinook salmon population.

On the Feather River, significant numbers of spring-run Chinook salmon as identified by run timing return to the Feather River Hatchery. However, coded-wire tag information from these hatchery returns and results of genetic testing indicate that substantial introgression has occurred between fall-run and spring-run Chinook salmon populations in the Feather River because of hatchery practices and the geographic and temporal overlap between the two runs in the river during spawning months. Nevertheless, Feather River spring-run Chinook salmon (including hatchery origin fish) are part of the ESU and thus their abundance and productivity contribute to the viability of the ESU.

Although recent CV spring-run Chinook salmon population trends are positive, annual abundance estimates display a high level of variation. The overall number of CV spring-run Chinook salmon remains well below estimates of historical abundance. CV spring-run Chinook salmon have some of the highest population growth rates in the Central Valley, but other than Butte Creek and Feather River, population sizes are very small relative to fall-run Chinook salmon populations (Good et al. 2005).

The historic component of the ESU is represented by three independent populations located in the same ecoregion and is therefore vulnerable to changes in the environment because it lacks spatial geographic diversity. The current geographic distribution of viable populations makes the CV spring-run Chinook salmon ESU vulnerable to catastrophic disturbance (Lindley et al. 2007; NMFS 2011). Such potential catastrophes include volcanic eruption of Mt. Lassen, prolonged drought conditions reducing coldwater pool adult holding habitat, and a large wildfire (approximately 30 kilometers maximum diameter) encompassing the Deer, Mill and Butte Creek watersheds. The CV spring-run Chinook salmon ESU remains at a moderate to high risk of extinction for the following reasons:

- The historic component of the ESU is spatially confined to relatively few remaining streams in its historical range.
- The ESU is composed of relatively small population that continues to display broad fluctuations in abundance.
- A large proportion of the population (in Butte Creek) faces the risk of high mortality rates resulting from high water temperatures during the adult holding period.

4.A.2.5 Threats and Stressors

NMFS (2014) described the threats and stressors to the CV spring-run Chinook salmon ESU as loss of historical spawning habitat, degradation of remaining habitat, reduced flows, warm temperatures, water withdrawals, commercial and recreational fisheries, and interactions with non-native fish and hatchery effects.

4.A.2.5.1 Reduced Access to and Quantity and Quality of Staging, Spawning, and Egg Incubation Habitat

Access to most of the historical upstream spawning habitat for spring-run Chinook salmon (Figure 4.A.2-1) has been eliminated or degraded by artificial structures (e.g., dams and weirs) associated with water storage and conveyance, flood control, and diversions and exports for municipal, industrial, agricultural, and hydropower purposes (Yoshiyama et al. 1998). Current spawning and juvenile rearing habitat is restricted to the mainstem Sacramento River and a number of its tributaries. Suitable summer water temperatures for adult and juvenile spring-run Chinook salmon holding and rearing are thought to occur at elevations from 492 to 1,640 feet (150 to 500 meters), most of which are now blocked by impassible dams. Habitat loss has resulted in a reduction in the number of independent spawning populations from an estimated 18-19 historically to 3 today (Good et al. 2005).

Upstream diversions and dams have decreased downstream flows and altered the seasonal hydrologic patterns. These factors have been identified as resulting in delayed upstream migration by adults, increased mortality of outmigrating juveniles, and are responsible for making some streams uninhabitable by spring-run Chinook salmon (Yoshiyama et al. 1998; DWR 2005). Dams and reservoir impoundments and associated reductions in peak flows have blocked gravel recruitment and reduced flushing of sediments from existing gravel beds, thereby reducing and degrading available spawning grounds. Further, reduced flows may decrease attraction cues for adult spawners, causing migration delays and increases in straying (DWR 2005). Adult salmon migration delays can reduce fecundity and increase susceptibility to disease and harvest (McCullough 1999).

Dams and other passage barriers also limit the geographic locations where spring-run Chinook salmon can spawn. In the Sacramento and Feather Rivers, restrictions to upstream movement and spawning site selection for spring-run Chinook salmon may increase the risk of hybridization with fall-run salmon, as well as co-occurrence contributing to an increased risk of redd superimposition. In creeks that are not affected by impassable dams, such as Deer and Mill Creeks, adult spring-run Chinook salmon have a greater opportunity to migrate upstream into areas where geographic separation from fall-run salmon reduces the risk of hybridization.

Up until 2012, spring-run Chinook salmon upstream migration season, the RBDD, located on the Sacramento River, was a barrier and impediment to adult spring-run Chinook salmon upstream migration. Although the dam was equipped with fish ladders, migration delays were reported when the dam gates were closed. Mortality from increased predation by Sacramento pikeminnow on juvenile salmon passing downstream through the fish ladder may also still affect abundance of salmon produced on the Sacramento River (Hallock 1991). The dam gates were placed in a permanent open position beginning in September 2011, and a new pump facility with a state-of-the-art fish screen was subsequently constructed. The elimination of dam gate operations is expected to benefit both upstream and downstream migration.

In the Feather River, all spring-run Chinook salmon spawning (and most fall-run Chinook salmon spawning) occurs in the low flow channel. Though suitable flows and spawning substrates are available downstream, colder water temperatures and proximity to the Feather River Fish Hatchery appear to draw most fish into the low flow channel. The proportion of salmon spawning in the low flow channel has increased significantly since the completion of the Oroville Complex and Feather River Hatchery (FRH). The significant shift in the distribution of salmon spawning in the Feather River to the upper reach of the low flow channel is perhaps one of the major factors affecting any in-channel production of CV spring-run Chinook salmon as a result of redd superimposition mortality. Since they spawn later in the fall, fall-run Chinook salmon may destroy a significant proportion of the redds of earlier spawning CV spring-run Chinook salmon (NMFS 2014 Appendix B).

In 2002, DWR conducted an Instream Flow Incremental Methodology (IFIM) habitat analysis for the lower Feather River (DWR 2004). This analysis drew on the earlier IFIM work of Sommer et al. (2001b), but added an additional 24 transects, and included additional spawning observations. The upper reach above Thermalito Afterbay Outlet (RM 59) to the Fish Barrier Dam (RM 67.25) and the lower reach below Thermalito Afterbay Outlet downstream to the confluence with Honcut Creek (RM 44) were modeled separately due to their distinct channel morphology and flow regime. The weighted usable area (WUA) for Chinook salmon spawning in the upper reach peaked at 800 cfs. In the lower reach, the WUA rises from the beginning modeled flow (500 cfs) and ~60% of maximum and then peaks near 1,700 cfs, after which it descends to a habitat index of ~30% of maximum at 7,000 cfs.

Since the ESU was listed as threatened in 1999, the availability of suitable tributary habitat for spring-run Chinook salmon has expanded. The removal of Seltzer Dam on Clear Creek in 2000 opened up 10 miles of habitat, and a consistent run of spring-run Chinook salmon has developed. The removal of a partial low-flow barrier on Cottonwood Creek in 2010 improved access to 30 miles of habitat (NMFS 2011). Additionally, the removal of Wildcat Dam in 2010 along with the completion of fish ladders at Eagle Canyon Dam and North Battle Feeder Dam opened up about 10 miles of habitat on Battle Creek, which, like Clear Creek, have now established what appears to be a consistent spring-run Chinook salmon population. The Battle Creek Salmon and Steelhead Restoration Project will eventually remove five dams on Battle Creek, install fish screens and ladders on three dams, and end the diversion of water from the North Fork to the South Fork. When the program is completed, a total of 42 miles of mainstem habitat and 6 miles of tributary habitat will be accessible to anadromous salmonids, including CV spring-run Chinook salmon (NMFS 2011). The Central Valley Recovery Plan for Salmonids (NMFS 2014) described criteria in order for the ESU to reach viable status, including spatial structure distribution to expand into nine total watersheds throughout the Central Valley.

The 2009 CVP/SWP BiOp includes a phased fish passage program, intended to expand spring-run Chinook salmon habitat to areas upstream of Shasta Dam. Phases of the fish passage program include habitat evaluations through January 2012, pilot reintroductions through January 2015, and implementation of the long-term program by January 2020 (NMFS 2011b).

High flows, such as those released from dams to draw down storage for flood control during heavy runoff periods, have the potential to scour spring-run Chinook salmon redds down to the depth of the eggs and injure eggs or sac-fry in the gravel, or to pile more gravel and fines on top of redds so that alevins are unable to emerge or are suffocated. These same flows are important for maintaining rearing habitat and high-quality spawning gravel. River-specific geomorphic studies evaluated the bedload mobilization flow for the affected rivers. The future probability of occurrence of flow releases exceeding the bedload mobilization flow is based on the historic hydrograph since the respective dam was constructed. This is because scouring flows are generally a result of flood control operations during high runoff periods, which will not likely change in the near future.

In Clear Creek, sampling was conducted at the U.S. Geological Survey (USGS) Clear Creek near Igo gauge during high flows in January and February 1998 to estimate a flow threshold that initiated coarse sediment transport (McBain & Trush and Matthews 2000). Sampling bedload movement during a 2,600 cfs flow showed that mainly sand was being transported. During a 3,200 cfs flow, medium gravels were being transported. Particles slightly greater than 32 millimeters (mm) were being transported by the 3,200 cfs (D84 = 7.5 mm) flow while no particles larger than 11 mm were sampled during the 2,600 cfs flow (D84 = 1.8 mm). Their initial estimate for a coarse sediment transport initiation threshold is in the 3,000 to 4,000 cfs range. Marked rock experiments at Reading Bar, the first alluvial reach downstream of the Clear Creek canyon, suggest that large gravels and cobbles (the D84) are not significantly mobilized by a 2,900 cfs flow.

The majority of post-Whiskeytown Dam floods are produced from tributaries downstream of Whiskeytown Dam, but floods larger than about 3,000 cfs are caused by uncontrolled spillway releases from Whiskeytown Dam, as happened in WY 1983 (19,200 cfs, the largest post-regulation flood), 1997 (15,900 cfs), and 1998 (12,900 cfs) floods. These flows are the result of heavy runoff from the upper Clear Creek watershed and are not affected by Reclamation water release operations. Reclamation does not make controlled releases into Clear Creek that exceed the bedload mobilization point. A probability of exceedance plot for Whiskeytown Dam is shown in Figure 4.A.2-6. Instantaneous flows of 3,000 cfs occur on average about once every 2 years and flows of 4,000 cfs occur about once every 3 years (Figure 4.A.2-7). One-day average flows of 3,000 cfs occur about once every 5 years.

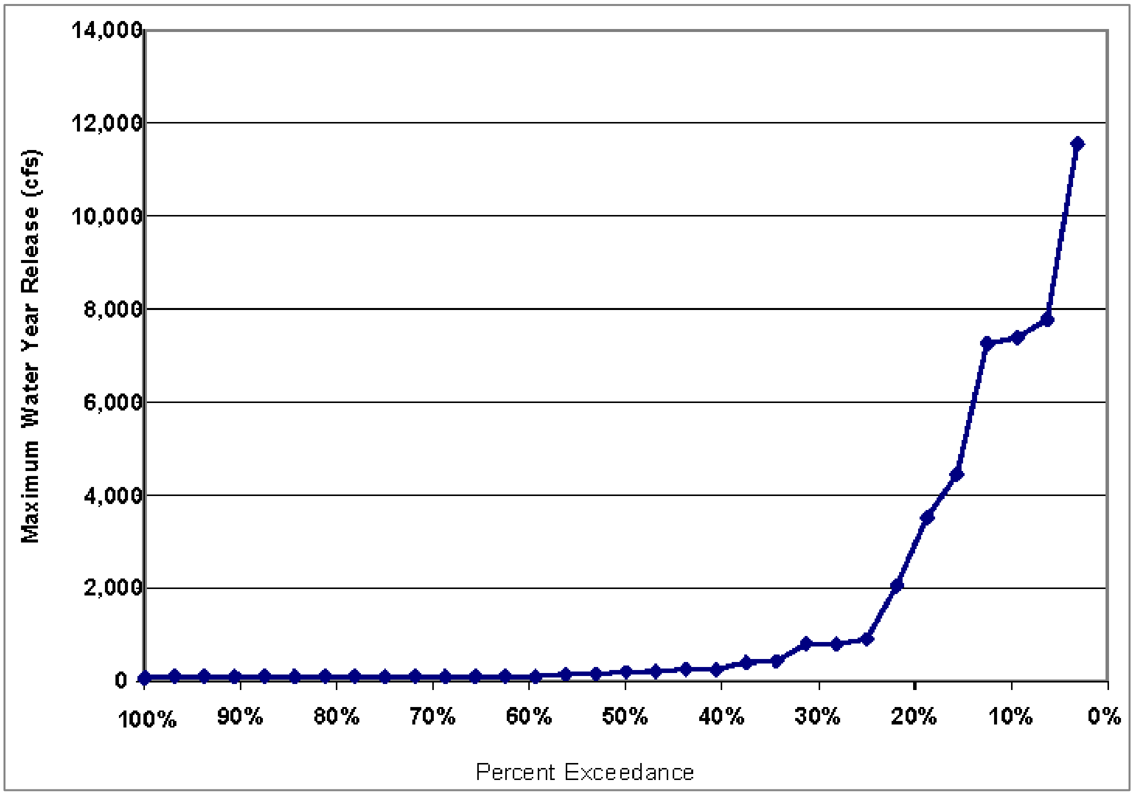

Figure 4.A.2-6. Yearly Probability of Exceedance for Releases from Whiskeytown Dam on Clear Creek based on Historical Dam Operations Records.

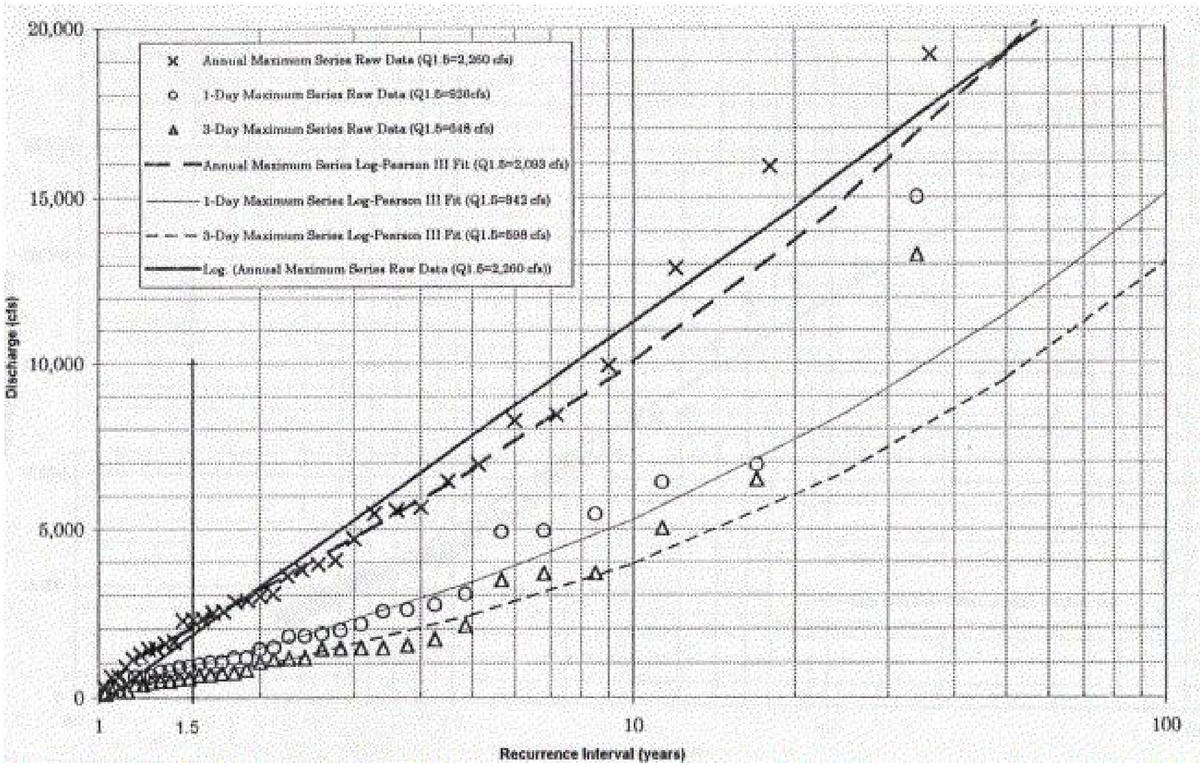

Figure 4.A.2-7. Clear Creek near Igo (Station 11-372000) Flood Frequency Analysis of Annual Maximum, 1-Day Average, and 3-Day Average Flood Series for Post-dam (1964–97) Data

Table 4.A.2-2 shows the stage discharge relationship in Clear Creek at Igo. Using the 5-inch redd depth as the threshold for redd dewatering, a 100-cfs flow drop in the 100 to 300 cfs range could start to dewater the shallowest redds. A flow drop of 150 cfs in the 300 to 800 cfs range could start to dewater redds, and a flow drop of 300 cfs between 800 and 1,800 cfs could start to dewater redds. Flows over 500 cfs in Clear Creek are the result of uncontrolled runoff or pulse flows prescribed through collaboration with fishery agencies for the benefit of fish and habitat.

Table 4.A.2-2. Stage Discharge Relationship for the Clear Creek at Igo USGS Gauge, Station 11372000.

Stage, inches	Discharge, cfs
33.12	101
38.52	200
42.72	301
46.2	400
49.32	501
52.2	602
54.72	702
57	803
59.16	903
61.08	1,000

4.A.2.5.2 Reduced Rearing and Out-Migration Habitat

Juvenile spring-run Chinook salmon prefer natural stream banks, floodplains, marshes, and shallow water habitats as rearing habitat during out-migration. Channel margins throughout the Delta have been leveed, channelized, and fortified with riprap for flood protection and island reclamation, reducing and degrading the quality of natural habitat available for juvenile Chinook salmon rearing (Brandes and McLain 2001). Artificial barriers further reduce and degrade rearing and migration habitat and delay juvenile out-migration. Juvenile out-migration delays can reduce fitness and increase susceptibility to diversion screen impingement, entrainment, disease, and predation. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening and altering the seasonal timing of the hydrograph, reducing the extent and duration of seasonal floodplain inundation and other flow-dependent habitat used by migrating juvenile Chinook salmon (70 FR 52488; September 2, 2005) (Sommer et al. 2001a; DWR 2005).

Recovery of floodplain habitat in the Central Valley has been found to contribute to increases in production in Chinook salmon (Sommer et al. 2001a), but little is known about the potential benefit available to migrating spring-run Chinook salmon.

The potential adverse effects of dam operations include reductions in seasonal river flows, delays in juvenile emigration, and increased seasonal water temperature. In addition, exposure to a higher proportion of agricultural return flows, and exposure to reduced dissolved oxygen concentrations (e.g., Stockton Deep Water Ship Channel) will likely affect the survival and success of reestablishing spring-run Chinook salmon on the San Joaquin River in the future (Regional Water Resources Control Board 2003).

4.A.2.5.3 Predation

For discussion on predation of Central Valley spring-run Chinook salmon, see Section 4.A.1.4.3, *Predation*, in Sacramento River winter-run Chinook as the effects of predation are essentially the same.

4.A.2.5.4 Harvest

Harvest Commercial and recreational harvest of spring-run Chinook salmon in the ocean and inland fisheries has been a subject of management actions by the California Fish and Game Commission and the Pacific Fishery Management Council. The primary concern is the incidental harvest of spring-run Chinook salmon as part of fisheries primarily targeting hatchery produced fall- and late fall-run salmon. Naturally reproducing spring-run Chinook salmon are less able to withstand high harvest rates when compared to hatchery-based stocks (California Hatchery Scientific Review Group 2012).

Central Valley origin Chinook salmon of all races are harvested in commercial and recreational fisheries off the coast of California. Central Valley origin fall-run Chinook salmon are the primary target of this harvest. Despite the relatively high abundance of hatchery-produced fall-run Chinook salmon, ocean fisheries are often constrained to protect ESA-listed Chinook salmon stocks (including Sacramento River winter-run Chinook salmon, Central Valley spring-run Chinook salmon, and California Coastal Chinook salmon), which constitute less than 10% of available Chinook salmon (Winans et al. 2001). This “mixed-stock” fishery is managed by using stock-specific differences in ocean distribution, age-at-maturity, size-at-date, and/or timing of river entry to help minimize harvest of sensitive stocks (NMFS 2000). However, such management strategies are only partially effective (NMFS 2010).

For example, spring-run Chinook salmon return to freshwater in the spring and thus avoid most ocean harvest during the year in which they mature. However, spring-run Chinook salmon that mature at age 4 (or older) are subjected to an additional full season of harvest at “impact levels” comparable to those directed at Central Valley fall-run Chinook salmon (Pyper et al. 2012). Harvest managers define “impact rate” as the proportion of a particular stock that will suffer mortality associated with the ocean fishery. Fall-run Chinook salmon often experience impact rates between 40 and 70% (PFMC) 2001).

Fifteen years have elapsed since NMFS last updated their spring-run Chinook salmon ocean harvest BiOp (NMFS 2000). The 2000 BiOp did not report an estimated “impact rate” for ocean harvest effects on spring-run Chinook salmon. The BiOp reached a non-jeopardy opinion for the impacts of ocean harvest primarily by referring to the growth in Central Valley spring-run Chinook salmon population that was occurring at that time. Though NMFS (2010) did not provide a quantitative analysis of spring-run Chinook salmon harvest, Grover et al. (2004) estimated that 2/3 of spring-run Chinook salmon matured at age 4, indicating that a large fraction of the spring-run Chinook salmon population is annually subject to very high impact rates (40 to 70%) which will greatly influence population productivity and abundance. Harvest of age-3 spring-run Chinook salmon is likely to be comparable to that experienced by winter- run Chinook salmon (which also mature and return to fresh water, missing most of the ocean fishing season). Though a comparable analysis for spring-run Chinook salmon is not available, Winship et al. (2013) applied a simulation model and showed that a 25% impact rate (much less than that likely experienced by age 4 spring-run Chinook salmon) on winter-run Chinook salmon substantially decreased population abundance and population resiliency relative to alternatives with less harvest.

Harvest pressure of this intensity can also alter diversity in age at-maturity, a critical factor for population viability (NMFS 2010). The ocean fishery is thought to select against fish that mature later because fish that would do so are vulnerable to harvest for more years (Ricker 1981; Hankin and Healey 1986; Franks and Lackey 2015), and age at maturity has moderate heritability (Hankin et al. 1993). As such, reduced ocean harvest would contribute substantially to age at-maturity diversity (certainly demographically, if not genetically) and thereby enhance population viability (Lewis et al. 2015). A downward shift in size and age at maturity also affect fitness by reducing fecundity and reproductive rates (Calduch-Verdiell et al. 2014). Since size and age-at-maturity are heritable, selection for earlier adult maturity leads to a feedback loop in which younger and smaller adults produce offspring that mature earlier at smaller sizes.

Because survivorship has been reduced in incubating eggs and rearing and emigrating wild salmon relative to hatchery-reared individuals, naturally reproducing populations are less able to withstand high harvest rates compared to hatchery-based stocks (Knudsen et al. 1999). NMFS (2011) reports that ocean harvest had not changed appreciably since the 2005 status review (Good et al. 2005), except for extreme reductions in 2008 through 2010. The ocean salmon fisheries were closed in 2008 and 2009 and substantially restricted in 2010.

Because adult spring-run Chinook salmon hold in pool habitats during the summer months, they are vulnerable to illegal harvest (poaching). Various watershed groups have established public outreach and educational programs in an effort to reduce poaching. In addition, CDFW wardens have increased enforcement against illegal harvest of spring-run Chinook salmon.

Reduced Genetic Diversity and Integrity Interbreeding of wild spring-run Chinook salmon with both wild and hatchery fall-run Chinook salmon has the potential to dilute and eventually eliminate the adaptive genetic distinctiveness and diversity of the few remaining naturally reproducing spring-run Chinook salmon populations (DFG 1995; Sommer et al. 2001b; Araki et al. 2007). CV spring- and fall-run Chinook salmon spawning areas were historically isolated in time and space (Yoshiyama et al. 1998). However, the construction of dams has eliminated access to historical upstream spawning areas of spring-run Chinook salmon in the upper tributaries and streams of many river systems. Restrictions to upstream access, particularly on the Sacramento and Feather Rivers, has forced CV spring-run Chinook salmon individuals to spawn in lower elevation areas also used by fall-run individuals, potentially resulting in hybridization of the two races. The importance of introgression and competition between spring- and fall-run Chinook salmon is demonstrated by the successful recovery of spring-run Chinook salmon in Clear Creek and Battle Creek. In both cases, management actions to limit access of fall-run Chinook salmon have allowed spring-run Chinook salmon population to quickly establish themselves.

Hybridization between spring- and fall-run salmon is a particular concern on the Feather River, where both runs are produced by the Feather River Hatchery. Management of the Feather River hatchery and brood stock selection practices have been modified in recent years (e.g., tagging early returning adult salmon showing phenotypic and run timing characteristics of spring-run Chinook salmon for subsequent use as selected brood stock and genetic testing of potential brood stock) in an effort to reduce potential hybridization as a result of hatchery operations. Future plans are being considered to use a physical weir to help segregate and isolate adults showing spring-run characteristics and later-arriving fish showing characteristics of fall-run fish to reduce the risk of hybridization and redd superimposition in spawning areas of the river.

In many of the other Central Valley tributaries, such as Deer and Mill Creeks, the risk of hybridization is reduced by the ability of the runs to segregate spatially in the watersheds.

Further, in an effort to improve juvenile survival and the contribution of the Feather River Hatchery to the adult spring-run Chinook salmon population, the spring-run Chinook salmon program at the hatchery has released juvenile spring-run Chinook salmon downstream of the hatchery (San Pablo Bay) in the past. This increased the straying rates into nonnatal spawning areas of adults migrating upstream to spawn (DFG 2001). Recent changes in hatchery management by CDFW, however, have modified juvenile planting with a greater number of juvenile fish released into the Feather River in an effort to improve imprinting and reduce straying, which may reduce potential for hybridization with spring-run Chinook salmon populations in other watersheds (McReynolds et al. 2006).

4.A.2.5.5 Entrainment

The vulnerability of juvenile spring-run Chinook salmon to entrainment and salvage at the CVP/SWP export facilities varies in response to multiple factors, including the seasonal and geographic distribution of juvenile salmon in the Delta, operation of Delta Cross Channel gates, hydrodynamic conditions i.e. instantaneous velocities and instantaneous velocity fields occurring in the central and southern regions of the Delta (Old and Middle Rivers),\ and export rates at project and nonproject facilities. The loss of fish to entrainment mortality affects has been hypothesized to affect Chinook salmon populations (Kjelson and Brandes 1989). Juvenile spring-run Chinook salmon reaching the central and southern Delta will have an increased risk of entrainment/salvage.

Tidally averaged flow (or net flow) in Old and Middle rivers (OMR flows) are often negative as a result of export through the Federal and state export facilities. The hydrodynamic conditions associated with negative OMR flows have been hypothesized by NMFS (2009b) to be associated with increased southward movement of emigrating juveniles in those channels resulting in delayed emigration through the Delta, and directly or indirectly increasing vulnerability to the many stressors within the central and south Delta.

Recent independent science reviews have observed that numerous parameters influence juvenile salmonid movement. These do not include tidally averaged flow, but do include instantaneous flow velocities which are perceived by the fish in its immediate surrounding environment, detection of chemical constituents in the water by chemo-sensory organs that elicit migratory behavioral responses, and spatial distribution of the migrating fish across the river channel in the vicinity of junctions that affect ultimate route selection (Anderson et al. 2012; Monismith et al. 2014). Nevertheless, previous studies have observed increased entrainment of tagged salmonids at the CVP/SWP facilities when exports are increased (NMFS 2009, Zeug and Cavallo 2014

CVP/SWP exports have been shown to affect water velocities and direction at locations nearer to the export facilities. Farther away from the export facilities, there is considerably smaller influence on the magnitude of the tidal flow and instantaneous velocities within the lower San Joaquin River channel (Cavallo et al. 2015).

Chinook salmon interact with complex velocity fields during both upstream adult and downstream juvenile migration through the Delta. Where velocity fields change as a result of CVP/SWP export operations during the period that salmon are migrating through Delta channels it may contribute to the use of false migration pathways, delays in migration, or increased movement of migrating salmon toward the export facilities leading to an increase in entrainment risk (Monismith et al. 2014). During the past several years, additional investigations have been designed using radio or acoustically tagged juvenile Chinook salmon to monitor migration behavior through the Delta channels and to assess the effects of changes in hydraulic cues and CVP/SWP export operations on migration (San Joaquin River Group Authority 2010; Delaney et al. 2014; Cavallo et al. 2015). These studies are ongoing.

Incidental take of juvenile spring-run Chinook salmon at the CVP/SWP export fish salvage facilities is routinely monitored and reported as part of export operations. Salvage monitoring and the protocol for identifying juvenile spring-run Chinook salmon from other Central Valley

Chinook salmon have been refined over the past decade. Run identification was originally determined based on the length of each fish and the date it was collected. Subsequent genetic testing has been used to refine species identification, but this is currently only effective for identifying spring-run Chinook salmon from Deer, Mill and Butte Creeks (Banks et al. 2000; Harvey et al. 2011). Unlike winter-run Chinook salmon, spring-run Chinook salmon do not have a specific incidental take allotment. Take of “large juveniles” at the export facilities are used as a surrogate for what may include spring-run Chinook salmon smolts. Analysis of CWT recoveries at export salvage indicate very low rates of loss for juvenile spring Chinook (Zeug and Cavallo 2014).

In addition to CVP/SWP exports, over 2,200 small water diversions exist throughout the Delta, along with unscreened diversions located on the tributary rivers (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978; Schneeberger and Jude 1981; Zeitoun et al. 1981; Weisberg et al. 1987). Many of the juvenile salmon migrate downstream through the Delta during the late winter or early spring when many of the agricultural irrigation diversions are not operating or are only operating at low levels. Juvenile salmon also migrate primarily in the upper part of the water column and are less vulnerable to an unscreened diversion located near the channel bottom. While unscreened diversions used to flood agricultural fields (e.g., rice fields) during the winter have the potential to divert and strand juvenile salmonids, there are no quantitative estimates of the potential magnitude of entrainment losses for juvenile Chinook salmon migrating through the rivers and Delta, although at a population level the effects are thought to be small (Moyle and White 2002). Draining these fields can also provide flow attractions to upstream migrating adult salmon, resulting in migration delays or stranding losses, although the loss of adult fish and the effects of these losses on the overall population abundance of returning adult Chinook salmon are also unknown.

Spring-run Chinook salmon may move into the Colusa Drain via Yolo Bypass into the Knights Landing Ridge-cut or up the Sacramento River, then moving through the Knights Landing outfall gates. Once in the canal fish migrate upstream until barriers are reached that prevent further migration. Unless rescued at these points they die and are lost to the population. In 2015 a pickett weir was installed in front of the Knights Landing Outfall Gates that should prevent most fish from moving through the radial gates.

Despite these potential detrimental effects, flooding agricultural fields can increase nutrient loading to downstream habitats and increase productivity, and increase base flows during low stream flow periods. Many of the larger water diversions located in the Central Valley and Delta (e.g., Glenn Colusa Irrigation District, Reclamation District 108 Wilkins Slough, Poundstone, and Sutter Mutual Water Company Tisdale Pumping Plants, Contra Costa Water District Old River and Alternative Intake Project, and others) have been equipped with positive barrier fish screens to reduce and avoid the loss of juvenile Chinook salmon and other fish species.

4.A.2.5.6 Exposure to Toxins

Toxic chemicals have the potential to be widespread throughout the Delta, or may occur on a more localized scale in response to episodic events (stormwater runoff, point source discharges). These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to affect fish health and condition, and adversely affect salmon distribution and abundance. Chinook salmon may experience both waterborne chronic and acute exposure, but also bioaccumulation and chronic dietary exposure. For example, selenium is a naturally occurring constituent in the return flow of agricultural drainage water from the San Joaquin River that is then dispersed downstream into the Delta (Nichols et al. 1986). Exposure to selenium in the diet of juvenile Chinook salmon results in toxic effects (Saiki and Lowe 1987; Hamilton et al. 1986, 1990; Hamilton and Buhl 1990). Selenium exposure has been associated with agricultural and natural drainage in the San Joaquin River basin and petroleum refining operations adjacent to San Pablo and San Francisco Bays. Other contaminants of concern for Chinook salmon include, but are not limited to, mercury, copper, oil and grease, pesticides, herbicides, ammonia[2], and localized areas of depressed dissolved oxygen (e.g., Stockton Deep Water Ship Channel, return flows from managed freshwater wetlands). As a result of the extensive agricultural development in the Central Valley, exposure to pesticides and herbicides is a significant concern for salmon and other fish species in the Plan Area (Bennett et al. 2001). In recent years, changes have been made in the composition of herbicides and pesticides used on agricultural crops in an effort to reduce potential toxicity to aquatic and terrestrial species. Modifications have also been made to water system operations and agricultural wastewater discharges (e.g., agricultural drainage water system lock-up and holding prior to discharge) and municipal wastewater treatment and discharges. Concerns remain, however, regarding the toxicity of contaminants such as pyrethroids that adsorbed to sediments and other chemicals (selenium and mercury, as well as other contaminants) on salmon.

Mercury and other metals such as copper have also been identified as contaminants of concern for salmon and other fish as a result of direct toxicity and impacts such as those related to acid mine runoff from sites such as Iron Mountain Mine (U.S. Environmental Protection Agency 2006). Tissue bioaccumulation may adversely affect the fish, but also represents a human health concern (Gassel et al. 2008). These materials originate from a variety of sources, including mining operations, municipal wastewater treatment, agricultural drainage in the tributary rivers and Delta, nonpoint runoff, natural runoff, and drainage in the Central Valley, agricultural spraying, and a number of other sources. The State Water Resources Control Board (State Water Board), Central Valley Regional Water Quality Control Board, U.S. Environmental Protection Agency (EPA), U.S. Geological Survey (USGS), DWR (DWR), and others have ongoing monitoring programs designed to characterize water quality conditions and identify potential toxicants and contaminant exposure to Chinook salmon and other aquatic resources in the Plan Area. Programs are in place to regulate point source discharges as part of the National Pollutant Discharge Elimination System (NPDES) program as well as efforts to establish and reduce total daily maximum loads (TMDL) of various constituents entering the Delta. Regulations have been updated to help reduce chemical exposure and adverse effects on aquatic resources and habitat conditions in the Plan Area. These monitoring and regulatory programs are ongoing.

Sublethal concentrations of toxics may interact with other stressors on salmonids, possibly increasing their vulnerability to mortality because of exposure to seasonally elevated water temperatures, predation, or disease (Werner 2007). For example, Clifford et al. (2005) found in a

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[2] Ammonia in water generally forms some amount of ammonium. Therefore, the use of the term *ammonia* implies that both ammonia and ammonium may be present.

laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common pyrethroid, esfenvalerate, were more susceptible to infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Although not tested on spring-run Chinook salmon, a similar response is likely due to the physiological similarity.

Iron Mountain Mine, located adjacent to the upper Sacramento River, has been a source of trace elements and metals that are known to adversely affect aquatic organisms (Upper Sacramento River Fisheries and Riparian Habitat Advisory Council 1989). Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (Bureau of Reclamation 2004). The EPA's Iron Mountain Mine remediation program has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s.

4.A.2.5.7 Increased Water Temperature

Water temperature is among the physical factors that affect the value of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive life stages, such as during incubation or rearing. The Central Valley is the southern limit of spring-run Chinook salmon geographic distribution, so increased water temperatures are often recognized as an important stressor to California populations.

The tolerance of spring-run Chinook salmon to water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors such as predator avoidance (Myrick and Cech 2004; Reclamation 2004). Higher water temperatures can lead to physiological stress, reduced growth rate, prespawning mortality, reduced spawning success, and increased mortality of salmon (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2008). Exposure to seasonally elevated water temperatures may occur because of reductions in flow, upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate, and solar radiation.

The recommendations included in this Biological Assessment (BA) were developed by Boles et al. (1988) based on previous temperature studies of Chinook salmon and other salmonids (Table 4.A.2-3). An overview of temperature effects on Chinook salmon follows.

Table 4.A.2-3. Recommended Water Temperatures for All Life Stages of Chinook Salmon in Central Valley Streams as Presented in Boles et al. (1988).[a]

Life Stage	Temperature (°F)
Migrating adult	<65
Holding adult	<60
Spawning	53 to 57.5[b]
Egg incubation	<55
Juvenile rearing	53 to 57.5[c]
Smoltification	<64[d]

[a] The lower thermal limit for most life stages was about 38°F.
[b] Can have high survival when spawned at up to 60°F, provided temperatures drop quickly to less than 55°F.
[c] Temperature range for maximum growth rate based on Brett (1952, as cited in Boles et al. 1988).
[d] Marine and Cech 2004
Note: °F = degrees Fahrenheit.

The temperature recommendation for migrating adults was based on Hallock et al. (1970, as cited in Boles et al. 1988), who found Chinook immigration into the San Joaquin River was impeded by temperatures of 70°F, but resumed when the temperature fell to 65°F. There was also a low dissolved oxygen correlation in timing.

The temperature recommendations for adult holding and spawning, and for egg incubation were based on laboratory studies of Sacramento River Chinook egg survival (Seymour 1959). Egg mortality was high at constant temperature of 60°F, but was considerably reduced at temperatures between 55°F and 57.5°F. However, sac-fry mortality remained very high (greater than 50%) at temperatures above 56°F, presumably due to “aberrations in sequential physiological development.” These were long-duration experiences that are not representative of river conditions. Table 4.A.2-4 shows the relationship between water temperature and mortality of Chinook eggs and pre-emergent fry compiled from a variety of studies. This is the relationship used for comparing egg mortality between scenarios. USFWS (1998) conducted studies to determine Sacramento River winter-run and fall-run Chinook early life temperature tolerances. They found that higher alevin mortality can be expected for winter-run Chinook salmon between 56°F and 58°F. Mortality at 56°F was low and similar to fall-run Chinook mortality at 50°F. Their relationships between egg and pre-emergent fry mortality and water temperature were about the same as that used in the mortality model in this BA (Table 4.A.2-4).

Table 4.A.2-4. Relationship between Water Temperature and Mortality of Chinook Salmon Eggs and Pre-emergent Fry used in the Reclamation Egg Mortality Model.

Water Temperature (EF)[a]	Egg Mortality[b]	Instantaneous Daily Mortality Rate (%)	Pre-Emergent Fry Mortality[b]	Instantaneous Daily Mortality Rate (%)
41-56	Thermal optimum	0	Thermal optimum	0
57	8% @ 24d	0.35	Thermal optimum	0
58	15% @ 22d	0.74	Thermal optimum	0
59	25% @ 20d	1.40	10% @ 14d	0.75
60	50% @ 12d	5.80	25% @ 14d	2.05
61	80% @ 15d	10.70	50% @ 14d	4.95
62	100% @12d	38.40	75% @ 14d	9.90
63	100% @11d	41.90	100% @ 14d	32.89
64	100% @ 7d	65.80	100% @10d[c]	46.05

[a] This mortality schedule was compiled from a variety of studies each using different levels of precision in temperature measurement, the lowest of which was whole degrees Fahrenheit ($\pm 0.5^{o}$F). Therefore, the level of precision for temperature inputs to this model is limited to whole degrees Fahrenheit.

[b] These mortality schedules were developed by the USFWS and DFG for use in evaluation of Shasta Dam temperature control alternatives in June 1990 (Richardson et al. 1990)

[c] This value was estimated similarly to the preceding values but was not included in the biological assumptions for Shasta outflow temperature control FES (Reclamation 1991, as cited in U.S. Bureau of Reclamation 2008).

Temperature compliance points in the Sacramento River (generally between Bend Bridge and Balls Ferry) vary by water year type and date between April 15 and October 31 for winter-run Chinook salmon spawning, incubation, and rearing. The objective is to meet a daily average temperature of 56°F for incubation and 60°F for rearing. After October 31, natural cooling generally provides suitable water temperatures for all Chinook life cycles.

The theoretical upper lethal temperature that Sacramento River Chinook salmon can tolerate has been reported as 78.5°F (Orsi 1971, as cited in Boles et al. 1988). However, this result must be interpreted with several things in mind.

First, the theoretical maximum corresponds to the most temperature-tolerant individuals. It is not a generality that can be applied to an entire stock. Second, it is only a 48-hour LT 50 (lethal time for 50% mortality). This means it is a temperature that can only be tolerated for a short period. It does not indicate a temperature at which a Chinook could feed and grow. Third, indirect mortality factors (for example, disease and predation) would likely lead to increases in total mortality at temperatures well below this theoretical laboratory-derived maximum. For example, Banks et al. (1971, as cited in Boles et al. 1988) found Chinook growth rates were not much higher at 65°F than at 60°F, but the fish had higher susceptibility to disease at 65°F. Subacute and sublethal temperature thresholds have been identified for Central Valley Chinook salmon by Marine and Cech (2004). Sublethal impairment of predator avoidance, smoltification, and disease begins in the range of about 64° to 68°F.

Myrick and Cech (2001) show that Chinook salmon that complete juvenile and smolt phases in the 50 to 62°F range are optimally prepared for saltwater survival. Marine and Cech (2004) identified a smoltification threshold of <64 F for Central Valley Chinook salmon.

It is also important to note that operation of CVP/SWP facilities cannot influence (1) the water temperatures on many of the tributaries to the Sacramento and San Joaquin Rivers or (2) those other factors that affect water temperatures that are unrelated to the appropriation of water for use by the CVP/SWP. Reclamation is not aware of any actions taken by others to address those other factors that are beyond the control of Reclamation and DWR that influence water temperatures.

The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool, has reduced many of the temperature issues on the Sacramento River. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows further downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. As a result of the longitudinal gradient of seasonal water temperatures, the coldest temperatures and best areas for salmon spawning and rearing are typically located immediately downstream of the dam. Climate change modeling predicts that the Butte Creek run of spring-run Chinook (the largest population of spring-run Chinook) will be extirpated as a result of warming temperature, even with the cessation of water and power operations (Thompson et al. 2011).

Adult and juvenile spring-run Chinook salmon hold and rear in pools at higher elevations in the watershed. On several tributaries, prespawning adult mortality has been reported for adults that accumulate in high densities in a pool and are then exposed to elevated summer water temperatures. Flow reductions, resulting from natural hydrologic conditions during the summer, evapotranspiration, or surface and groundwater extractions may all contribute to exposure to elevated temperatures and increased levels of stress or mortality. In some areas, groundwater wells have been used to pump cooler water into the stream to reduce summer temperatures. Dense riparian vegetation, streams incised into canyons that provide shading, cool water springs, and availability of deep holding pools are factors that affect summer holding and rearing conditions for spring-run Chinook salmon.

The effects of climate change and global warming patterns, in combination with changes in precipitation and seasonal hydrology in the future are important factors that may adversely affect the health and long-term viability of CV spring-run Chinook salmon (Crozier et al. 2008). The rate and magnitude of these potential future environmental changes, and their effect on habitat value and availability for spring-run Chinook salmon, however, are subject to a high degree of uncertainty.

4.A.2.6 Description of Viable Salmonid Population (VSP) Parameters

As a way of measuring the conservation status of salmonids, NMFS developed the viable salmonid population (VSP) framework that is used to identify the attributes that can be used to assess the effects of management and conservation actions. The framework is known as the VSP concept (McElhany *et al*. 2000). The VSP concept measures population performance in term of four key parameters: abundance, population growth rate, spatial structure, and diversity. ESU/DPS viability is dependent on the number of populations, their separate status, their spatial relationship to each other, potential sources and likelihood of catastrophic disturbance, and the variability within each population and its habitat (NMFS 2014).

4.A.2.6.1 Abundance

The NMFS (2014) Recovery Plan set out criteria for moderate and low risk of extinction. Moderate risk criteria include census population size of 250 to 2,500 adults, or an effective population size of 50 to 500 adults. Low risk criteria include a census population size of greater than 2,500 adults, or an effective population size that is greater than 500 adults.

The historical abundance of Sacramento River spring-run Chinook salmon prior to commercial harvesting is difficult to quantify, in part, because the distinct nature of the run was not recognized by early workers (Yoshiyama et al. 1998). However, it is inferred from historical catch data that the abundance of spring-run Chinook salmon for the entire Central Valley prior to the 20th century numbered 700,000 fish, second only to fall-run (900,000 fish) (Yoshiyama et al. 1998). In 1878, nearly 200,000 salmon were individually counted on the McCloud River during a 40-day period preceding October 5; based on these observations, it is presumed that these salmon were mostly, if not solely, spring-run Chinook salmon (Yoshiyama et al. 1998). Spring-run Chinook salmon supported a substantial fishery in the late 1800s, with 567,000 fish reportedly caught in the Sacramento-San Joaquin commercial fishery in 1883 alone (Yoshiyama 1998).

Spring-run Chinook salmon may have been originally most abundant in the San Joaquin River system, which has a hydrology that is more snow-driven, compared to the more rain-driven Sacramento River system (Yoshiyama et al. 2001). Prior to construction of Friant Dam in 1939, nearly 50,000 spring-run Chinook salmon were counted in the San Joaquin River (NMFS 2014). By 1951, spring-run Chinook salmon were considered extirpated from the San Joaquin River system (Yoshiyama et al. 1998). Population estimates of returning spring-run Chinook salmon for the years immediately preceding and after the closure of Friant Dam in February 1944 are as follows (Fry 1961; Yoshiyama et al. 1998):

- 35,000 in 1943
- 5,000 in 1944
- 56,000 in 1945
- 30,000 in 1946
- 6,000 in 1947
- 2,000 in 1948

As discussed previously under *Status and Trends*, the population of spring-run Chinook salmon in the Central Valley (including all tributaries included in the ESU) has displayed broad fluctuations in adult abundance between 1960 and 2013, ranging from 427 adults in 1966 to 31,649 adults in 1998 (Figure 4.A.2-4). Presently, the only streams in the Central Valley that appear to host independent CV spring-run Chinook salmon populations of spring-run Chinook salmon are Battle, Clear, Butte, Deer, and Mill Creeks, and the Feather and Yuba Rivers. Although these populations are small (compared to historic numbers) and isolated, they are probably the best long-term indicators for population trends in the Central Valley. Figure 4.A.2-8 shows the annual run size estimates for these three populations since 1960. Generally, there was a positive trend in adult escapement for these three populations between 1992 and 2014. Adult spring-run Chinook salmon escapement to Mill, Deer, and Butte Creeks in 2013 was estimated to be 18,135 fish, the sixth largest escapement estimate since 1960, although the total escapement in 2014 declined to 6,592. Escapement numbers are dominated by Butte Creek returns, which typically represent approximately 70% of fish returning to these three creeks (CDFW 2014).

In Butte Creek, high water temperatures coupled with high fish densities have contributed to significant pre-spawning mortality of adults in the recent past. In 2002 and 2003, mean water temperatures in Butte Creek exceeded 21°C for 10 or more days in July (Williams 2006), which led to outbreaks of Columnaris (*Flexibacter columnaris*) and Ichthyophthiriasis (*Ichthyophthirius multifiis*) and the resultant loss of 20 to 30% of prespawning adults in 2002 and 65% (11,231) prespawning adults in 2003.

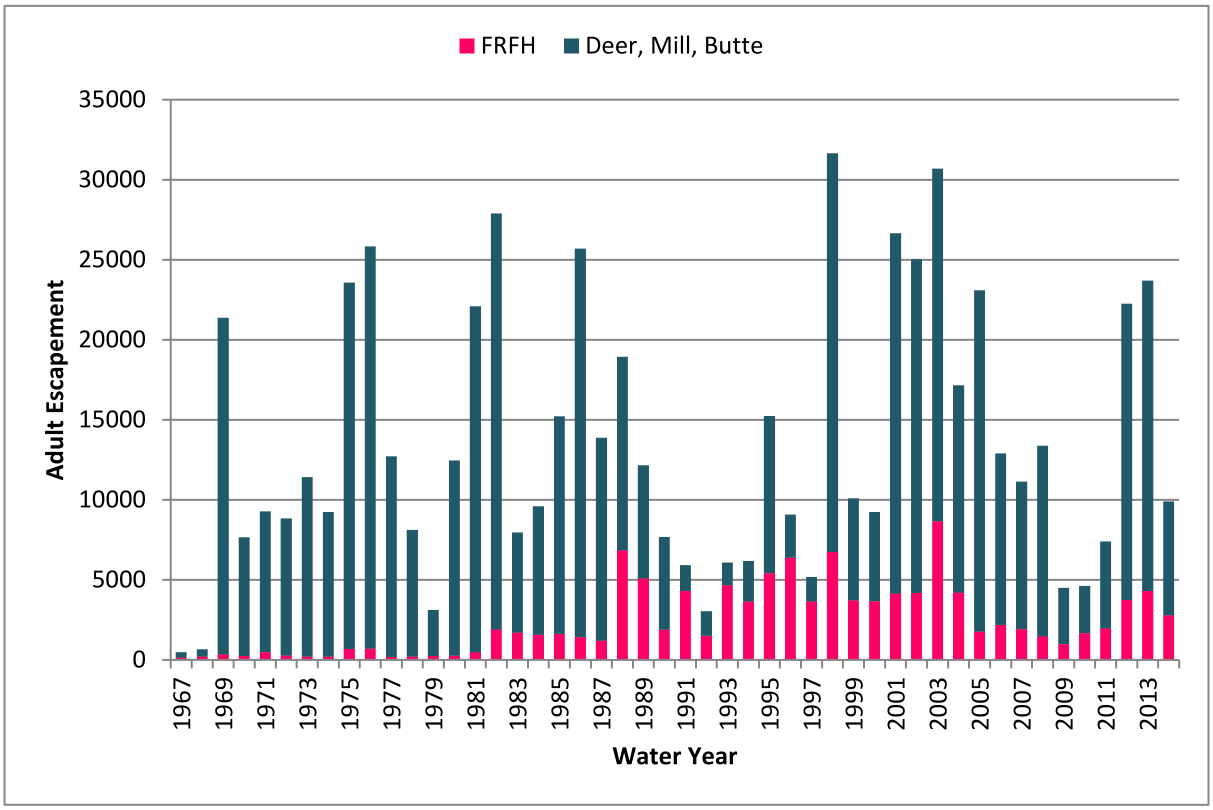

Figure 4.A.2-8. Composition of Central Valley Spring-Run Chinook Salmon Escapement, 1967–2013 (Source: CDFW 2014)

Spring-run Chinook salmon populations are beginning to be established in other Sacramento River tributaries. Escapement estimates in Clear Creek and Battle Creek generally increased from 2001 to 2013, with peak returns of 799 adults in Battle Creek in 2012 and 659 adults in Clear Creek in 2013 (Table 4.A.2-5).

Table 4.A.2-5. Number of Adult Spring-Run Chinook Salmon Returning to Other Sacramento River Tributaries, 1993–2014.

Water Year	Battle Creek	Clear Creek	Cottonwood Creek	Antelope Creek	Big Chico Creek
1993		1	1	3	38
1994		0		0	2
1995	66	2	8	7	200
1996	35		6	1	2
1997	107		0	0	2
1998	178	47	477	154	369
1999	73	35	102	40	27
2000	78	9	122	9	27
2001	111	0	245	8	39
2002	222	66	125	46	0
2003	221	25	73	46	81
2004	90	98	17	3	0
2005	73	69	47	82	37
2006	221	77	55	102	299
2007	291	194	34	26	0
2008	105	200	0	3	0
2009	194	120	0	0	6
2010	172	21	15	17	2
2011	157	8	2	6	124
2012	799	68	1	1	0
2013	608	659	1	0	0
2014	429	95	2	7	0

Hatchery spring-run Chinook salmon have been released from the FRFH since 1967 (Figure 4.A.2-9), and it is the only hatchery in the Central Valley that produces spring-run Chinook salmon (NMFS 2014). A significant portion (up to 1,000,000 smolts) of the CV spring-run Chinook salmon production has been released to acclimation net pens in San Pablo Bay (NMFS 2014), The annual spring-run Chinook salmon production target for FRFH is a maximum of 2 million smolts, which can be achieved by artificially spawning approximately 1,500 adults (i.e., 750 males and 750 females) (Cavallo et al. 2009).

The FRFH was originally constructed and managed to mitigate for the loss of Chinook salmon and steelhead spawning habitat from construction of Oroville Dam. Presently, the spring-run Chinook salmon program at FRFH is managed as an Integrated Recovery Program, which seeks to aid in the recovery and conservation of CV spring-run Chinook—that is, fish produced at FRFH are intended to spawn in the wild or be genetically integrated with the targeted natural population (Cavallo et al. 2009). As such, the FRFH spring-run Chinook salmon population is included in the CV spring-run Chinook salmon ESU based on its genetic linkage to the natural population and the potential development of a conservation strategy for the hatchery program.

Prior to 2004, the FRFH was operated by opening the ladder to the hatchery on September 1 and differentiating spring-run from fall-run adults by classifying adults that ascended the ladder from September 1 through September 15 as spring-run Chinook salmon. Because of concerns that this practice was leading to hybridization between spring- and fall-run Chinook salmon, hatchery operations were modified. Since 2007, the FRFH has been operated by keeping the fish ladder open from September 15 through June 30. Adult spring-run Chinook salmon ascending the ladder are marked with an external tag and returned to the river so that they can be identified as phenotypic CV spring-run Chinook salmon when they re-enter the ladder in September (NMFS 2014).

CV spring-run Chinook salmon populations are consistently found in the Feather River. Though this run is strongly influenced by Feather River Hatchery production, Feather River spring-run Chinook salmon (including hatchery origin fish) are part of the ESU and thus their abundance and productivity contribute to the viability of the ESU (NMFS 2011).

Since 1967, the number of adult spring-run Chinook salmon returning to the Feather River Fish Hatchery (FRFH) has ranged from about 6% to 77% of the annual escapement in the Central Valley (Figure 4.A.2-8).

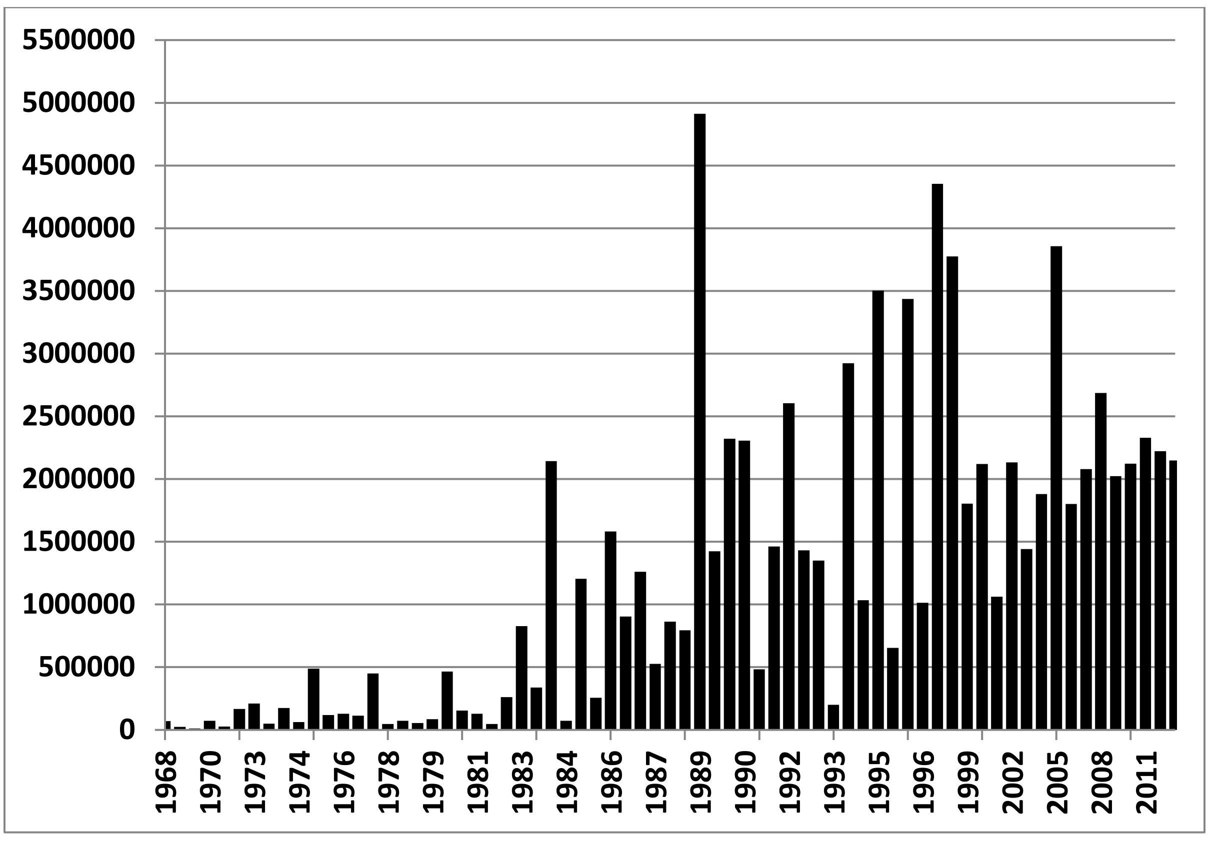

Figure 4.A.2-9. Number of Spring-run Chinook Salmon Released from Feather River Fish Hatchery, 1968–2013

4.A.2.6.2 Productivity

The NMFS (2014) Recovery Plan set out criteria for moderate and low risk of extinction. Moderate risk criteria for productivity are met when the run size has dropped below 500 individuals but is stable. Low risk is when no productivity decline is measureable. Long-term population growth and how the population varies temporally are important in analyzing a population's extinction subtleties (Lande 1993, 1998, Middleton and Nisbet 1997). Using this assumption past performance provides a useful predictor of future population dynamics. Populations should exhibit future tendencies that are consistent with those observed in the past in terms of the mean trajectory and variation exhibited over time. Cohort replacement rates (CRR) are indications of whether a cohort is replacing itself in the next generation (Table 4.A.2-6).

Table 4.A.2-6. Central Valley Spring-run Chinook Salmon Population Estimates from CDFW Grand Tab (2014) with Corresponding Cohort Replacement Rates for Years since 1986.

Year	Sacramento River Basin Escapement Run Size[a]	FRFH Population	Tributary Populations	5-Year Moving Average Tributary Population Estimate	Trib CRR[b]	5-Year Moving Average of Trib CRR	5-Year Moving Average of Basin Population Estimate	Basin CRR	5-Year Moving Average of Basin CRR
1986	3,638	1,433	2,205						
1987	1,517	1,213	304						
1988	9,066	6,833	2,233						
1989	7,032	5,078	1,954		0.89			1.93	
1990	3,485	1,893	1,592	1,658	5.24		4,948	2.30	
1991	5,101	4,303	798	1,376	0.36		5,240	0.56	
1992	2,673	1,497	1,176	1,551	0.60		5,471	0.38	
1993	5,685	4,672	1,013	1,307	0.64	1.54	4,795	1.63	1.36
1994	5,325	3,641	1,684	1,253	2.11	1.79	4,454	1.04	1.18
1995	14,812	5,414	9,398	2,814	7.99	2.34	6,719	5.54	1.83
1996	8,705	6,381	2,324	3,119	2.29	2.73	7,440	1.53	2.03
1997	5,065	3,653	1,412	3,166	0.84	2.77	7,918	0.95	2.14
1998	30,534	6,746	23,788	7,721	2.53	3.15	12,888	2.06	2.23
1999	9,838	3,731	6,107	8,606	2.63	3.26	13,791	1.13	2.24
2000	9,201	3,657	5,544	7,835	3.93	2.44	12,669	1.82	1.50
2001	16,869	4,135	12,734	9,917	0.54	2.09	14,301	0.55	1.30
2002	17,224	4,189	13,035	12,242	2.13	2.35	16,733	1.75	1.46
2003	17,691	8,662	9,029	9,290	1.63	2.17	14,165	1.92	1.43
2004	13,612	4,212	9,400	9,948	0.74	1.79	14,919	0.81	1.37
2005	16,096	1,774	14,322	11,704	1.10	1.23	16,298	0.93	1.19
2006	10,948	2,181	8,767	10,911	0.97	1.31	15,114	0.62	1.21
2007	9,726	2,674	7,052	9,714	0.75	1.04	13,615	0.71	1.00
2008	6,368	1,624	4,744	8,857	0.33	0.78	11,350	0.40	0.69
2009	3,801	989	2,812	7,539	0.32	0.69	9,388	0.35	0.60
2010	3,792	1,661	2,131	5,101	0.30	0.54	6,927	0.39	0.49
2011	5,036	1,969	3,067	3,961	0.65	0.47	5,745	0.79	0.53
2012	14,548	3,738	10,810	4,713	3.84	1.09	6,709	3.83	1.15
2013	23,696	4,294	19,402	7,644	9.10	2.84	10,175	6.25	2.32
2014	9,901	2,776	7,125	8,507	2.32	3.24	11,395	1.97	2.64
Median	9,066	3,657	4,744	7,644	1.03	1.94	10,175	1.09	1.37

[a] Only the escapement numbers from the Feather River Fish Hatchery (FRFH) and the Sacramento River tributaries are in this table. Sacramento River Basin run size is the sum of the escapement numbers from the FRFH and the tributaries.
[b] Abbreviations: CRR = Cohort Replacement Rate, Trib = tributary

The cohort replacement rate (CRR), which is a measure of the population's growth rate, is shown in Figure 4.A.2-10 for tributary and Sacramento River basin populations for brood years 1989 through 2013. The corresponding 5-year average CRR for brood years 1989 through 2013 is shown in Figure 4.A.2-11. Tributary and Sacramento River basin CCRs have fluctuated widely, but were generally positive (i.e., greater than 1.0) from 1989 through 2003 and were negative (i.e., less than 1.0) from 2004 through 2011. The positive CCR in 2011–2013, especially for the tributary population, suggests an increasing trend in the population following low abundance during 2004–2011 (Figure 4.A.2-10 and Figure 4.A.2-11).

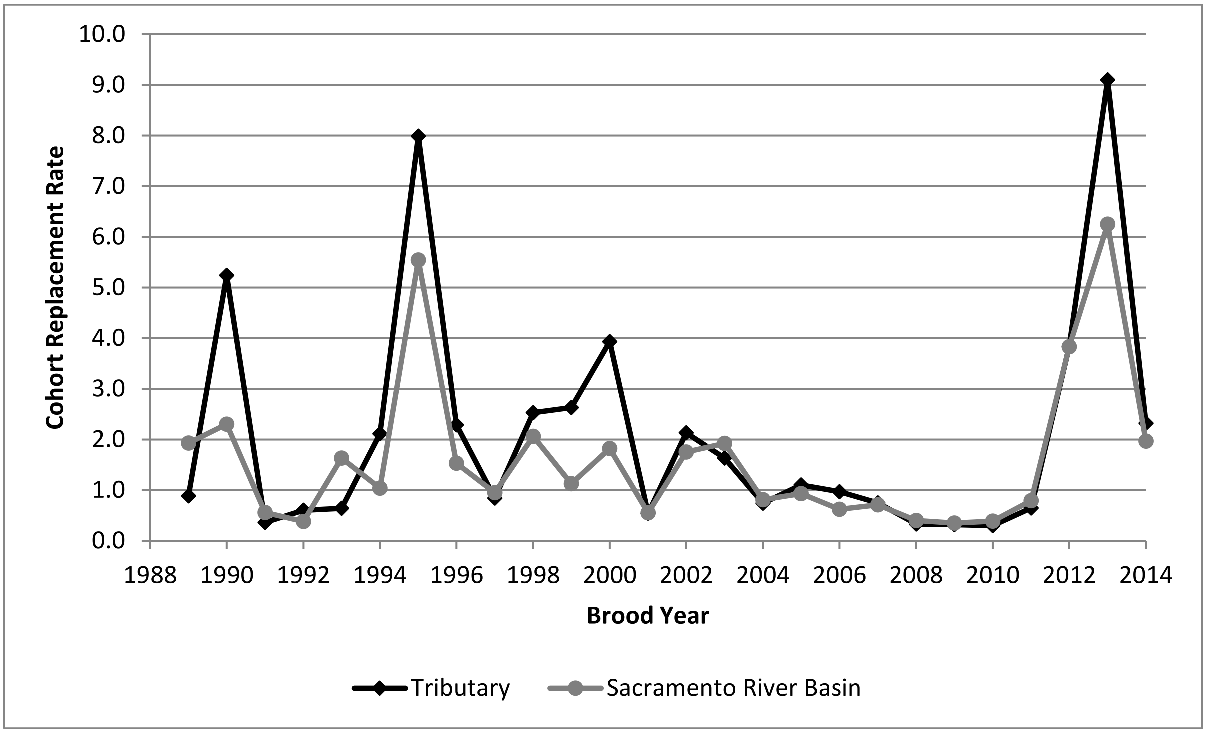

Figure 4.A.2-10. Cohort Replacement Rate for Tributary and Sacramento River Basin Populations, 1989–2014

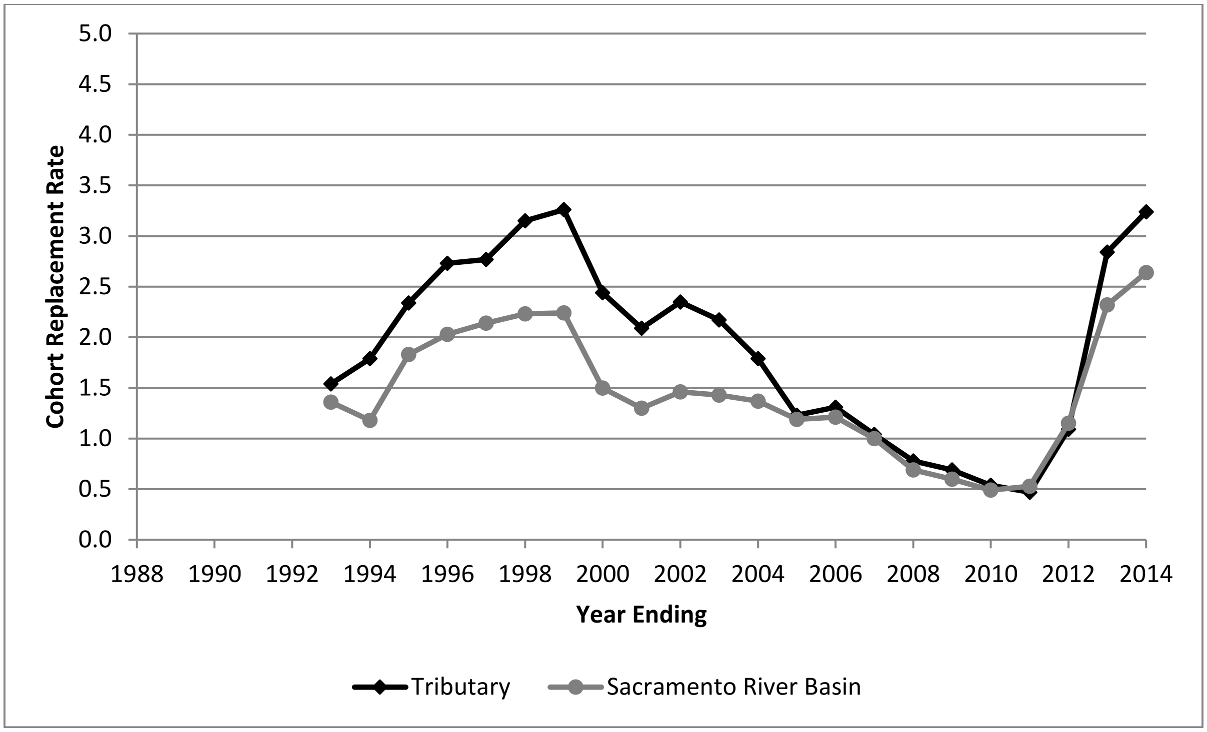

Figure 4.A.2-11. 5-Year Average Cohort Replacement Rate for Tributary and Sacramento River Basin Populations, 1989–2014

4.A.2.6.3 Spatial Structure

NMFS (2011) estimates that historically there were 18 or 19 independent populations, and up to eight smaller dependent populations, of spring-run Chinook salmon distributed among four diversity groups (distinct geographic regions) in the Central Valley. The four spring-run Chinook salmon diversity groups include (Figure 4.A.2-12):

- The basalt and porous lava diversity group composed of the upper Sacramento River, McCloud River, Pit River and Battle Creek watersheds;
- The northwestern California diversity group composed of streams that enter the mainstem Sacramento River from the northwest, such as Clear Creek;
- The northern Sierra Nevada diversity group composed of streams tributary to the Sacramento River from the east, and including the Mokelumne River; and
- The southern Sierra Nevada diversity group composed of streams tributary to the San Joaquin River from the east.

Of the 18 to 19 independent populations occurring within these four diversity groups, only three independent populations (Butte, Deer, and Mill Creeks) comprising one diversity group (Northern Sierra Nevada) remain. Collectively, the populations in Butte, Deer, and Mill Creeks have fluctuated broadly, but in recent years are showing a positive trend in abundance (Figure 4.A.2-8). In addition to these populations in the Northern Sierra Nevada diversity group, small populations also remain in Antelope and Big Chico Creeks and larger populations in the Feather and Yuba Rivers.

Historic spring-run Chinook salmon populations in the Basalt and Porous Lava and Southern Sierra Nevada diversity groups were extirpated, although Battle Creek has had a small but recently increasing population since 1995 (Figure 4.A.2-125). Historically, the Northwestern California diversity group contained only several dependent populations, but it currently contains a small but consistent population in Clear Creek and a small population in Beegum Creek (a tributary to Cottonwood Creek) (Table Figure 4.A.2-12).

Efforts are presently underway to restore a population in the San Joaquin River (Southern Sierra Nevada group), as part of the San Joaquin River Restoration Program (NMFS 2014). While the construction of dams is believed to have extirpated spring-run Chinook salmon from the Southern Sierra Nevada diversity group, there is some evidence to suggest that small numbers (<50) of spring-run Chinook salmon may opportunistically enter the Stanislaus and Tuolumne Rivers in some years (Franks 2015).

To meet the objective of “representation and redundancy” of spatial structure as described by Lindley et al. (2007), diversity groups need to contain multiple populations to survive. With only one of the four historical diversity groups containing viable independent populations (i.e., the Northern Sierra Nevada diversity group), the current spatial structure of CV spring-run Chinook salmon is severely reduced. To achieve diversity group recovery, the Central Valley Salmon and Steelhead Recovery Plan (NMFS 2014) proposes the following ESU-level recovery criteria:

- One population in the Northwestern California diversity group.
- Two populations in the Basalt and Porous Lava diversity group.
- Four populations in the Northern Sierra Nevada diversity group.
- Two populations in the Southern Sierra Nevada diversity group.

The existing populations on Clear Creek and Battle Creek, along with the completed and proposed habitat restoration projects, are anticipated to add to the spatial structure of the CV spring-run Chinook salmon ESU if these populations can reach viable status in their respective diversity group areas. The proposed plans to re-establish a spring-run Chinook salmon population in the San Joaquin River downstream of Friant Dam as part of the San Joaquin River Restoration Program will similarly add to the spatial structure of the CV spring-run Chinook salmon ESU; however, the San Joaquin River Restoration Program’s future long-term contribution to the CV spring-run Chinook salmon ESU is uncertain (NMFS 2014). In addition to restoring currently accessible watersheds, the final Central Valley Salmon and Steelhead Recovery Plan (NMFS 2014) recommends reestablishing populations into historical habitats currently blocked by large dams, such as Shasta Dam on the Sacramento River and Englebright Dam on the Yuba River.

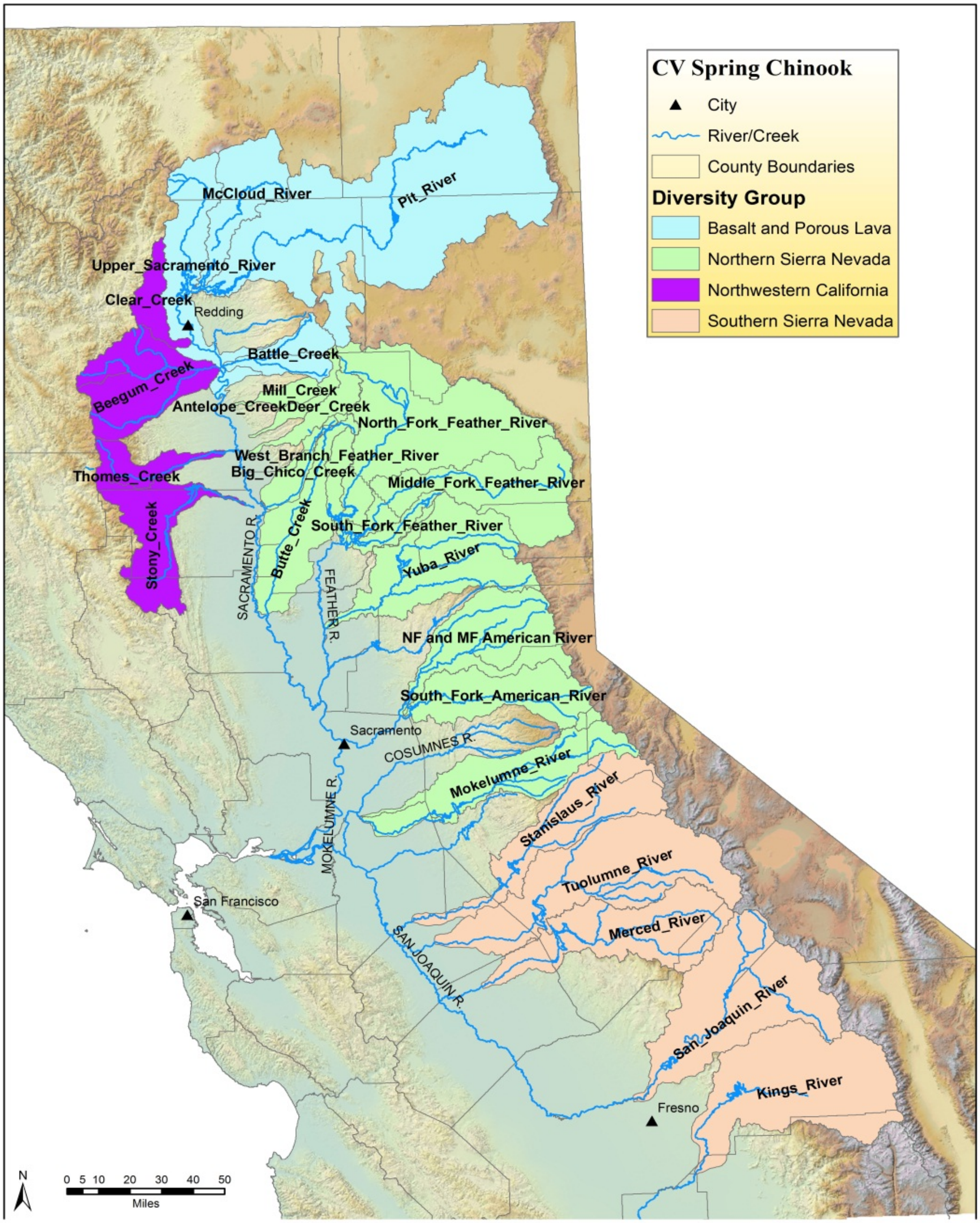

Figure 4.A.2-12. Diversity Groups for the Central Valley Spring-run Chinook Salmon ESU

4.A.2.6.4 Diversity

The genetic integrity of CV spring-run Chinook salmon has been compromised. Construction of dams has completely blocked access to primary spawning and rearing habitats, forcing spring-run Chinook salmon to spawn in the same areas as fall-run Chinook salmon. Exceptions occur in Butte, Deer, and Mill Creeks where CV spring-run Chinook salmon adults can access much of their historical spawning and rearing habitat. Small populations in Clear Creek and Battle Creek are separated by a segregation weir on Clear Creek and fall-run Chinook salmon are segregated from spring-run Chinook salmon on Battle Creek by the Coleman Hatchery weir. Presently, the CV spring-run Chinook salmon ESU is comprised of two known genetic complexes: all natural spawning populations of spring-run Chinook salmon in the Sacramento River and its tributaries, i.e., populations in Butte, Deer, and Mill Creeks, which have retained genetic integrity; and the Feather River population, in which the genetic integrity has been compromised as a result of introgression with fall-run Chinook salmon (Baerwald et al. 2011). The Feather River spring-run Chinook salmon have introgressed with the Feather River fall-run Chinook salmon, and Feather River Hatchery spring-run Chinook salmon may have impacted Yuba River spring-run Chinook salmon and likely have introgressed with Yuba River fall-run Chinook salmon (see discussion that follows). Finally, the apparent extirpation of the San Joaquin River basin spring-run Chinook salmon has further reduced the genetic diversity of the CV spring-run Chinook salmon ESU.

Although the Feather River Hatchery is presently operated to minimize the introgression of spring-run Chinook salmon with fall-run Chinook salmon, interbreeding of Feather River Hatchery spring-run fish with wild spring-run fish in other basins has been a concern (Joint Hatchery Review Committee 2001a). The practice of releasing Feather River Hatchery spring-run Chinook in San Pablo Bay to improve survival and reduce competition and predation impacts in-river increases the incidence of straying likely because of poor imprinting to their home hatchery waters (see Joint Hatchery Review Committee 2001b). As discussed in the public draft of the Hatchery and Stocking Program Environmental Impact Report/Environmental Impact Statement (ICF Jones & Stokes 2009), of the spring-run Chinook released in the Feather River, tag recoveries suggested that the great majority of spawners (over 98%) returned to the Feather River (Table 4.A.2-7) and 47% of these fish were recovered at the hatchery. A small proportion of spawners originating from on-site releases were also recovered in Battle Creek (0.02%) and the Yuba River (1.7%). Spring-run Chinook that were released from San Francisco Bay (including San Pablo Bay) strayed to a greater extent. According to tag recoveries, about 85% of those that survived at sea returned to the Feather River and 30% of these fish (26% overall) were recovered at the hatchery. Other recovery locations for off-site releases of Feather River Hatchery spring-run Chinook included the Yuba River (8%) and the Sacramento River (6%). Results from the 2010 and 2011 analysis of the proportion of Feather River hatchery-origin CV spring-run Chinook have shown that the stray rate for both net pen acclimated and Feather River release types have been similar (<2%) throughout the Central Valley (Kormos et. al. 2012, Palmer-Zwahlen and Kormos 2013). In addition, the straying into Butte Creek during the same period was found to be less than 0.9% and 0%. This is not to downplay the importance of straying from the Feather River hatchery as 1 or 2% of the total production could be a large amount when a stream has a small escapement number.

Table 4.A.2-7. Estimated Percentages of Feather River Hatchery Spring-Run Chinook Salmon Returning to Various Central Valley Streams (1987-2007).[a]

	Release Location	
Recovery Location	Feather River (%)	San Francisco Bay (%)
Feather River	98 (46)	85 (26)
American River	0	0.2
Battle Creek	0.02	0.8
Butte Creek	0	0.03
Merced River	0	0.03
Mokelumne River	0	0.07
Sacramento River	0	6
Tuolumne River	0	0.01
Yuba River	2	8

[a] Based on coded-wire tag recovery data from the Regional Mark Information System Database. Also shown is the percent tags recovered at the Feather River Hatchery (in parenthesis).

Other wild populations in the ESU potentially affected by Feather River Hatchery strays include Deer, Mill, Clear, and Antelope Creeks. Table 4.A.2-8 summarizes coded-wire tag data collected in these streams since 1988.

Table 4.A.2-8. Summary of Coded-Wire Tags from Spring-Run Chinook Salmon Collected in Mill, Deer, Clear and Antelope Creeks, 1988–2008.

Stream	Period of Record	No. Survey Years	No. Years Tagged Fish Observed	Percent of Fish with Tag[a]	Origins of Tagged Fish[b]
Mill Creek	1989–2008	18	0	0	
Deer Creek	1992–2008	12	0	0	
Clear Creek	2003–2014	12	7	3.5 (0.02–0.04)	Feather River Hatchery, Butte Creek (wild)
Antelope Creek	1993–2008	8	0	0	

[a] Average and range (in parentheses) of annual number of ad-clipped or coded-wire tagged fish observed as a percentage of the total number of fish examined.

[b] Dominant hatchery sources.

Source: Regional Mark Information System Database [online database].

Hatchery spring-run Chinook salmon were detected in Clear Creek in half the years that surveys were conducted. Four of the 160 carcasses examined (0.03%) in 2003, 2004, and 2008 had an adipose-clip or coded-wire tag (CWT). Coded-wire tags were detected in two of these fish, one of which originated from Butte Creek (wild) and the other from Feather River Hatchery (San Pablo Bay release). No tagged spring-run Chinook salmon have been observed in Mill, Deer, and Antelope Creek. Subject to the caveat that sampling effort was low, the total lack of observations of tagged spring-run Chinook in Mill, Deer, and Antelope Creek over 8 to 18 years of surveys suggests that the degree of hatchery influence on these populations is negligible.

4.A.2.6.5 ESU Viability

Given that CV spring-run Chinook salmon populations in Butte, Deer, and Mill Creeks represent the only historic populations in the ESU, these populations also represent the best long-term trend indicators for ESU viability. Lindley et al. (2007) concluded that Butte and Deer Creek spring-run Chinook salmon populations were at a low risk of extinction, based on population viability analysis (PVA) model results and other population viability criteria (i.e., population size, population decline, catastrophic events, and hatchery influence). However, Lindley et al. (2007) also concluded that based on the PVA model, the Mill Creek spring-run Chinook salmon population was at a moderate risk of extinction, while satisfying other viability criteria for low-risk status.

However, the CV spring-run Chinook salmon ESU fails to meet the "representation and redundancy rule" because all three existing populations occur within only one of the three diversity groups that historically contained multiple independent populations of CV spring-run Chinook salmon. Currently, there are only three independent populations of CV spring-run Chinook salmon and they all exist within close proximity to one another in the Northern Sierra Nevada diversity group, which puts them all at risk of being eliminated as a result of a single large catastrophic event (e.g., volcanic eruptions from Mount Lassen, large forest fires in the headwaters, and drought).

In the most recent (2011) 5-year status review of CV spring-run Chinook salmon, NMFS concluded that the ESU should remain classified as a threatened species; however, NMFS concluded the biological status of the ESU had worsened since the 2005 status review, and NMFS suggested that the Deer and Mill Creek populations could be moving towards a high risk of extinction (NMFS 2011). The recent increasing trend in adult abundance in the Butte, Deer, and Mill Creek may indicate a reversal or lessening of this trend. The increasing trend in abundance of spring-run Chinook salmon in Clear Creek and Battle Creek has placed these populations at a moderate extinction risk. Existing and planned restoration actions, particularly on Battle Creek is expected to assist in reducing the extinction risk, if these populations respond positively to these actions. Long-term recovery of the CV spring-run Chinook salmon ESU will require improved freshwater habitat conditions, reduced harvest impacts, abatement of threats throughout the entire ESU, and the establishment of populations in other tributaries or potentially upstream of Shasta Dam on the Sacramento River and Englebright Dam on the Yuba River, and in the San Joaquin River basin.

4.A.2.7 Relevant Conservation Efforts

Conservation actions initially put in place because of identified problems for winter-run Chinook salmon have most likely also benefitted spring-run Chinook salmon. These habitat and harvest related problems have been addressed and improved through restoration and conservation actions. The impetus that drove these actions stems primarily from the following actions.

- ESA Section 7 consultation Reasonable and Prudent Alternative Actions that address water operations related management of water temperature, flow, and operations of the CVP/SWP (NMFS 2009b).

- Regional Water Quality Control Board decisions requiring compliance with Sacramento River water temperature objectives, which resulted in the installation of the Shasta Temperature Control Device in 1998.

- A 1992 amendment to the authority of the CVP through the Central Valley Improvement Act to give fish and wildlife equal priority with other CVP objectives.

- Fiscal support of habitat improvement projects from the CALFED Bay-Delta Program (CALFED) (e.g., installation of a fish screen on the Glenn-Colusa Irrigation District diversion, Battle Creek Restoration Project).

- EPA actions to control acid mine runoff from Iron Mountain Mine.

- Ocean harvest restrictions implemented in 1995 and salmon season closures in 2007 and 2008.

Results of monitoring at the CVP/SWP and extensive experimentation over the past several decades have led to the identification of a number of m actions designed to reduce or avoid the potentially adverse effects of CVP/SWP export operations on salmon. Key to these actions have been State Water Board water rights decisions (D-1485, D-1641), BiOps issued on project export operations by NMFS and USFWS, CALFED programs (e.g., Environmental Water Account), and Central Valley Project Improvement Act actions. These requirements support multiple conservation efforts to enhance habitat and reduce entrainment of Chinook salmon by the CVP/SWP export facilities.

BiOps for CVP/SWP operations (e.g., NMFS 2009a) and other federal projects involving irrigation, water diversion, and fish passage have improved adverse effects on salmon in the Central Valley. In 1992, an amendment to the authority of the CVP through the Central Valley Project Improvement Act was enacted to give protection of fish and wildlife equal priority with other CVP objectives. From this act arose several programs that have benefited listed salmonids.

- The Anadromous Fish Restoration Program is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the program include fish passage, fish screening, riparian easement, and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment.

- The Anadromous Fish Screen Program 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 Water Acquisition Program is to acquire water supplies to meet the habitat restoration and enhancement goals of the Central Valley Project Improvement Act, and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for Central Valley salmon, with the primary focus on listed Chinook salmon and steelhead, by maintaining or increasing instream flows on the Sacramento River at critical times, and to reducing salmonid entrainment at the CVP/SWP export facilities through reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses. Two programs included under CALFED, the Ecosystem Restoration Program and the Environmental Water Account, were created to improve conditions for fish, including spring-run Chinook salmon, in the Central Valley. The Ecosystem Restoration Program Implementing Agency Managers selected a proposal for directed action funding written by the Central Valley Salmonid Project Work Team, an interagency technical working group led by CDFW, to develop a spring-run Chinook salmon escapement-monitoring plan. Long-term funding for implementation of the monitoring plan must still be secured.

A major restoration action currently under way is the Battle Creek Salmon and Steelhead Restoration Project, which is modifying facilities at Battle Creek Hydroelectric Project diversion dam sites located on the North and South Forks of Battle Creek and Baldwin Creek. The project will restore 48 miles (77 kilometers) of habitat in Battle Creek to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $100 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. It is thought that this restoration effort is the largest coldwater restoration project to date in North America.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Program elements in the Delta. The DRERIP team has created a suite of ecosystem and species conceptual models, including for spring-run Chinook salmon (Williams 2010), that document existing scientific knowledge of Delta ecosystems. The DRERIP team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Program for implementation. DRERIP conceptual models were used in the analysis of proposed conservation measures.

Recent habitat restoration initiatives sponsored and funded primarily by the Ecosystem Restoration Program have resulted in plans to restore ecological function to 9,543 acres of shallow-water tidal and marsh habitats in 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 adjacent to Suisun Marsh (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 Estuary in conjunction with tidal wetland restoration.

The EPA's Iron Mountain Mine remediation involves the removal of toxic metals in acidic mine drainage from the Spring Creek Watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s. 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 to dilute heavy metal contaminants 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.

To eliminate an impediment to migration of adult and juvenile spring-run Chinook salmon and other species, operation of the Red Bluff Diversion Dam ceased in 2011 and dam gates were placed in a permanent open position. A new pumping facility was built that includes a state-of-the-art fish screen.

Since 1986, DWR's Delta Fish Agreement Program has approved approximately $49 million for projects that benefit salmon and steelhead production in the Sacramento–San Joaquin basins and Delta. The Delta Fish Agreement projects that benefit CV spring-run Chinook salmon include water exchange programs on Mill and Deer Creeks; enhanced law enforcement from San Francisco Estuary upstream to the Sacramento and San Joaquin Rivers and their tributaries; design and construction of fish screens and ladders on Butte Creek; and screening of diversions in Suisun Marsh and San Joaquin River tributaries. The Spring-Run Salmon Increased Protection Project provides overtime wages for CDFW wardens to focus on reducing illegal take and illegal water diversions on upper Sacramento River tributaries and adult holding areas, where the fish are vulnerable to poaching. This project covers Mill, Deer, Antelope, Butte, Big Chico, Cottonwood, and Battle Creeks, and has been in effect since 1996. Through the Delta-Bay Enhanced Enforcement Program, initiated in 1994, ten wardens focus their enforcement efforts on salmon, steelhead, and other species of concern from the San Francisco Estuary upstream into the Sacramento and San Joaquin River basins. These two enhanced enforcement programs have likely had significant benefits to spring-run Chinook salmon attributed to CDFW, although results have not been quantified.

The Mill and Deer Creek Water Exchange projects will provide new wells that enable diverters to bank groundwater in place of stream flow, thus leaving water in the stream during critical migration and oversummering periods. On Mill Creek, several agreements between Los Molinos Mutual Water Company, Orange Cove Irrigation District, CDFW, and DWR allow DWR to pump groundwater from two wells into the Los Molinos Mutual Water Company canals to pay back Los Molinos Mutual Water Company water rights for surface water released downstream for fish. Although the Mill Creek Water Exchange project was initiated in 1990 and the agreement allows for a well capacity of 25 cubic feet per second (cfs), only 12 cfs has been developed to date. In addition, it has been determined that a base flow of greater than 25 cfs is needed from April through June for upstream passage of adult spring-run Chinook salmon in Mill Creek. In some years, water diversions from the creek are curtailed by amounts sufficient to provide for passage of upstream migrating adult spring-run Chinook salmon and downstream migrating juvenile steelhead and spring-run Chinook salmon.

The Feather River Hatchery is making efforts to segregate spring-run from fall-run Chinook salmon to enhance and restore the genotype of spring-run Chinook salmon in the Feather River (DFG 2001; McReynolds et al. 2006).

Seltzer Dam on Lower Clear Creek was removed in 2000, thereby opening up approximately 10 miles of stream habitat to anadromous salmonids including CV spring-run Chinook salmon. Since this dam removal, there has been extensive gravel augmentation and regulation of instream flows and water temperatures both as part of the Clear Creek Restoration Program and as required by NMFS' CVP-OCAP BiOp. This program has been successful in restoring Clear Creek habitat conditions such that the watershed now supports a small but increasing population of spring-run Chinook.

Recent conservation actions have improved habitat conditions for Butte Creek spring-run Chinook salmon. Completion of the Willow Slough Weir Project (new culverts and a new fish ladder) in 2010 improved fish passage through the Sutter Bypass. In addition, since 2000, real-time coordinated operations of the DeSabla Centerville Project (FERC Project No. 803) have been implemented to reduce the water temperature-related effects of the project on spring-run Chinook salmon adults during the summer.

The U.S. Army Corps of Engineers initiated a long-term gravel augmentation program in 2010 that is intended to improve spawning habitat in the uppermost reach of the lower Yuba River. Other lower Yuba River habitat restoration actions that are reasonably certain to occur in the next several years include improved fish passage at Daguerre Point Dam (known to have passage problems at high flows), a long-term program to add woody material to the river in an effort to increase habitat complexity, and a riparian enhancement project intended to improve rearing habitat in the short- and long-term. In addition, the FERC re-licensing process for the Yuba River Project is likely to include monitoring studies of spring-run Chinook in the Lower Yuba River over the next five years.

The San Joaquin River Restoration Program (SJRRP) calls for a combination of channel and structural modifications along the San Joaquin River below Friant Dam, releases of water from Friant Dam to the confluence of the Merced River, and the reintroduction of spring-run Chinook salmon. The first flow releases from Friant Dam in support of the SJRRP occurred in October 2009, and juvenile spring-run Chinook salmon were released below Friant Dam in 2013 and 2014.

To help reduce the effects of the Red Bluff Diversion Dam operation on migration of adult and juvenile salmonids and other species, the dam gates are now maintained in a permanent open position, thereby facilitating greater upstream and downstream migration. Changes in dam operations have benefited both upstream and downstream migration by salmon and have contributed to a reduction in juvenile predation mortality. In 2009, Reclamation received funding for the Fish Passage Improvement Project at the Red Bluff Diversion Dam to build a pumping facility to provide reliable water supply for high-valued crops in Tehama, Glenn, Colusa, and northern Yolo Counties while providing year-round unimpeded fish passage. This project was completed in 2012 and is expected to eliminate passage issues for spring-run Chinook salmon and other migratory species.

Seasonal constraints on sport and commercial fisheries south of Point Arena benefit spring-run Chinook salmon. CDFW has implemented enhanced enforcement efforts to reduce illegal harvests. CV spring-run Chinook salmon is a state-listed fish that is protected by specific in-river fishing regulations.

4.A.2.8 Recovery Goals

The recovery plan for Central Valley salmonids, including CV spring-run Chinook salmon, was released by NMFS on July 22, 2014. The overarching goal is the removal of, among other listed salmonids, CV spring-run Chinook salmon from the federal list of endangered and threatened wildlife (NMFS 2014). Recovery goals usually can be subdivided into discrete component objectives that, collectively, describe the conditions (criteria) necessary for achieving the goal.

Recovery objectives are the parameters of the goal, and criteria are the values for those parameters. For the ESU to achieve recovery, each of the Diversity Groups should support both viable and dependent populations and meet goals for redundancy and distribution. More specifically, to achieve recovery the CV spring-run Chinook ESU should display the following characteristics:

- One population in the Northwestern California Diversity Group at low risk of extinction
- Two populations in the Basalt and Porous Lava Diversity Group at low risk of extinction
- Four populations in the Northern Sierra Diversity Group at low risk of extinction
- Two populations in the Southern Sierra Diversity Group at low risk of extinction
- Maintain multiple populations at moderate risk of extinction

Criteria for low risk of extinction include a census population size that is >2,500 adults, or has an effective population size that is >500, no productivity decline that is apparent, no catastrophic event that has occurred within the last 10 years, and hatchery influence is at low levels. Criteria for moderate extinction risk include: a census population that is 250 to 2,500 adults, or has an effective population that is 50 to 250 adults, run sizes are <500, but are stable, no apparent decline in populations growth rate that stems from a catastrophic event that has happened in the last 10 years, and hatchery influence is moderate.

4.A.2.9 References

4.A.2.9.1 Written References

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4.A.3 Steelhead, California Central Valley (*Oncorhynchus mykiss*)

4.A.3.1 Introduction

This section provides information on the basic biology, life history, status, and threats and stressors of California Central Valley (CCV) steelhead in the action area.

4.A.3.2 Status

The CCV steelhead evolutionarily significant unit (ESU) was listed as a threatened species under the federal Endangered Species Act (ESA) on March 19, 1998 (63 *Federal Register* [FR] 13347). On November 4, 2005, the National Marine Fisheries Service (NMFS) proposed that all west coast steelhead be reclassified from ESUs to Distinct Population Segments (DPSs) and proposed to retain CCV steelhead as threatened (70 FR 6130) (Figure 4.A.3-1). On January 5, 2006, after reviewing the best available scientific and commercial information in a status review (Good et al. 2005), National Marine Fisheries Service (NMFS) issued its final rule to retain the status of CCV steelhead as threatened and applied its hatchery listing policy to include the Coleman National Fish Hatchery and Feather River Hatchery steelhead programs as part of the DPS (71 FR 834).

In its latest 5-year status review, NMFS determined that the CCV steelhead DPS should remain classified as threatened (NMFS 2011). However, based on new information, NMFS determined that the status of the DPS was worse than the previous review (Good et al. 2005), and the extinction risk of the DPS increased.

4.A.3.3 Critical Habitat

Critical habitat for the CCV steelhead DPS was designated by NMFS on September 2, 2005 (70 FR 52488), and includes 2,308 miles of stream habitat in the Central Valley and an additional 254 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex (Figure 4.A.3-2). Critical habitat for CCV steelhead includes stream reaches such as those of the Sacramento, Feather, and Yuba Rivers; Deer, Mill, Battle, and Antelope Creeks in the Sacramento River basin; the San Joaquin River and its tributaries; and the Delta. Critical habitat includes stream channels in the designated stream reaches and the lateral extent as defined by the ordinary high-water line. In areas where the ordinary high-water line has not been defined, the lateral extent of critical habitat is defined by the bank-full elevation (the level at which water begins to leave the channel and move into the floodplain and is reached at a discharge that generally has a recurrence interval of 1 to 2 years on the annual flood series) (70 FR 52488).

Within these areas, the PBFs essential for the conservation of the CCV steelhead DPS are those sites and habitat components that support one or more life stages, including:

1. Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development;
2. Freshwater rearing sites with:
    a. (i) Water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility;

   b. (ii) Water quality and forage supporting juvenile development; and

   c. (iii) Natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

3. Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.

4. Estuarine areas free of obstruction and excessive predation with:

   a. Water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater;

   b. Natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and

   c. Juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

4.A.3.3.1 Spawning Habitat

CCV steelhead are limited to spawning downstream of dams on nearly every major tributary within the Sacramento and San Joaquin River systems. Freshwater spawning sites are those with water quantity and quality conditions and substrate supporting spawning, egg incubation, and larval development. These would include sites with coarse gravel having good inter-gravel flow usually at the tail of a pool or in a riffle. Water velocities over redds are generally 20 to 155 cm/sec, and the depths are 10 to 155 cm (Moyle 2002). Optimal temperatures for steelhead spawning are reported to be 39 degrees Fahrenheit [°F] to 52°F (McEwan and Jackson 1996). Spawning habitat for CCV steelhead primarily occurs in mid to upper elevation reaches or immediately downstream of dams located throughout the Central Valley that contain suitable environmental conditions (e.g., seasonal water temperatures, substrate, and dissolved oxygen) for spawning and egg incubation.

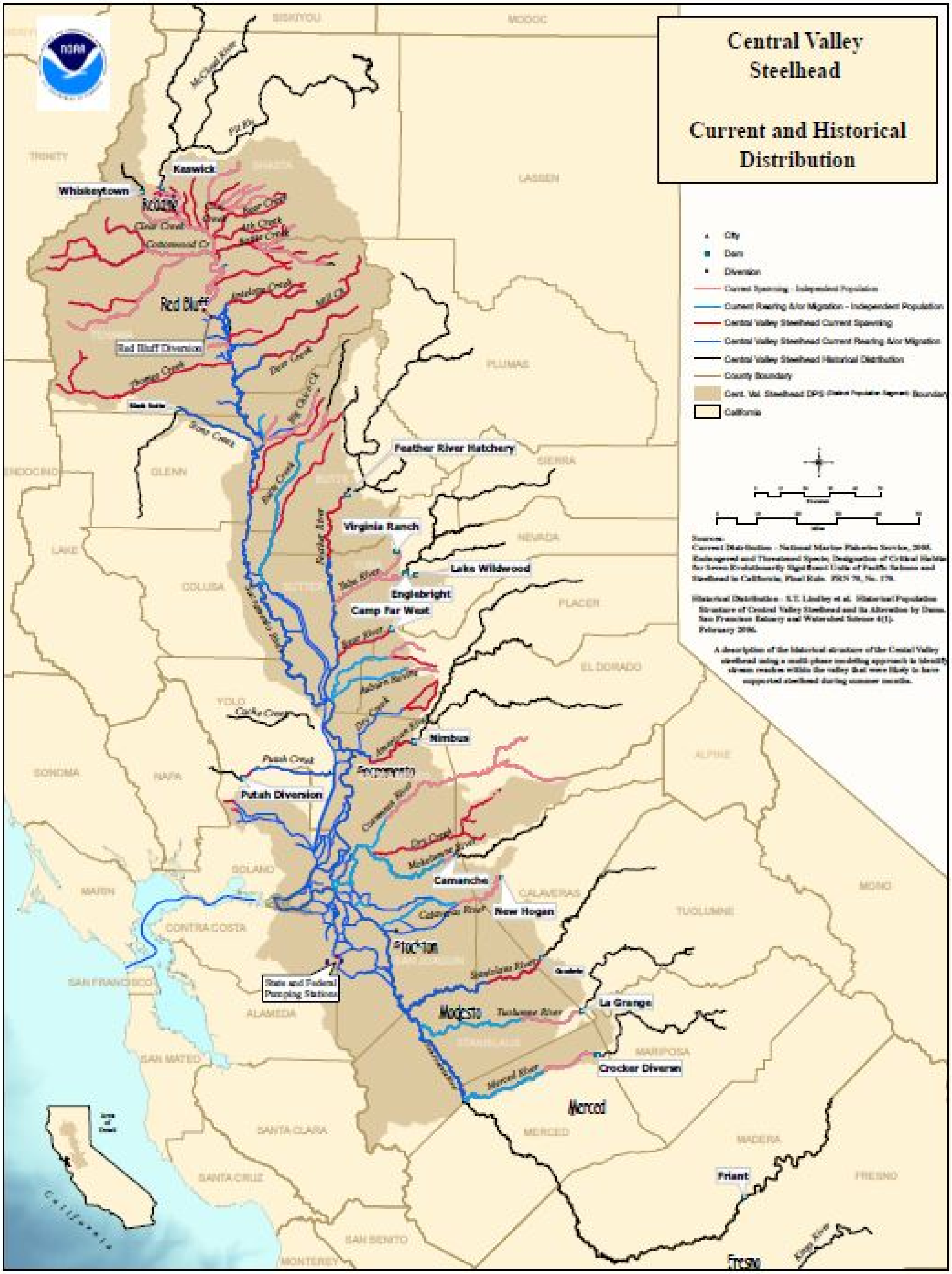


Figure 4.A.3-1. CCV steelhead Distinct Population Segment Boundary, and Current and Historical Distribution (Source: NMFS 2014)

Figure 4.A.3-2. CCV steelhead Designated Critical Habitat and Distribution (http://www.westcoast.fisheries.noaa.gov/publications/gis_maps/maps/salmon_steelhead/esa/steelhead/ccv_steelhead.pdf

4.A.3.3.2 Freshwater Rearing Habitat

Freshwater steelhead rearing sites contain suitable water quantity and floodplain connectivity to form and maintain physical habitat conditions that support juvenile growth and mobility, water quality (e.g., water temperatures) and provide forage supporting juvenile development, and include natural cover such as shade, submerged and overhanging large wood, log jams, beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks. Spawning areas and migratory corridors may also function as rearing habitat for juveniles, which feed and grow before and during their out-migration. Rearing habitat value is strongly affected by habitat complexity, food supply, and the presence of predators. The channeled, leveed, and riprapped river reaches and sloughs common in the lower Sacramento and San Joaquin Rivers and throughout the Delta, however, typically have low habitat complexity and low abundance of food organisms, and offer little protection from predation by fish and birds. Freshwater rearing habitat has a high conservation value because juvenile steelhead are dependent on the function of this habitat for successful survival and recruitment to the adult population.

4.A.3.3.3 Freshwater Migration Corridors

Freshwater migration corridors should be free from obstructions (passage barriers and impediments to migration) and excessive predation with water quantity (instream flows) and quality conditions (seasonal water temperatures) and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival. Optimal freshwater steelhead migration corridors (including river channels, channels through the Delta, and the Bay-Delta estuary) support mobility, survival, and food supply for juveniles and adults. Migratory corridors are typically downstream of the spawning area and include the lower Sacramento and San Joaquin Rivers, the Delta, and the San Francisco Bay complex extending to coastal marine waters. These corridors allow the upstream passage of adults and the downstream emigration of juvenile steelhead and of kelts. Migratory corridor conditions are strongly affected by the presence of passage barriers, which can include dams, unscreened or poorly screened diversions, and degraded water quality. For freshwater migration corridors to function properly, they must provide adequate passage, provide suitable migration cues, reduce false attraction, avoid areas where vulnerability to predation is increased, and avoid impediments and delays in both upstream and downstream migration. Juvenile CCV steelhead that emigrate from the San Joaquin River tributaries are exposed to very degraded migration corridors with low habitat value, high temperatures and degraded water quality (Reclamation 2011). Substantial amounts of flow and significant numbers of juvenile CCV steelhead from the Sacramento River enter the Delta Cross Channel (when in operation) and Georgiana Slough into the central Delta (Singer et al. 2013). Similarly, juvenile CCV steelhead from the San Joaquin River tributaries enter into the Old River, Turner, and Columbia Cuts. Juvenile CCV steelhead entering into the central Delta can suffer higher mortality rates than those traveling down the main portions of the Sacramento River and San Joaquin River (Delaney et al. 2014). Higher mortality rates are thought to stem from longer migration, with higher temperatures, higher predation rates, low water quality, and higher exposure to contaminants (Reclamation 2011). Entrainment at the State and Federal facilities causes mortality, but recent acoustic telemetry studies demonstrate that, once fish reach the proximity of the export facilities, salvage at the CVP can provide higher survival to the western Delta than volitional migration; even during positive OMR conditions (SJRG 2011; SJRG 2013).

4.A.3.3.4 Estuarine Habitat

Ideal estuarine areas are free of migratory obstructions and excessive predation with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and salt water; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

A portion of steelhead smolts swim through the Delta Cross Channel and Georgiana Slough into the interior Delta where they are, based upon Chinook studies, expected to be subjected to lower survival (Newman and Brandes 2010) and can be subjected to both the Federal and State fish facilities (Singer et al. 2013). Delta hydraulics has been modified as a result of CVP/SWP actions. Within the central and southern Delta, net water movement is towards the pumping facilities, though recent independent science reviews have concluded that instantaneous velocities and olfaction are most important for juvenile salmon navigation (Anderson et al. 2012; Monismith et al. 2014).Operations of upstream reservoir releases and diversion of water from the southern Delta have been manipulated to maintain a “static” salinity profile in the western Delta near Chipps Island (the X2 location). This area of salinity transition, the low salinity zone (LSZ), is an area of high productivity. Historically, this zone fluctuated in its location in relation to the outflow of water from the Delta and moved westwards with high Delta inflow (i.e., floods and spring runoff) and eastwards with reduced summer and fall flows. This variability in the salinity transition zone has been substantially reduced by the operations of the CVP/SWP projects.

The current condition of the estuarine habitat has been substantially degraded from historic conditions (Cloern and Jassby 2012). Over 90% of the fringing fresh, brackish, and salt marshes have been lost to anthropogenic uses (Nichols et al. 1986). This loss of the fringing marshes reduces the availability of forage species and eliminates the cycling of nutrients from the marsh vegetation into the water column of the adjoining waterways (Cloern 2007). The channels of the Delta have been deepened and the levees raised and armored with stone riprap. This simplifies the habitat by reducing the incorporation of woody debris and vegetative material into the nearshore area. It minimizes habitat complexity by reducing local variations in water depth and velocities, and simplifies the community structure of the nearshore environment (Moyle et al. 2010; Mount et al. 2012). Upstream reservoir releases and diversion of water from the southern Delta have been manipulated to maintain a “static” salinity profile in the western Delta near Chipps Island (Moyle et al. 2010). Heavy urbanization and industrial actions have lowered water quality and introduced persistent contaminants to the sediments surrounding points of discharge.

4.A.3.3.5 Marine Habitats

Most juvenile steelhead rear in coastal marine waters for a period of approximately 1 to 2 years before returning to Central Valley streams to spawn (Burgner et al. 1992). During their marine residence, steelhead forage on krill and other marine organisms. Offshore marine areas with water quality conditions and food, including squid, crustaceans, and fish (fish become a larger component in the steelhead diet later in life [Moyle 2002]) that support growth and maturation are important habitat elements, although marine habitats were not included as PBFs for CCV steelhead.

Results of oceanographic studies have shown variation in ocean productivity off the West Coast within and among years. Changes in ocean currents and upwelling have been identified as significant factors affecting nutrient availability, and phytoplankton and zooplankton production in near-shore surface waters. Although the effects of ocean conditions on steelhead growth and survival have not been investigated, recent observations have shown a significant decline in the abundance of adult Chinook and coho salmon returning to California rivers and streams. This decline has been hypothesized to be the result of declines in ocean productivity and associated high mortality rates during the period when these fish were rearing in near-shore coastal waters (MacFarlane et al. 2008). The importance of changes in ocean conditions on growth, survival, and population abundance of CCV steelhead, although potentially similar to that of Chinook salmon, is largely unknown (e.g., Peterson et al. 2012, Sharma et al. 2013).

4.A.3.4 Life History

Steelhead have two life history types: stream-maturing and ocean-maturing. Stream-maturing steelhead enter fresh water in a sexually immature condition and require several months to mature before spawning, whereas ocean-maturing steelhead enter fresh water with mature gonads and spawn shortly after river entry. A variation of the two forms occurs in the Central Valley and primarily migrates into the system in the fall, then holds in suitable habitat until spawning during the winter and early spring (McEwan and Jackson 1996). Peak immigration seems to have occurred historically in the fall from late September to late October (Hallock 1989), with peak spawning typically occurring January through March (Hallock et al. 1961; McEwan and Jackson 1996). Unlike Pacific salmon, steelhead are capable of spawning more than once before death (Busby et al. 1996). Most juvenile steelhead spend two years rearing, although some spending less and a very few spending more (Hallock et al. 1961). Central Valley steelhead typically spend two years in the ocean before returning to their natal stream to spawn.

4.A.3.4.1 Immigration and Holding

CCV steelhead generally leave the ocean and migrate upstream from August through March (Busby et al. 1996; Hallock et al. 1957; NMFS 2009a), and spawn from December through April (Newton and Stafford 2011; Reclamation 2008). Peak immigration seems to have occurred historically in the fall from late September to late October, with some creeks such as Mill Creek showing a small run in mid-February (Hallock 1989). Timing of upstream migration into tributaries suitable for spawning corresponds with higher flow events (e.g., freshets), associated lower water temperatures, and increased turbidity. The peak period of adult immigration into the Sacramento River appears to be during fall months with fewer immigrants in the winter (as reviewed in McEwan 2001). Holding behavior is probably similar to summer-run, where adults ascend into an area of cool, well oxygenated water, where they hold until they spawn.

4.A.3.4.2 Spawning

CCV steelhead generally spawn from December through April (Newton and Stafford 2011; Reclamation 2008). Peak spawning typically occurs from January through March in small streams and tributaries where cold, well-oxygenated water is available year-round (Table 4.A.3-1) (Hallock et al. 1961; McEwan and Jackson 1996). After reaching a suitable spawning area, the female steelhead selects a site with good intergravel flow, digs a redd, and deposits eggs while an attendant male fertilizes them. Eggs are covered with gravel dislodged just upstream. The length of time it takes for eggs to hatch varies in response to water temperature. Optimal spawning temperatures range between from 4°C and 11°C (39°F to 52°F), with egg mortality beginning at about 13°C (55°F) (McEwan and Jackson 1996). Hatching of steelhead eggs in hatcheries takes about 30 days at 10.6°C (51°F).

Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al. 1996). It is, however, rare for steelhead to spawn more than twice before dying; individuals that do spawn more than twice tend to be females (Busby et al. 1996). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al. 1996).

4.A.3.4.3 Egg to Parr

The length of time it takes for eggs to hatch varies in response to water temperature. Optimal spawning temperatures range between from 4 degrees Celsius [°C] and 11°C (39°F to 52°F), egg mortality begins at about 13°C (55°F) (McEwan and Jackson 1996). Hatchery steelhead eggs hatch in about 30 days at 10.6°C (51°F). Fry generally emerge from gravel 4 to 6 weeks after hatching, but factors such as redd depth, gravel size, siltation, and water temperature can speed or retard the time to emergence (Shapovalov and Taft 1954, as cited in McEwan and Jackson 1996). After hatching, alevins remain in the gravel for an additional two to five weeks while absorbing their yolk sacs, and emerge in spring or early summer (Barnhart 1986). Upon emergence, fry inhale air at the stream surface to fill their air bladders, absorption of the remaining portion of their yolk sac usually takes a few days, and they then start to feed actively, often in schools (Barnhart 1986; NMFS 1996).

Newly emerged juveniles move to shallow; protected areas with lower water velocities associated with the stream margin, and soon establish feeding locations in the juvenile rearing habitat (Shapovalov and Taft 1954, as cited in McEwan and Jackson 1996). As the parr increase in size and swimming ability, they begin to exhibit a preference for higher flow and deeper mid channel areas (Hartman 1965; Everest and Chapman 1972; Fontaine 1988).

Steelhead juvenile rearing during the summer takes place primarily in higher velocity areas in pools, although young-of-the-year (YOY) also are abundant in glides and riffles. Productive steelhead habitat is characterized by habitat complexity, primarily in the form of large and small woody debris and boulders. Cover is an important habitat component for juvenile steelhead both as velocity refugia and as a means of avoiding predation (Meehan and Bjornn 1991, as cited in McEwan and Jackson 1996). Optimal water temperatures for growth range from 15°C (59°F) to 20°C (68°F) (McCullough *et al.* 2001, Spina 2006). Cherry *et al.* (1975) found preferred temperatures for rainbow trout ranged from 11°C (51.8°F) to 21°C (69.8°F) depending on acclimation temperatures (cited in Myrick and Cech 2001).

4.A.3.4.4 Smolt Pre-smolt Migration

About 70% of CCV steelhead spend 2 years within their natal streams before migrating out of the Sacramento-San Joaquin system as smolts, with small percentages (29%) and (1%) spending 1 or 3 years, respectively (Hallock et al. 1961). Juvenile steelhead smolts emigrate primarily from natal streams in response to the first heavy runoff in the late winter through spring (Hallock et al. 1961). Emigrating CCV steelhead use the lower reaches of the Sacramento and San Joaquin Rivers and the Delta as a migration corridor to the ocean. Nobriga and Cadrett (2001) verified these temporal findings (spring migration) based on analysis of captures in U.S. Fish and Wildlife Service (USFWS) salmon monitoring conducted near Chipps Island.

4.A.3.4.5 Ocean Behavior

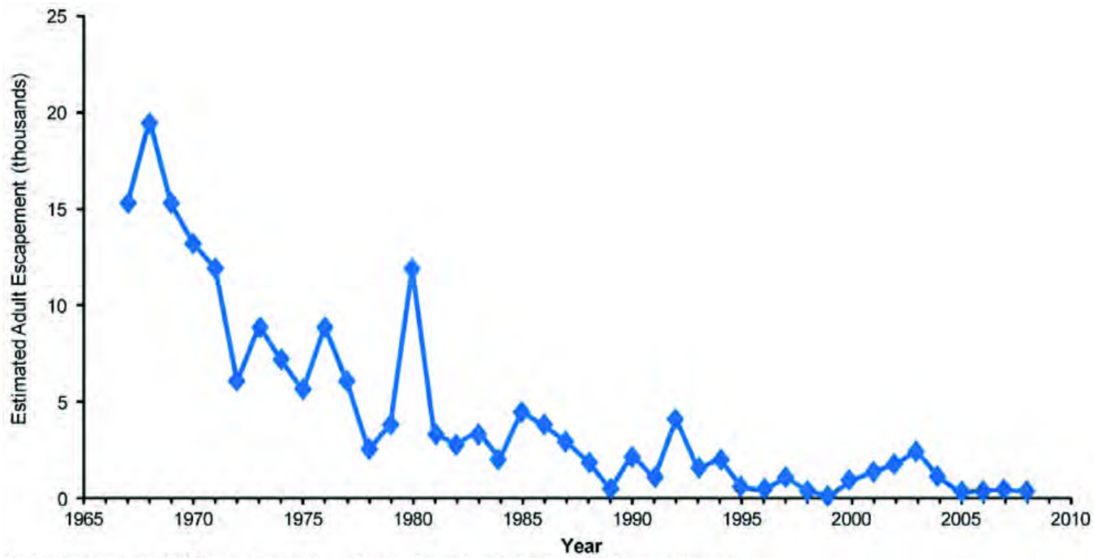
Most juvenile steelhead rear in coastal marine waters for a period of approximately 1 to 2 years before returning to Central Valley rivers as adults to spawn (Burgner et al. 1992). Unlike Pacific salmon, steelhead do not appear to form schools in the ocean (Behnke 1992). Burgner (1992) reported that no hatchery (coded wire tag [CWT]) steelhead from California were recovered from open ocean surveys from 1980–1988, with only a small number of disk-tagged fish being caught. Ocean migration and distribution of CCV steelhead stocks is unknown because of the paucity of data on ocean distribution. Steelhead experience most of their marine phase mortality soon after they enter the Pacific Ocean (Pearcy 1992). Ocean mortality is poorly understood, however, because few studies have been conducted to evaluate the importance of various factors, including predation mortality, changes in ocean currents, water temperatures, and coastal upwelling, on steelhead survival. Possible causes of ocean mortality include predation, competition, starvation, osmotic stress, unauthorized driftnet fisheries on the high seas, disease, advective losses, and other poor environmental conditions (Wooster 1983; Cooper and Johnson 1992; Pearcy 1992). Competition between steelhead and other species for limited food resources in the Pacific Ocean may be a contributing factor to declines in steelhead populations, particularly during years of low productivity (Cooper and Johnson 1992).

Ocean and climate conditions such as sea surface temperatures, air temperatures, strength of upwelling, El Niño events, salinity, ocean currents, wind speed, and primary and secondary productivity affect all facets of the physical, biological, and chemical processes in the marine environment. Some of the conditions associated with El Niño events include warmer water temperatures, weak upwelling, low primary productivity (which leads to decreased zooplankton biomass), decreased southward transport of subarctic water, and increased sea levels (Pearcy 1992). For juvenile steelhead, warmer water and weak upwelling are possibly the most important of the ocean conditions associated with El Niño. Because of the weakened upwelling during an El Niño year, juvenile California steelhead must migrate more actively offshore through possibly stressful warm waters with numerous inshore predators. Strong upwelling is probably beneficial because of the greater transport of smolts offshore, beyond major concentrations of inshore predators (Pearcy 1992). Investigations are currently under way to examine decadal oscillations in coastal marine environmental conditions and the associated biological changes that may affect the survival, growth, and recruitment of steelhead to the adult population.

4.A.3.4.6 Status and Trends

Historical CCV steelhead run sizes are difficult to estimate given the paucity of data but it is postulated that it may have approached 1 to 2 million adults annually (McEwan 2001). By the early 1960s, steelhead run size had declined to approximately 40,000 adults (McEwan 2001), along with the decline in accessible habitat (Figure 4.A.3-3). Over the past 35 years, the total number of steelhead minus hatchery escapement entering the upper Sacramento River at the Red Bluff Diversion Dam have declined substantially (Table 4.A.3-1). The reduction in numbers from an average of 6,574 fish from 1967 to 1991, to an average of 1,282 fish from 1992 to 2006, represents a significant drop in the upper Sacramento River populations. Although data are limited, similar population reductions are expected to have occurred throughout the Sacramento–San Joaquin system.

The most recent status review of the CCV steelhead DPS (NMFS 2011) found that the status of the population appears to have worsened since the 2005 status review (Good et al. 2005); however, the status review concluded that the DPS should remain classified as threatened. Analysis of data from the Chipps Island monitoring program indicates that natural steelhead production has continued to decline and that hatchery origin fish represent an increasing fraction of the juvenile production in the Central Valley. In recent years, the proportion of hatchery produced juvenile steelhead in the catch has exceeded 90%, and in 2010 was 95% of the catch (NMFS 2011).



Note: Trapping in 2008 was discontinued after June due to high water temperatures.

SOURCE: Data from 1967 to 1991: Mills, T.J. and F. Fisher. 1994. Central Valley anadromous sport fish annual run-size, harvest, and population estimates, 1967 through 1991. August 1994. California Department of Fish and Game. Sacramento, CA. Data from 1992 to 2008, California Department of Fish and Game, Red Bluff. Unpublished data.

Figure 4.A.3-3. Estimated Historical Total Spawner Escapement Minus Hatchery Escapement of CCV steelhead in the Upper Sacramento River Upstream of the Red Bluff Diversion Dam (1967–2008)

Table 4.A.3-1. Temporal Occurrence of Adult and Juvenile CCV steelhead in the Central Valley.

CVV steelhead relative abundance	High				Medium				Low			
a) Adults												
Location Sacramento River	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[a] Immigration, RBDD												
[b] Holding, RBDD												
[c] Spawning, eggs, alevins, Keswick, RBDD												
[d] Kelt migration, RBDD												
b) Juveniles												
[e] Juvenile rearing, Keswick												
[e] Smolt emigration, RBDD												
a) Adults												
Location Feather River	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[f] Immigration, Feather River at Sac. R.confluence												
[b] Holding, Low flow channel												
[c] Spawning, eggs, alevins, low and high flow channel												
[d] Kelt emigration, Feather R. at Sac. R. confluence												
b) Juveniles												
[g] Juvenile rearing, low-flow, high-flow channel												
[g,h] Smolt emigration, Feather R. at Sac. confluence												
a) Adults												
Location American River	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
[d] Immigration, confluence with Sac. R.												
[b] Holding, Watt Avenue, Nimbus												
[c] Spawning, eggs, alevins, below Nimbus												

[d] Kelt emigration, Sac. R. confluence												
b) Juveniles												
[c] Juvenile rearing, below Nimbus, Sac. R. confluence												
[i] Smolt emigration, Sac. R. confluence												
Sources: [a](CDFG unpublished counts at RBDD 1966-1994); [b](D. Swank pers. comm.); [c](Reclamation 2008); [d] Inferred from spawning period; [e]Gaines and Martin 2002; [f] Hallock 1961; [g](Bilski and Kindopp 2009); [h] NMFS Oroville BiOp 2009; [i] SWRI 2001 Abreviations: RBDD = Red Bluff Diversion Dam, Keswick = Keswick Dam, Nimbus = Nimbus Fish Hatchery, Sac. R. = Sacramento River												

4.A.3.5 Threats and Stressors

In its latest 5-year status review, NMFS determined that the CCV steelhead DPS should remain classified as threatened. However, based on new information, NMFS determined that the status of the CCV steelhead DPS was worse than the previous review (Good et al. 2005), and the DPS faces an even greater extinction risk (NMFS 2011). This review found that the decline in natural production of steelhead had continued unabated since the 2005 status review, and the level of hatchery influence on the DPS corresponds to a moderate risk of extinction (NMFS 2011). A large factor affecting all the listed salmonids is the loss of spawning and rearing habitat upstream of various dams. The limiting factors that affect steelhead survival are high water temperatures, low flows and flow fluctuations, limited spawning and rearing habitat, blocked or delayed passage, and unscreened river diversions. CCV steelhead hatcheries currently include very few natural origin fish in their broodstock (USFWS 2012; California Hatchery Scientific Review Group 2012) and, as indicated previously, hatchery origin steelhead appear to be more abundant than natural origin fish (Figure 4.A.3-5). Given practices of CCV steelhead hatcheries, when hatchery origin steelhead spawn in-river they will likely exhibit poor fitness and will impair fitness of natural origin fish where introgression occurs (Araki et al. and others). Other factors that may influences steelhead distribution and abundance include predation; contaminants, harvest, operations, and disease.

The following conditions are important threats and stressors to CCV steelhead.

4.A.3.5.1 Reduced Access to and Quantity and Quality of Staging, Spawning, Egg Incubation, and Rearing Habitat

Adult steelhead historically migrated upstream into higher gradient reaches of rivers and tributaries where water temperatures were cooler, turbidity was lower, and gravel substrate size was suitable for spawning and egg incubation (McEwan 2001). Steelhead are known to migrate upstream into higher gradient and elevation reaches of the rivers and streams than fall-run Chinook salmon, which predominantly spawn at lower elevations in the valley floor. Most historical adult staging/holding, spawning, and rearing habitat for CCV steelhead is no longer accessible to upstream migrating steelhead. Access to this habitat has been blocked by artificial structures (i.e., dams and weirs) associated with water storage and conveyance; diversions; flood control; and municipal, industrial, agricultural, and hydropower purposes (Table 4.A.3-1) (McEwan and Jackson 1996; McEwan 2001; Reclamation 2004; Lindley et al. 2006; NMFS 2007). These impediments and barriers to upstream passage limit the geographic distribution of steelhead to lower elevation habitats in the Central Valley.

Steelhead in the Central Valley migrate upstream into the mainstem Sacramento River and major tributaries (e.g., American and Feather Rivers; Mill, Deer, Clear and Battle Creeks), and are also known to occur in tributaries to the San Joaquin River (e.g., Mokelumne, Cosumnes, Stanislaus, Merced, Tuolumne Rivers), where they spawn and rear. Steelhead do not currently spawn in the mainstem San Joaquin River.

4.A.3.5.2 Low Instream Flows and Flow Fluctuations

Adverse effects to steelhead stocks in the Sacramento and San Joaquin rivers have been mostly attributed to water development (McEwan and Jackson 1996). Specific examples include dams blocking access to upstream habitats, inadequate instream flows caused by water diversions, rapid flow fluctuations due to water conveyance needs and flood control operations, inadequate cold-water releases from upstream reservoirs, and juvenile entrainment into unscreened or poorly screened water diversions.

Reduced flows from dams and upstream water diversions can lower attraction cues for adult spawners, causing straying and delays in spawning or the inability to spawn (DWR 2005). Adult steelhead migration delays can reduce fecundity and egg viability and increase susceptibility to disease and harvest.

Measures to minimize effects on salmon will usually, though not always, result in concomitant effects on steelhead. However, life history differences between steelhead and Chinook salmon may also lead to different, and potentially conflicting, flow requirements for each species. Although the most important flow needs for steelhead in Central Valley rivers are for cold water during the summer and early fall, increased flows for Chinook salmon are typically scheduled for the spring and mid-fall migration periods. In some cases, such as the temperature criteria for winter-run Chinook salmon from Keswick to Red Bluff Diversion Dam (RBDD), reservoir operations coincide with steelhead requirements. Differences in the timing of flow needed by different species can create difficult management dilemmas, particularly during an extended drought.

4.A.3.5.3 Reduced Out-Migration Habitat

CCV steelhead emigrations usually occur during the winter through spring after the physiological transformation into smolts occurs in preparation for ocean entry. Emigrating smolts use the lower Sacramento River channels as a migration corridor to the ocean, spending little time rearing in this area. Modification of natural flow regimes from upstream reservoir operations has resulted in dampening of the hydrograph in most Central Valley rivers. Reductions in flow rates have also resulted in increased water temperature and residence time, and reductions in dissolved oxygen levels in localized areas of the Delta (e.g., Stockton Deep Water Ship Channel), which affect the value of migration habitat. Reduced dissolved oxygen levels in the lower San Joaquin River during late summer and early fall have been identified as a barrier and/or impediment to migration for CCV steelhead (Regional Water Resources Control Board 2003; Jassby and Van Nieuwenhuyse 2005). The data derived from the California Data Exchange Center files indicate that dissolved oxygen depressions occur during all migratory months, with significant events occurring from November through March when CCV steelhead adults and smolts would be utilizing this portion of the San Joaquin River as a migratory corridor (NMFS 2012).

Much of the Delta has been leveed, channelized, and fortified with riprap for flood protection, reducing and degrading the quality and availability of natural habitat for use by steelhead during migration (McEwan 2001). Channel margins have been considerably reduced because of the construction of levees and the armoring of their banks with riprap (Williams et al. 2009). These shallow-water habitat areas provide refuge from unfavorable hydraulic conditions and predation, as well as foraging habitat for out-migrating juvenile steelhead. Benefits for larger steelhead are likely much less than for foraging Chinook salmon fry, although the habitat may serve an important function as holding areas during downstream migration (Burau and Perry. 2007), thereby improving connectivity along the migration route.

Furthermore, impacts on the value, quantity, and availability of suitable habitat are likely to reduce fitness and increase susceptibility to entrainment, disease, exposure to contaminants, and predation.

4.A.3.5.4 Predation by Nonnative Species

Restriction of steelhead to mainstem habitats below dams may expose eggs and rearing juveniles to higher encounter rates with predators than would be expected in historical headwater habitats (McEwan and Jackson 1996). Predatory fish are generally found in higher numbers and species in main-stem rivers than headwater streams. Thus, losses to predators are probably greater in main-stem rivers as compared to what might be expected in historical spawning areas (CALFED 1998). However, essentially very little is known about predation on CCV steelhead. Native species such as the Sacramento pikeminnow are a potentially significant source of mortality in the Sacramento River at locations with anthropogenic structures (e.g., dams, bridges, or diversion structures) that provide ambushing sites and at times block migration upstream providing sites for aggregation. Tucker et al (1998) found salmonids present in pikeminnow and striped bass stomachs at Red Bluff Diversion Dam, although RBDD is no longer operated and does not present a barrier to predatory fish migration, thus lowering aggregation of these predators. On the Mokelumne River, Merz (2003) found that striped bass consumed 11–28% of hatchery Chinook production in the Woodbridge Dam after-bay, although a modern bladder type dam has been installed since that time lowering the possibility of predator aggregation due to the barrier. Predation on any species of fish is usually size dependent with smaller fish suffering heavier predation pressure. USFWS trawl data from Chipps Island indicates that a minor percentage of steelhead emigrate as YOY (Nobriga and Cadrett 2001). This would imply that most predation on steelhead occurs upstream of the Delta where the habitat use of small size classes has been shown to be affected by the presence of potential predators (Brown and Brasher 1995) and predation risk appears to be affected by habitat quality. However, predation by nonnative species is of particular concern. In general, the effect of nonnative predation on the CCV steelhead DPS is unknown, but predation is most likely a threat in areas with high densities of nonnative fish (e.g., small and large mouth bass, striped bass, and catfish), and where large numbers of recently released hatchery fish are aggregated, which would allow opportunistic predators to prey on out-migrating juvenile steelhead. However, steelhead were not listed as a prey item for any Delta fish by Turner and Kelly (1966), even though they were more abundant at that time. The lack of steelhead in the stomachs of Delta piscivores is consistent with the observation that few steelhead emigrate as YOY, and suggests predation pressure on the relatively large steelhead smolts migrating through the Delta may be lower than for juvenile Chinook. Nobriga and Feyrer (2007) investigated the feeding ecology of piscivorous fishes in nearshore habitats during 2001 and 2003 and no steelhead were found in any of the 570 striped bass stomachs, 320 largemouth bass stomachs, or 282 Sacramento pikeminnow foreguts examined. Predation risk may covary with increased temperatures. Metabolic rates of nonnative, predatory fish increase with increasing water temperatures based on bioenergetics studies (Loboschefsky et al. 2012; Miranda et al. 2010). Upstream gravel pits and flooded ponds, such as those that occur on the San Joaquin River and its tributaries, attract nonnative predators (DWR 2005). Nonnative aquatic vegetation, such as Brazilian waterweed (*Egeria densa*) and water hyacinth (*Eichhornia crassipes*), provide suitable habitat for nonnative predators (Brown and Michniuk 2007). The low spatial complexity of channelized waterways (e.g., riprap-lined levees that provide virtually no cover protection from predators) and general low habitat diversity elsewhere in the Delta reduces refuge cover and protection of steelhead from predators (Raleigh et al. 1984; Missildine et al. 2001; 70 FR 52488, September 2, 2005).

4.A.3.5.5 Harvest

Steelhead have been, and continue to be, an important recreational fishery in inland rivers throughout the Central Valley. Although there are no commercial fisheries for steelhead, steelhead fisheries include recreational fisheries in the Central Valley, recreational fishing for steelhead of hatchery origin is popular, but harvest is restricted to only visibly marked fish of hatchery origin (adipose fin clipped). Unmarked steelhead (adipose fin intact) must be released, reducing the take of naturally spawned wild fish. There is some concern about hooking and handling stress, causing mortality of steelhead parr and smolts on popular rivers such as the American and Feather. High water temperatures in the summer and fall likely contribute to any mortality caused by angling. The level of illegal harvest of Chinook salmon and steelhead in the Delta and bays is unknown. The effects of recreational fishing and this unknown level of illegal harvest on the abundance and population dynamics of wild CCV steelhead have not been quantified.

4.A.3.5.6 Reduced Genetic Diversity and Integrity

Artificial propagation programs for steelhead in Central Valley hatcheries present multiple threats to the wild steelhead population including reduced fitness resulting from hatchery practices causing domestication selection, mortality of natural steelhead in fisheries targeting hatchery origin steelhead, competition for prey and habitat, predation by hatchery origin fish on younger natural fish, disease transmission, and impediments to fish passage imposed by hatchery facilities. It is now recognized that Central Valley hatcheries are a significant and persistent threat to wild Chinook salmon and steelhead populations and fisheries (NMFS 2009b). One major concern with hatchery operations is the genetic introgression by hatchery origin fish that spawn naturally and interbreed with local natural populations (USFWS 2001; Reclamation 2004; Goodman 2005). Such introgression introduces maladaptive genetic changes to the wild steelhead stocks (McEwan and Jackson 1996; Myers et al. 2004).Steelhead broodstock at the Nimbus and formerly at the Mokelumne River hatcheries are of Eel and Mad River origin, which is an out-of-DPS source. Hatchery operations that include insufficient numbers of natural origin steelhead have been found to decrease steelhead fitness via domestication selection (Araki et al. 2007). Taking eggs and sperm from a large pool of individuals is a method for ameliorating loss of genetic diversity, but artificial selection for traits that assure individual success in a hatchery setting (e.g., rapid growth and tolerance to crowding) are avoidable by management actions that protect natural origin steelhead from hatchery steelhead introgression and which include natural origin steelhead as hatchery broodstock (HSRG 2014).

The increase in Central Valley hatchery production has reversed the composition of the steelhead population, from 88% naturally produced fish in the 1950s (McEwan 2001) to an estimated 23% to 37% naturally produced fish by 2000 (Nobriga and Cadrett 2001), and less than 10% currently (NMFS 2011). Scientific information available for other areas (e.g., HSRG 2014) suggests Central Valley steelhead hatcheries practices have substantially contributed to reduced viability of the listed steelhead populations (NMFS 2012).

4.A.3.5.7 Entrainment

Juvenile steelhead migrating downstream through the Delta can become vulnerable to entrainment and salvage at the CVP/SWP export facilities, primarily between February and May. Multiple factors can influence the vulnerability of juvenile steelhead to entrainment by CVP/SWP export facilities, including the geographic distribution of steelhead in the Delta and hydrodynamic factors

Tidally averaged flow (or net flow) in Old and Middle rivers (OMR flows) are often negative because of export through the Federal and state export facilities. The hydrodynamic conditions associated with negative OMR flows have been hypothesized by NMFS (2009b) to be associated with increased southward movement of emigrating juveniles in those channels, resulting in delayed emigration through the Delta, and directly or indirectly increasing vulnerability to the many stressors within the central and south Delta. Previous studies have observed increased entrainment of tagged salmonids at the CVP/SWP facilities when exports are increased (NMFS 2009b, Zeug and Cavallo 2014). Recent independent science reviews have observed numerous parameters that influence juvenile salmonid movement and that tidally averaged flows or velocities cannot be detected by juvenile salmonids. These include instantaneous flow velocities which are perceived by the fish in its immediate surrounding environment, detection of chemical constituents in the water by chemo-sensory organs that elicit migratory behavioral responses, and spatial distribution of the migrating fish across the river channel in the vicinity of junctions that affect ultimate route selection (Anderson et al. 2012; Monismith et al. 2014).

DWR and Reclamation (1999) found significant relationships between total monthly exports in January through May and monthly steelhead salvage at CVP/SWP facilities. As described previously, the hydrodynamic effect of exports on water velocities on a scale perceivable to juvenile salmonids occurs primarily in the south Delta. Steelhead reaching the south Delta are more likely to be entrained if exports are higher, but also because louver efficiency at export fish facilities increases at higher export levels (Karp et al. 1995). During the past several years, additional investigations have used radio- or acoustically tagged juvenile and adult (post spawning adults) steelhead to monitor their migration behavior through the Delta channels and to assess the effects of changes in hydraulic cues and CVP/SWP export operations on migration (Holbrook et al. 2009; Perry et al. 2010; San Joaquin River Group Authority 2010; Delaney et al. 2014; Cavallo et al. 2015). These studies are ongoing, but so far have confirmed that the hydrodynamic effect of exports on juvenile salmonids occurs primarily in closer proximity to the export facilities. Studies have also been conducted to assess the potential losses of juvenile steelhead to predation by adult striped bass during passage through Clifton Court Forebay (Clark et al. 2009). Results of these studies have estimated that prescreen losses of juvenile steelhead in Clifton Court Forebay are greater than 80%.

In addition to CVP/SWP export facilities, there are more than 2,200 small water diversions in the Delta, of which the majority are unscreened (Herren and Kawasaki 2001). The risk of entrainment is a function of the size of juvenile fish and the slot opening of the screen mesh (Tomljanovich et al. 1978; Schneeberger and Jude 1981; Zeitoun et al. 1981; Weisberg et al. 1987). Although entrainment/salvage of steelhead at the CVP/SWP export facilities is well documented, it is unclear how many juvenile steelhead are entrained at other unscreened Delta diversions. Because steelhead are moderately large (greater than 200-millimeter fork length) and relatively strong swimmers when out-migrating, the effects on steelhead of small in-Delta agricultural water diversions are thought to be lower than those on other Central Valley salmonids. In addition, many of the juvenile steelhead migrate downstream through the Delta during the late winter or early spring before many of the agricultural irrigation diversions are operating. Steelhead may move into the Colusa Drain via Yolo Bypass into the Knights Landing Ridge-cut or up the Sacramento River, then moving through the Knights Landing outfall gates. Once in the canal fish migrate upstream until barriers are reached that prevent further migration. Unless rescued at these points, they die and are lost to the population. In 2015 a pickett weir was installed in front of the Knights Landing Outfall Gates that should prevent most fish from moving through the radial gates. Power plants have the ability to impinge juvenile steelhead on the existing intake screens. However, use of cooling water is currently low with the retirement of older units. Furthermore, newer units are equipped with a closed-cycle cooling system that virtually eliminates the risk of impingement of juvenile steelhead.

4.A.3.5.8 Exposure to Toxins

Toxic chemicals are widespread throughout the Sacramento and San Joaquin River basins and may occur on a more localized scale in response to episodic events (e.g., storm water runoff, point source discharges, etc.). Most anthropogenic chemicals and waste materials, including toxic organic and inorganic chemicals, eventually accumulate in sediment. Exposure to contaminated sediments may cause deleterious effects to listed salmonids if a fish swims through a plume of the resuspended sediments or rests on contaminated substrate and absorbs the toxic compounds through one of several routes: dermal contact, ingestion, or uptake across the gills. 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. The degree of exposure to the salmonids depends on their trophic level and the amount of contaminated forage base they consume. These toxic substances include mercury, selenium, copper, pyrethroids, and endocrine disruptors with the potential to affect fish health and condition, and negatively affect steelhead distribution and abundance directly or indirectly. Some loads of toxics, such as selenium, are much higher in the San Joaquin River than the Sacramento River because they are naturally occurring in the alluvial soils and have been leached by irrigation water and concentrated by evapotranspiration (Nichols et al. 1986). This may indicate that the potential effects of chronic exposure could be greater for steelhead of San Joaquin River origin. Additionally, agricultural return flows that may contain toxic chemicals are widely distributed throughout the Sacramento and San Joaquin Rivers and the Delta, although dilution flows from the rivers may reduce chemical concentrations to sublethal levels. Sublethal concentrations of toxic substances may interact with other stressors on salmonids, such as increasing their vulnerability to predation or disease (Werner 2007). For example, Clifford et al. (2005) found in a laboratory setting that juvenile fall-run Chinook salmon exposed to sublethal levels of a common pyrethroid, esfenvalerate, were more susceptible to infectious hematopoietic necrosis virus than those not exposed to esfenvalerate. Although not tested on steelhead, a similar response is likely; however, juvenile steelhead generally migrate through the Delta in a comparatively shorter time than Chinook salmon. The short duration may decrease juvenile steelhead exposure and susceptibility to toxic substances in the Delta. Adult migrating steelhead may be less affected by toxins in the Delta because they are not feeding, and thus not bioaccumulating toxic exposure, and they are moving rapidly through the system.

Iron Mountain Mine, located adjacent to the upper Sacramento River, has been a source of trace elements that are known to adversely affect aquatic organisms (Upper Sacramento River Fisheries and Riparian Habitat Advisory Council 1989). Storage limitations and limited availability of dilution flows have caused downstream copper and zinc levels to exceed salmonid tolerances and resulted in documented fish kills in the 1960s and 1970s (Reclamation 2004). The U.S. Environmental Protection Agency's Iron Mountain Mine remediation program has removed toxic metals in acidic mine drainage from the Spring Creek watershed with a state-of-the-art lime neutralization plant. Contaminant loading into the Sacramento River from Iron Mountain Mine has shown measurable reductions since the early 1990s.

Ammonia[3] released from the City of Stockton Wastewater Treatment Plant contributes to the low dissolved oxygen in the adjacent Deep Water Ship Channel. In addition to the adverse effects of the lowered dissolved oxygen on salmonid physiology, ammonia is toxic to salmonids at low concentrations. Actions have been implemented to remedy this source of ammonia, by modifying the treatment train at the wastewater facility (NMFS 2012).

4.A.3.5.9 Increased Water Temperature

Water temperature is among the physical factors that affect the value of habitat for salmonid adult holding, spawning and egg incubation, juvenile rearing, and migration. Adverse sublethal and lethal effects can result from exposure to elevated water temperatures at sensitive life stages, such as during incubation or rearing. Water temperature criteria for some life stages of salmonids in the Central Valley have been listed by NMFS (2009a) (Table 4.A.3-2). The tolerance of

[3] Ammonia in water generally forms some amount of ammonium. Therefore, the use of the term *ammonia* implies that both ammonia and ammonium may be present.

steelhead to water temperatures depends on life stage, acclimation history, food availability, duration of exposure, health of the individual, and other factors such as predator avoidance (Myrick and Cech 2004; Reclamation 2004). Higher water temperatures can lead to physiological stress, reduced growth rate, reduced spawning success, and increased mortality of steelhead (Myrick and Cech 2001). Temperature can also indirectly influence disease incidence and predation (Waples et al. 2007). Exposure to seasonally elevated water temperatures may occur from reductions in flow because of upstream reservoir operations, reductions in riparian vegetation, channel shading, local climate, and solar radiation. The installation of the Shasta Temperature Control Device in 1998, in combination with reservoir management to maintain the cold water pool, has reduced many of the temperature issues on the Sacramento River. During dry years, however, the release of cold water from Shasta Dam is still limited. As the river flows farther downstream, particularly during the warm spring, summer, and early fall months, water temperatures continue to increase until they reach thermal equilibrium with atmospheric conditions. Because of the longitudinal gradient of seasonal water temperatures, the coldest water and, therefore, the best areas for steelhead spawning and rearing are typically located immediately downstream of the dam.

Increased temperature can also arise from a reduction in shade over rivers by tree removal (Watanabe et al. 2005). Because river water is typically in thermal equilibrium with atmospheric conditions by the time it enters the Delta, this issue is caused primarily by actions upstream of the Delta. Because the Delta channels are relatively wide, additional riparian vegetation will not significantly reduce water temperatures.

Juvenile CCV steelhead hold and rear in riffles and pools at higher elevations in the watershed. Flow reductions, resulting from natural hydrologic conditions during the summer, evapotranspiration, or surface and groundwater extractions may all contribute to exposure to elevated temperatures and increased levels of stress or mortality. Dense riparian vegetation, streams incised into canyons that provide shading, cool water springs, and availability of deep holding pools are factors that affect summer rearing conditions for CCV juvenile steelhead. The effects of climate change and global warming patterns, in combination with changes in precipitation and seasonal hydrology in the future are important factors that may adversely affect the health and long-term viability of CCV steelhead (Lindley et al. 2007).

Myrick (1998; Myrick and Cech 2000) found the preferred temperatures for Mokelumne River Fish Installation, Feather River Hatchery, and naturally spawned Feather River juvenile steelhead placed into thermal gradients were between 62.5°F and 68°F (17 and 20°C). Myrick and Cech (2005) also found that Nimbus-strain steelhead had a higher growth rate at 66°F (19°C) than groups of steelhead raised at lower temperatures. This is considerably warmer than the rearing temperature recommended by McEwan and Jackson (1996). Feather River snorkel survey observations and temperature data from summer 1999 also appear to corroborate Myrick's (1998; Myrick and Cech 2000) results. Steelhead in the American River have been observed in snorkel surveys, captured by seining, and passive integrated transponder (PIT) tagged in habitats with a daily average temperature of 72°F and a daily maximum over 74°F (California Department of Fish and Game [DFG] and the U.S. Bureau of Reclamation [Reclamation] unpublished data, as cited in U.S. Bureau of Reclamation 2008).

Table 4.A.3-2. Recommended Water Temperatures (°F) that Provide for Highest Survival for Life Stages of Steelhead in Central Valley Streams from McEwan and Jackson (1996), Myrick (1998), Myrick and Cech (2000, 2001), and Piper et al. (1982), Bell (1991), Zaugg (1981).

Life Stage	Temperature Recommendation (°F)
Migrating adult	46–52
Holding adult	50–56
Spawning	39–52
Egg incubation	48–52
Juvenile rearing	<65
Smoltification	<54

4.A.3.6 Description of Viable Salmonid Population (VSP) Parameters

NMFS measures the conservation status of salmonids, with the viable salmonid population (VSP) framework and uses it to identify the attributes needed to assess the effects of management and conservation actions. The framework is known as the VSP concept (McElhany *et al*. 2000). The VSP concept measures population performance in term of four key parameters: abundance, population growth rate, spatial structure, and diversity.

4.A.3.6.1 Abundance

Historical CCV steelhead run sizes are difficult to estimate given the paucity of data but it is postulated that it may have approached 1 to 2 million adults annually (McEwan 2001). By the early 1960s, steelhead run size had declined to approximately 40,000 adults (McEwan 2001), along with the decline in accessible habitat. Hallock *et al.* (1961) estimated an average of 20,540 adult steelhead through the 1960s in the Sacramento River upstream of the Feather River. Over the past 35 years, total escapement minus hatchery escapement of steelhead populations in the upper Sacramento River has declined substantially (Figure 4.A.3-3). The reduction in numbers from an average of 6,574 fish from 1967 to 1991, to an average of 1,282 fish from 1992 to 2006, represents a significant drop in the upper Sacramento River populations.

The available data on occurrence currently is limited to redd surveys and the returns at hatcheries on a small number of creeks and rivers. Because of difficult conditions in conducting redd surveys during the winter-spring spawning period of CCV steelhead, hatchery data is more reliable. To get a more broad view of abundance American River steelhead redd counts were included in the analysis, as some of the fish spawning in the river are naturally produced, and therefore part of the DPS.

One of the better data sources is Coleman National Fish Hatchery (CNFH), which operates a weir on Battle Creek. The Battle Creek weir is continually in place during the hatchery spawning season, which typically runs from August through February. Because of changes in hatchery operations there are nuances to the data. In 2005, NMFS requested that CNFH stop transferring hatchery (adipose fin clipped) above the weir. CNFH also transferred 1,000 hatchery steelhead to Keswick Reservoir in 2003 and these fish are not included into data. Although all CCV steelhead have been marked since 1998, prior to 2002, hatchery and natural-origin steelhead in Battle Creek were not differentiable, and all steelhead were managed as a single, homogeneous stock. Abundance estimates of natural-origin steelhead in Battle Creek began in 2001. These estimates of steelhead abundance include all *O. mykiss*, including resident and anadromous fish. The result is that the only unbiased time series for Battle Creek is the number of unclipped (wild) steelhead since 2001, which have declined slightly since that time, mostly because of the high returns observed in 2002 and 2003 (Figure 4.A.3-4). Returning steelhead to CNFH have not shown consistent returns over the years. Between 2003 and 2012, the number of hatchery steelhead has ranged from 624 to 2,968. Wild steelhead represent a small fraction of overall returns, but their numbers have remained relatively steady, typically 200–500 fish each year (Figure 4.A.3-5).

Clear Creek steelhead population appears to have increased in abundance with the removal of Saeltzer Dam in 2000. The number of redds observed in surveys has steadily increased since 2001 (Figure 4.A.3-6). The average redd index from 2001 to 2011 is 157, which represents somewhere between 128 and 255 spawning steelhead each year, which are most likely wild steelhead, as no hatchery fish are stocked within Clear Creek.

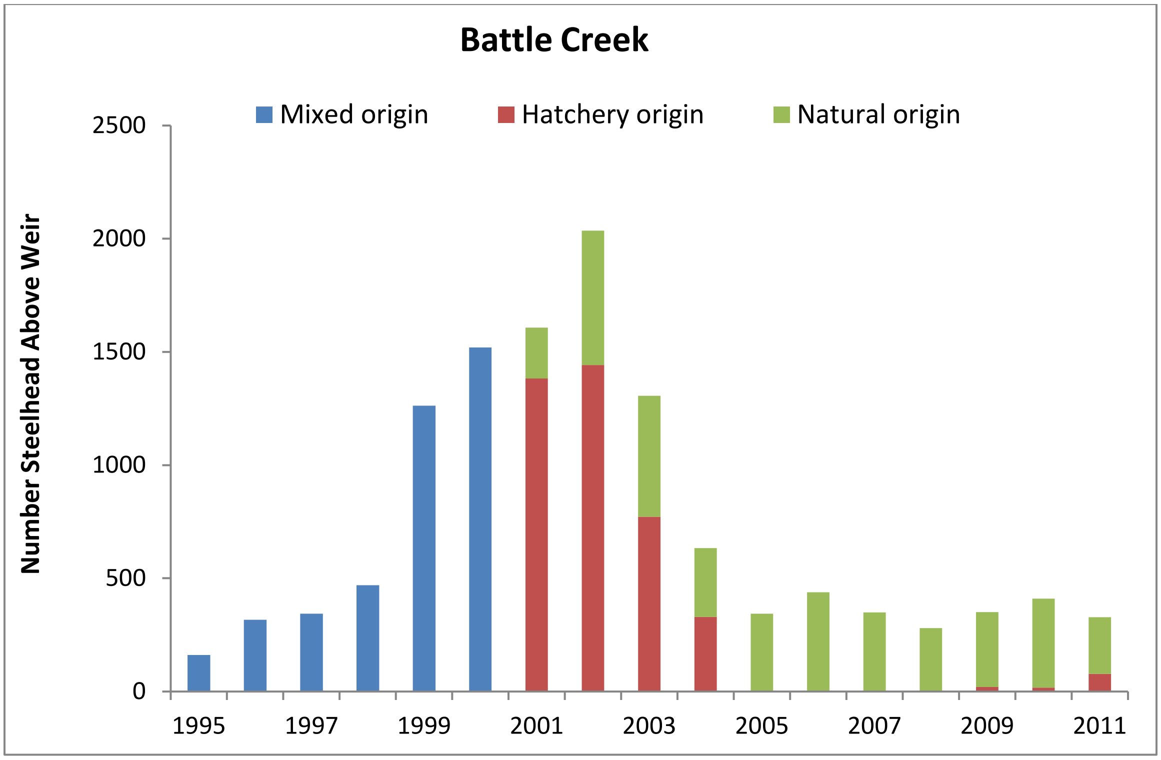


Figure 4.A.3-4. Steelhead Returns to Battle Creek from 1995-2009. Starting in 2001, fish were classified as either wild (unclipped) or hatchery produced (clipped). Includes fish passed above the weir during broodstock collection and fish passing through the fish ladder March 1 to August 31. Data are from USFWS.

Redd counts on the American River have averaged 164 (2002–2007, 2010), with redd population estimates ranging from 164–479 based upon 1 redd per female and 82–240 based upon 2 redds per female (Hannon and Deason 2008; Hannon et al. 2003; Chase 2010).

The Mokelumne River Hatchery has raised Feather River Hatchery steelhead since 2002. The annual escapement (2002–2010) has averaged 99 fish. A full 32% of the total return was unmarked and there is a high probability that these fish included non-anadromous forms, which are not included in the DPS. In a study of 119 naturally produced O. mykiss tagged with acoustic tags in 2007–2008 less than 5% migrated to the ocean (Workman et al. 2008).

Steelhead escapement to the Feather River Hatchery has decreased over time, with recent hatchery returns shown in Figure 4.A.3-5. Most steelhead in the Feather River are hatchery-derived stock, with stocking levels remaining fairly constant and it may be that in-river and ocean survival is low for this stock.

The pumping facilities in the South Delta provide another means of measuring relative abundance of steelhead within the Sacramento-San Joaquin system and the ratio of hatchery (adipose clipped) fish and wild steelhead (CDFG; NMFS 2011). Salvage of steelhead at the pumping facilities has varied over time (1993–2010) and the number of “wild” or unclipped steelhead has declined since 100% adipose fin clipping was instituted for CCV steelhead in 1998 (Figure 4.A.3-6).

Catches of steelhead at Coleman and the Feather River hatcheries dropped sharply in 2009 and 2010 following three consecutive drought years 2007–2009 and a below normal water year in 2010. These conditions may have added to low in river survival and could have been compounded by poor ocean upwelling conditions in 2005 and 2006, which may have limited foods sources along the Northern California coast (Lindley et al. 2009). “Wild” (non-adipose clipped) steelhead escapement numbers appear to have been affected to a lesser degree based upon hatchery returns and instream red counts on Clear Creek, and the American and Mokelumne Rivers.

Based upon the available data on CCV steelhead there has been a steady decline since the 1960’s and 1970’s and a precipitous decline from postulated historical numbers; however, there seems to be no clear trends since 2000. Numbers of unclipped steelhead seem to be holding at a steady rate and in some cases even increasing (Clear Creek), but they number in the hundreds and make up a very small proportion of the total population.

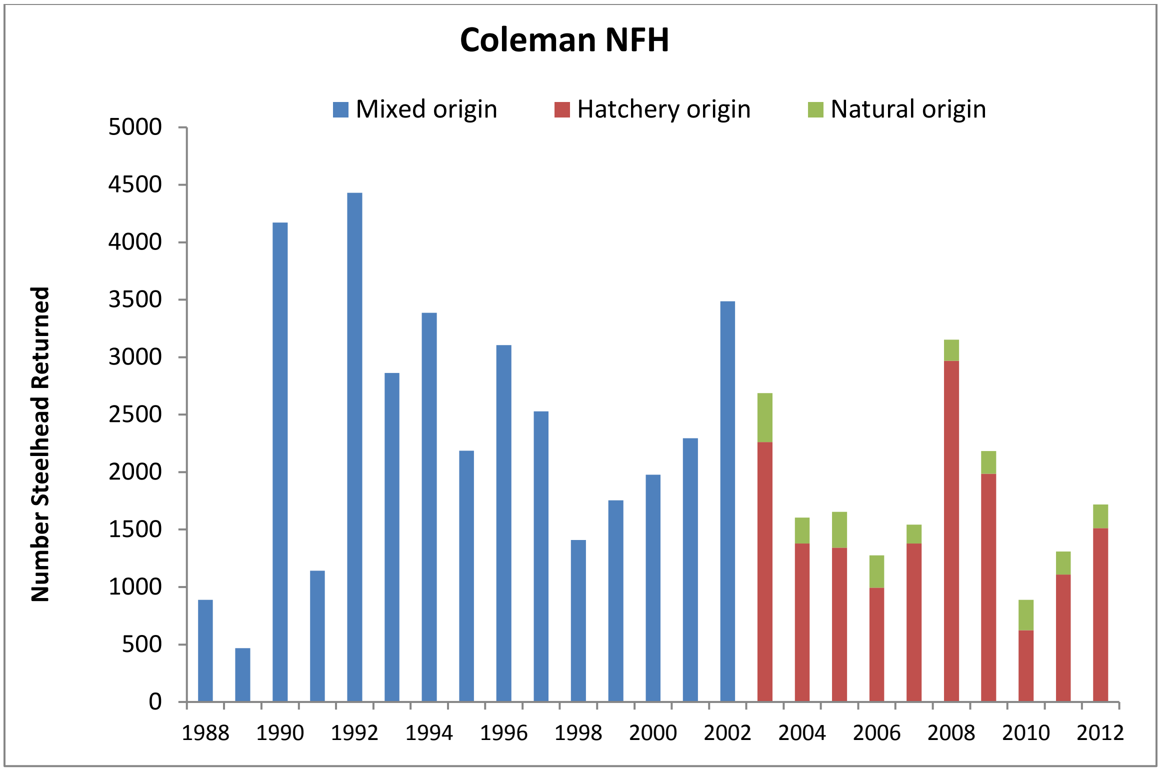


Figure 4.A.3-5. Number of Steelhead that Returned to the Coleman National Fish Hatchery Each Year. Adipose fin-clipping of hatchery smolts started in 1998, and since 2003 all returning steelhead have been categorized by origin.

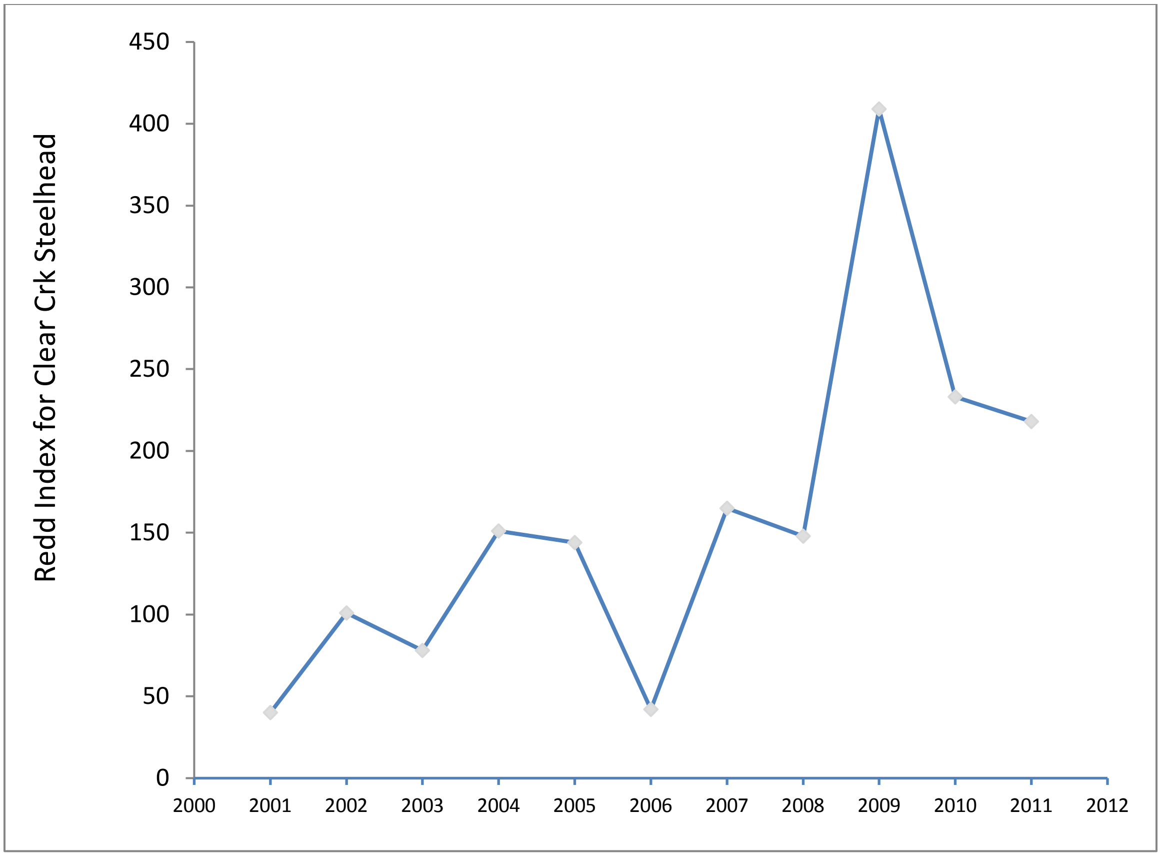


Figure 4.A.3-6. Redd Counts from USFWS Surveys on Clear Creek from 2001-2011.

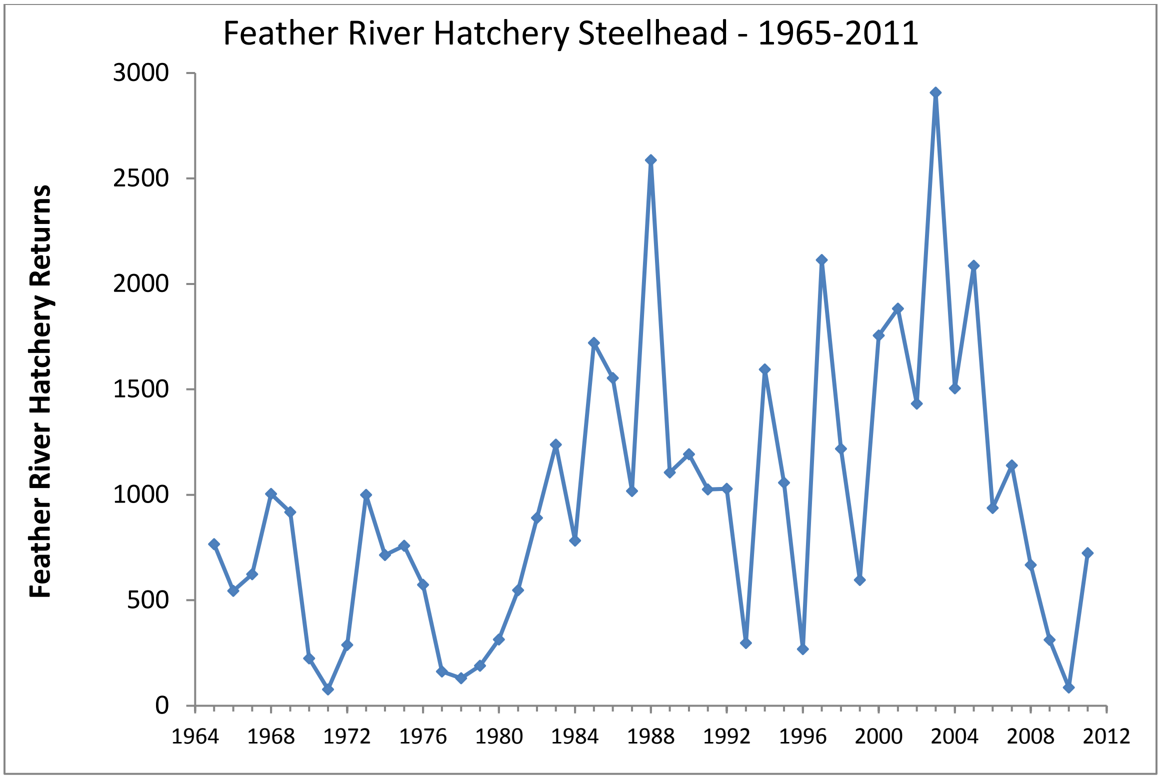


Figure 4.A.3-7. Number of Steelhead that Returned to the Feather River Fish Hatchery Each Year

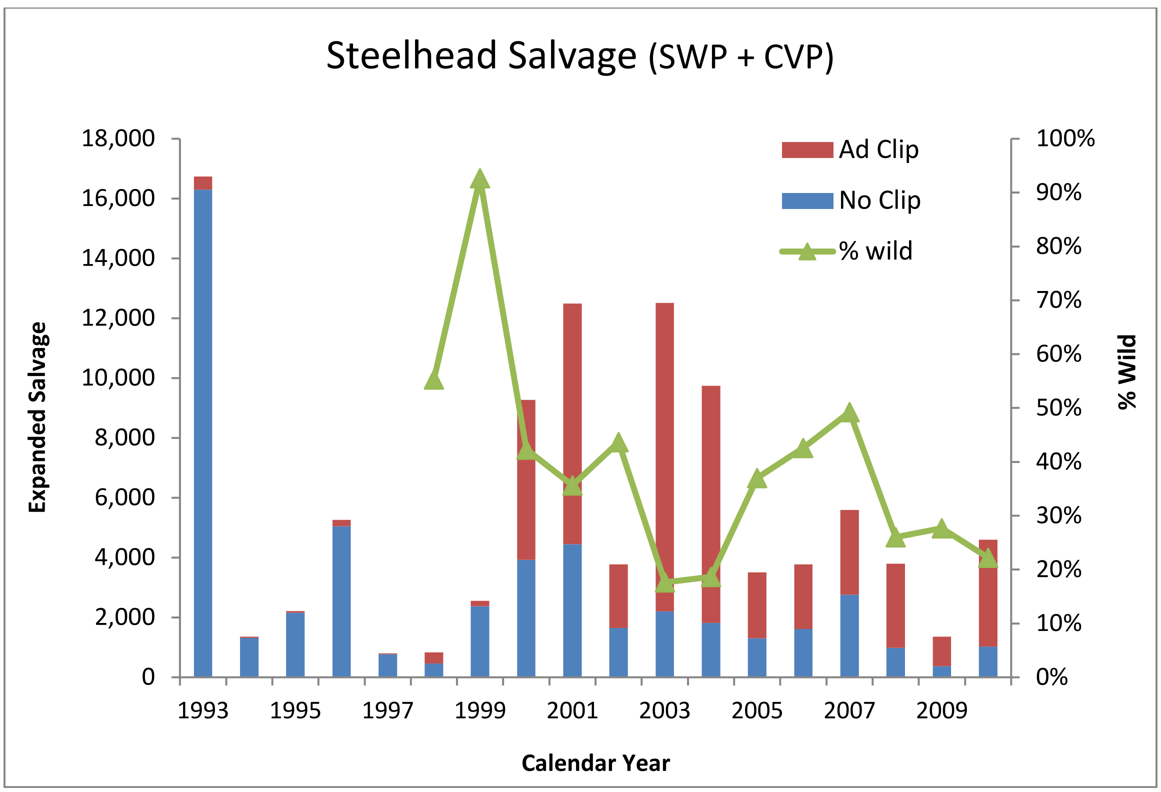


Figure 4.A.3-8. Steelhead Salvaged in the Delta Fish Collection Facilities from 1993 to 2010

4.A.3.6.2 Productivity

Using incidental catches in trawl gear as a proxy it is estimated that 100,000 to 300,000 unclipped (wild) juvenile steelhead emigrate from the Central Valley each season (Good et al. 2005). Low numbers of steelhead caught by California Department of Fish and Wildlife (CDFW) and USFWS in the Mossdale trawl survey indicate that productivity within the San Joaquin River tributaries is low. The Chipps Island midwater trawl data collected by USFWS provides an additional source showing the trend over time (Williams et al. 2011).

Nobriga and Cadrett (2001) estimated that 400,000 to 700,000 wild steelhead smolts are produced each year based on the ratio of wild (unclipped) versus hatchery (clipped) steelhead caught in the Chipps Island Trawl Survey 1998–2000.

The percentage of natural steelhead production as measured in the Chipps Island Trawl by USFWS has steadily declined over the years and hatchery fish are increasingly represented in the catch to the point where in 2007, 2010, and 2011 they represented over 90% to the total steelhead smolts caught (Figure 4.A.3-9). Because the total number of marked hatchery steelhead has been consistent, this indicated a decline in natural production of CCV steelhead.

In the Mokelumne River the overall trend suggests that redd numbers have slightly increased over the years (2001–2012), but many may be resident rainbow trout. Satterthwaite et al. (2010) postulates that Mokelumne steelhead are likely to be a mix of resident and anadromous life histories, with the resident form being favored because of intermediate growth patterns and highly variable survival during emigration and ocean residency (Figure 4.A.3-10).

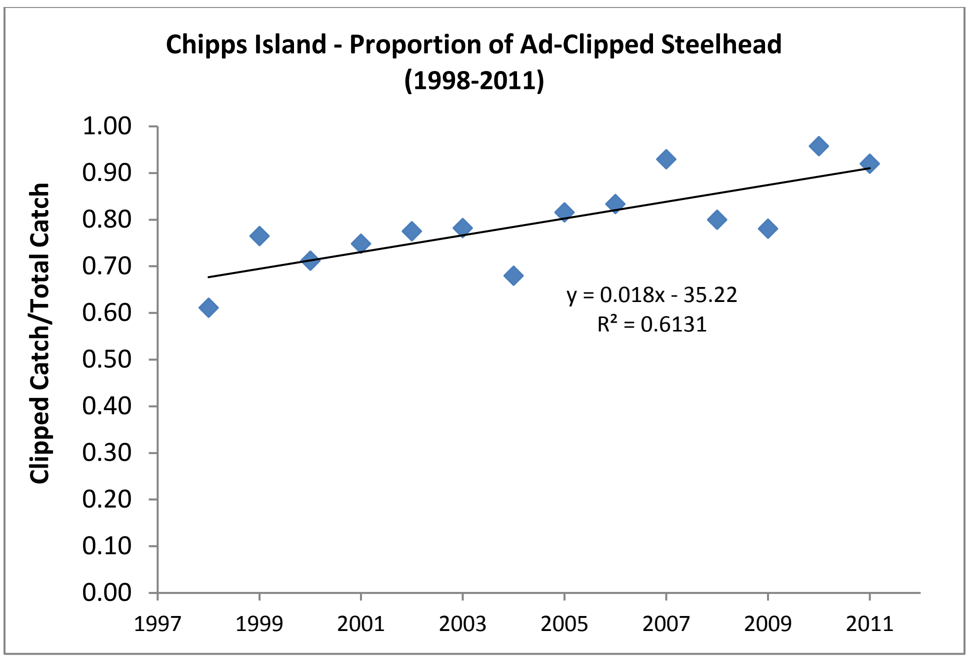


Figure 4.A.3-9. Fraction of Steelhead Catch Bearing an adipose Fin Clip in USFWS Chipps Island Midwater Trawl Survey from 1998 to 2011

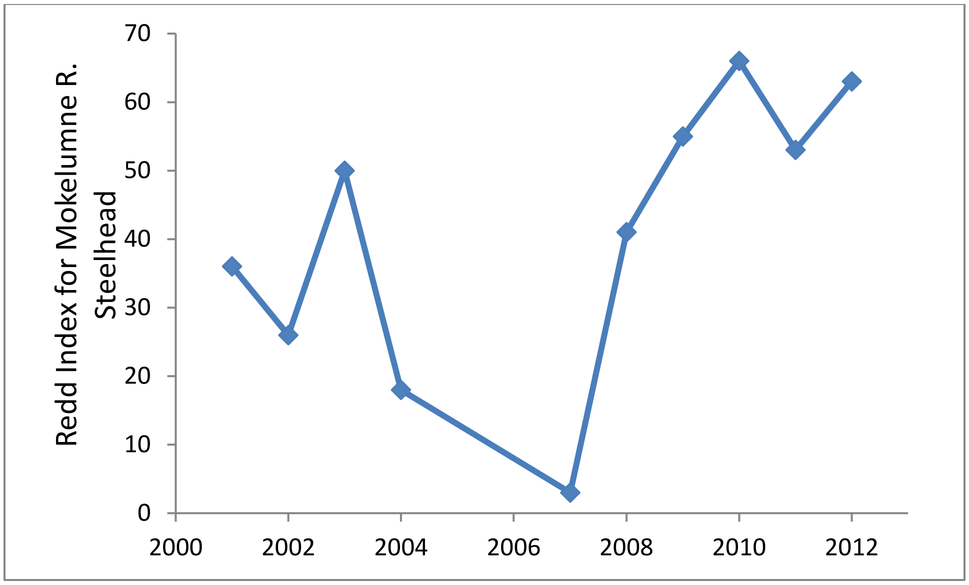


Figure 4.A.3-10. Redd Counts from EBMUD surveys on Mokelumne River 2001-2012

Some populations of wild CCV steelhead appear to be improving (Clear Creek), while others seem to be holding steady (Battle Creek) even with historic dry conditions and poor ocean upwelling, when compared to survival of hatchery fish (NMFS 2011). Since 2003 steelhead have been sorted into wild and hatchery fish based upon whether they have their adipose fin clipped and only wild fish are allowed upstream of the hatchery weir into upper Battle Creek. From Figure 4.A.3-5 it can be seen that wild fish have had fairly steady escapement of about 200–300 fish per year. It is also clear that the wild fish are heavily outnumbered by their hatchery counterparts, which have shown much larger fluctuations in escapement, ranging from 624 to 2,968 adults per year.

4.A.3.6.3 Spatial Structure

CCV steelhead were widely distributed historically throughout the Sacramento and San Joaquin Rivers (Figure 4.A.3-10) (Busby et al. 1996; McEwan 2001). Steelhead inhabited waterways from the upper Sacramento and Pit River systems (now inaccessible because of Shasta and Keswick Dams) south to the Kings River and possibly the Kern River systems, and in both east- and west-side Sacramento River tributaries (Yoshiyama et al. 1996). Lindley et al. (2006) estimated that there were historically at least 81 independent CCV steelhead populations distributed primarily throughout the eastern tributaries of the Sacramento and San Joaquin Rivers.

The geographic distribution of spawning and juvenile rearing habitat for CCV steelhead has been greatly reduced by the construction of dams (McEwan and Jackson 1996; McEwan 2001). Presently, impassable dams block access to 80% of historically available habitat and all spawning habitat for approximately 38% of historic populations (Lindley et al. 2006). Existing wild steelhead stocks in the Central Valley inhabit the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks, and a few wild steelhead are produced in the American and Feather Rivers (McEwan and Jackson 1996).

CCV steelhead are well distributed below dams blocking passage to headwater tributaries (Good et al. 2005; NMFS 2011). Studies of SR/CA ratios within the primordia of otoliths by Zimmerman et al. (2009) conclusively showed anadromy occurring in San Joaquin tributaries, but at lower levels than what occurs in the Sacramento River and its tributaries.

Screw trap monitoring of emigrating juvenile Chinook has detected small numbers of steelhead smolts in the Stanislaus, Tuolumne, Mokelumne, and Calaveras Rivers, and other streams thought previously to contain only resident rainbow trout (McEwan 2001). Small numbers of steelhead smolts have been captured on the Stanislaus River each year since the beginning of monitoring in 1995 (S.P. Cramer and Associates 2000; FISHBIO 2012, 2013a). Only one emigrating smolt was captured in a screw trap during the 2012 season on the Tuolumne River, but the efficiency of screw traps can be low, so it is unlikely that only one smolt emigrated from the system (FISHBIO 2013b). No juvenile rainbow trout had been caught in rotary screw traps in the Merced River since monitoring began in 1999 until 2012 when 381 were captured (FISHBIO 2013c). This capture event might have been propagated by a rapid increase in the hydrograph over a 24-hour period due to an intense storm event in the drainage. Using weirs with counting cameras, 15 *O. mykiss* (steelhead and resident forms) were detected migrating upstream in the Tuolumne River and 82 in the Stanislaus River in 2012 (FISHBIO 2012, 2013a). On the Merced River, one adult steelhead was detected by a fish-counting weir in 2012. Annual Kodiak trawl surveys by CDFW and USFWS captured 17 juvenile rainbow trout in the Mossdale survey in the San Joaquin River (USFWS 2013).

Low numbers of both immigrating adults and outmigrating juveniles suggest that CCV steelhead populations within the San Joaquin tributaries are at low levels. If the CCV steelhead DPS were to lose these populations, the spatial structure of the DPS would be greatly impacted and would further affect the viability of the DPS.

Providing passage to steelhead over impassable dams does have the potential to greatly increase the spatial diversity of CCV steelhead. Habitat created for spring-run Chinook salmon downstream of Friant Dam under the San Joaquin River Restoration Program (SJRRP) also has the potential to benefit CCV steelhead (NMFS 2011).

4.A.3.6.4 Diversity

Genetic Diversity: Due to an over 80% decline in habitat and diversity of habitats, CCV steelhead abundance and growth rates continue to decline (Lindley *et al.* 2006). Population reductions were supported by genetic analysis (Nielsen *et al.* 2003). In a genetic analysis of steelhead populations from the Central Valley Garza and Pearse (2008) found that below dam populations were more closely related to each other than to populations above the barrier, which is unlike coastal populations. This suggests that populations above barriers contain more of the ancestral heredity than those below barriers where out-of-basin stock transfers and inter-hatchery transfers have occurred.

The majority of annual spawning runs are comprised of hatchery origin fish whose management compromises CCV steelhead genetic diversity and puts the wild population at high risk of extinction (Lindley et al. 2007). Four Central Valley hatcheries (Coleman National Fish Hatchery, Feather River Fish Hatchery, Nimbus Fish Hatchery, and Mokelumne River Fish

Hatchery) when combined release approximately 1.6 million yearling steelhead smolts each year. These hatchery programs were intended to mitigate for loss of habitat above impassable dams, but now drive a large percentage of the steelhead population groups within the CCV steelhead DPS. Two of these hatcheries Nimbus and Mokelumne) started their programs with out-of-basin stock from the Eel and Mad Rivers, although the Mokelumne River hatchery stopped importing eggs from Nimbus Hatchery in 1998, thus these programs are not considered part of the DPS.

Life-History Diversity: Steelhead can be divided into two life history types based on their state of sexual maturity at the time of river entry and the duration of their spawning migration: stream-maturing and ocean-maturing. Stream-maturing steelhead enter fresh water in a sexually immature condition and require several months to mature prior to spawning, whereas ocean-maturing steelhead enter fresh water with well-developed gonads and spawn shortly after river entry. These two life history types are more commonly referred to by their season of freshwater entry (i.e., summer [stream-maturing] and winter [ocean-maturing] steelhead). A variation of the two forms occurs in the Central Valley and primarily migrates into the system in the fall, then spawns during the winter and early spring, although this form is referred to as *winter-run* (McEwan and Jackson 1996). There are, however, indications that summer steelhead were present in the Sacramento River system prior to the commencement of large-scale dam construction in the 1940s (Interagency Ecological Program Steelhead Project Work Team 1999; McEwan 2001).

Between 1944 and 1947, annual counts of summer-run steelhead passing through the Old Folsom Dam fish ladder during May, June, and July ranged from 400 to 1,246 fish (Gerstung 1971). After 1950, when the fish ladder at Old Folsom Dam was destroyed by flood flows, summer-run steelhead were no longer able to access their historic spawning areas, and perished in the warm water downstream of Old Folsom Dam.

At present, only winter-run (ocean maturing) steelhead currently are found in California Central Valley rivers and streams (Moyle 2002; McEwan and Jackson 1996). The summer form of steelhead have been extirpated from the Central Valley because impassable dams have blocked steelhead from accessing suitable holding and staging habitat, such as cold-water pools in the headwaters of California Central Valley streams (Lindley *et al.* 2006).

Juvenile steelhead growth rates are highly correlated with freshwater residence time, with faster growth resulting in earlier smolt ages and smaller sizes at smolting (Peven *et al.* 1994, Seelbach 1993). In a scale analysis study of adult steelhead caught in the Sacramento River upstream the Feather River confluence, 70 had smolted at age-2, 29 at age-1, and one at age-3 (Hallock et al. 1961). Seventeen of the adults had spawned previously, with three fish on their third spawning migration, and one on its fifth. Most CCV steelhead adults return to their natal stream at age-2 to age-4 years (Hallock *et al.* 1961, McEwan and Jackson 1996).

Deer and Mill creeks were monitored from 1994 to 2010 by the CDFW using rotary screw traps to capture downstream migrating juvenile steelhead (Johnson and Merrick 2012). Fish in the fry stage averaged 34 and 41 mm FL in Deer and Mill, respectively, while those in the parr stage averaged 115 mm FL in both streams. Silvery parr (beginning to smolt) averaged approximately 181 mm, while smolts (fully smolted fish) averaged 210 mm in Deer and 204 mm in Mill Creek. Timing of emigration by silvery parr and smolts was March to May, while fry and parr migration was later (May and June) and then again with the onset of rains in the fall (October through December) (Johnson and Merrick 2012).In contrast to the upper Sacramento River tributaries, Lower American River juvenile steelhead have been shown to smolt at a very large size (270 to 350 mm FL), and nearly all smolt at age-1 (Sogard et al. 2012).

4.A.3.6.5 DPS Viability

All indicators point to a continued decline in abundance of CCV steelhead and an increasing proportion being hatchery propagated (Good *et al.* 2005; NMFS 2011). The static release of hatchery steelhead (numbers/year), coupled with the increasing percentage of hatchery fish, would indicate a continued decline of wild fish that choose anadromy as a benefit to the species survival.

CCV steelhead within the San Joaquin River tributaries show very low overall abundance in spite of recent restoration efforts.

4.A.3.7 Relevant Conservation Efforts

Because Chinook salmon are a commercially important fish and steelhead are not a State listed species, few conservation actions are specific to steelhead. Efforts by the CDFW to restore CCV steelhead are described in *Steelhead Restoration and Management Plan for California* (McEwan and Jackson 1996). Measures to protect steelhead throughout the state of California have been in place since 1998, including 100% marking of all hatchery steelhead, zero bag limits for unmarked steelhead, and gear restrictions designed to protect rearing parr and smolts. The CCV steelhead Project Work Team, an interagency technical working group led by CDFW, drafted a proposal to develop a comprehensive steelhead monitoring plan that was selected by the CALFED Bay-Delta Program (CALFED) Ecosystem Restoration Program Implementing Agency Managers for directed action funding. Long-term funding for implementation of the monitoring plan still needs to be secured.

BiOps for CVP/SWP operations (e.g., NMFS 2009a) and other federal projects involving irrigation and water diversion and fish passage, for example, have improved adverse effects on steelhead in the Central Valley. In 1992, an amendment to the authority of the CVP through the Central Valley Project Improvement Act was enacted to give protection of fish and wildlife equal priority with other Central Valley Project objectives. Several programs under this act have benefited listed salmonids. The USFWS's Anadromous Fish Restoration Program is engaged in monitoring, education, and restoration projects designed to contribute toward doubling the natural populations of select anadromous fish species residing in the Central Valley. Restoration projects funded through the program include fish passage, fish screening, riparian easement, and land acquisition, development of watershed planning groups, instream and riparian habitat improvement, and gravel replenishment. The program combines federal funding with state and private funds to prioritize and construct fish screens on major water diversions. The goal of the Water Acquisition Program is to acquire water supplies to meet the habitat restoration and enhancement goals of the Central Valley Project Improvement Act, and to improve the ability of the U.S. Department of the Interior to meet regulatory water quality requirements. Water has been used to improve fish habitat for CCV steelhead by maintaining or increasing instream flows on Butte and Mill Creeks and the San Joaquin River at critical times. Additionally, salmonid entrainment at the CVP/SWP export facilities is decreased by reducing seasonal diversion rates during periods when protected fish species are vulnerable to export related losses.

Two programs included under CALFED, the Ecosystem Restoration Program and the Environmental Water Account, were created to improve conditions for fish, including steelhead, in the Central Valley. Restoration actions implemented by the Ecosystem Restoration Program 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 on tributary drainages with high potential for CCV steelhead and spring-run Chinook salmon production. Additional ongoing actions include efforts to enhance fishery monitoring and directly support salmonid production through hatchery releases. A major CALFED Ecosystem Restoration Program action currently under way is the Battle Creek Salmon and Steelhead Restoration Project. The project will restore 77 kilometers (48 miles) of habitat in Battle Creek to support steelhead and Chinook salmon spawning and juvenile rearing at a cost of over $90 million. The project includes removal of five small hydropower diversion dams, construction of new fish screens and ladders on another three dams, and construction of several hydropower facility modifications to ensure the continued hydropower operations. It is thought that this restoration effort is the largest cold-water restoration project to date in North America.

Saeltzer Dam on Lower Clear Creek was removed in 2000, thereby opening up approximately 10 miles of stream habitat to anadromous salmonids including steelhead. Since this dam removal, there has been extensive gravel augmentation and regulation of instream flows and water temperatures both as part of the Clear Creek Restoration Program and as required by NMFS' BiOp (2009). This program has been successful in restoring Clear Creek habitat conditions such that the watershed now supports a small but increasing population of steelhead.

Recent conservation actions have improved habitat conditions for Butte Creek steelhead. Completion of the Willow Slough Weir Project (new culverts and a new fish ladder) in 2010 improved fish passage through the Sutter Bypass. In addition, since 2000, real-time coordinated operations of the DeSabla Centerville Project (FERC Project No. 803) have been implemented to reduce the water temperature-related effects of the project on spring-run Chinook salmon adults during the summer, which will also benefit steelhead parr.

The U.S. Army Corps of Engineers initiated a long-term gravel augmentation program in 2010 that is intended to improve spawning habitat in the uppermost reach of the lower Yuba River. Other lower Yuba River habitat restoration actions that are reasonably certain to occur in the next several years include improved fish passage at Daguerre Point Dam (known to have passage problems at high flows), a long-term program to add woody material to the river in an effort to increase habitat complexity, and a riparian enhancement project intended to improve rearing habitat in the short- and long-term. In addition, the FERC re-licensing process for the Yuba River Project is likely to include monitoring studies of *O. mykiss* in the Lower Yuba River over the next five years.

The San Joaquin River Restoration Program (SJRRP) calls for a combination of channel and structural modifications along the San Joaquin River below Friant Dam, releases of water from Friant Dam to the confluence of the Merced River, and the reintroduction of spring-run Chinook salmon. The first flow releases from Friant Dam in support of the SJRRP occurred in October 2009. Though this program is focused on spring-run Chinook salmon, it has the potential to improve habitat for steelhead as well.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements in the Delta. The DRERIP team has created a suite of ecosystem and species conceptual models, including steelhead (Williams 2010), that document existing scientific knowledge of Delta ecosystems. The team has used these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation.

Oroville Facilities Federal Energy Regulatory Commission relicensing efforts on the Feather River have considered instream flows and temperature management for steelhead spawning and juvenile rearing downstream of the dam. However, relicensing is not yet complete.

Multiple fish passage projects have been recently implemented for steelhead and other salmonids in the Sacramento and San Joaquin Watersheds. Multiple large diversions on the Sacramento River (e.g., Glenn-Colusa Irrigation District, Reclamation District 108, Reclamation District 1004, Sutter Mutual, and Wilkins Slough) have been equipped with positive barrier fish screens to reduce entrainment of steelhead and other salmonids. The Woodbridge Irrigation District Dam on the Mokelumne River was designed to improve upstream and downstream passage of steelhead and other salmonids by installing fish screens and fish ladders at the dam.

Mitigation under the Delta Fish Agreement has increased the number of wardens enforcing harvest regulations for steelhead and other fish in the Delta and upstream tributaries by creating the Delta Bay Enhanced Enforcement Program. Initiated in 1994, the program currently consists of nine wardens and a supervisor.

Many smaller tributaries to the Sacramento and San Joaquin Rivers have local watershed conservancies with master plans to contribute to conservation and recovery of steelhead and other salmonids.

4.A.3.8 Recovery Goals

- The recovery plan for Central Valley salmonids, including CV steelhead, was released by NMFS on July 22, 2014. The overarching goal is the removal of, among other listed salmonids, CV steelhead from the federal list of endangered and threatened wildlife (NMFS 2014). Recovery goals usually can be subdivided into discrete component objectives, which, collectively, describe the conditions (criteria) necessary for achieving the goal. Recovery objectives are the parameters of the goal, and criteria are the values for those parameters. For the ESU to achieve recovery, each of the Diversity Groups should support both viable and dependent populations and meet goals for redundancy and distribution. More specifically, to achieve recovery the CV steelhead ESU should display the following characteristics: One population in the Northwestern California Diversity Group at low risk of extinction.

- Two populations in the Basalt and Porous Lava Flow Diversity Group at low risk of extinction Four populations in the Northern Sierra Diversity Group at low risk of extinction

Two populations in the Southern Sierra Diversity Group at low risk of extinction Maintain multiple populations at moderate risk of extinction Criteria for low risk of extinction include a census population size that is >2,500 adults, or has an effective population size that is >500, no productivity decline that is apparent, no catastrophic event that has occurred within the last 10 years, and hatchery influence is at low levels. Criteria for moderate extinction risk include: a census population that is 250 to 2,500 adults, or has an effective population that is 50 to 250 adults, run sizes are <500, but are stable, no apparent decline in populations growth rate that stems from a catastrophic event that has happened in the last 10 years, and hatchery influence is moderate.

4.A.3.9 References

4.A.3.9.1 Written References

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4.A.4 Southern DPS Green Sturgeon (Acipenser medirostris)

4.A.4.1 Introduction

This section provides information on the basic biology, life history, status, and threats and stressors of the southern DPS of North American green sturgeon in the action area.

4.A.4.2 Status

The North American green sturgeon is composed of two distinct population segments (DPSs): the Northern DPS, which includes all populations in the Eel River and northward; and the Southern DPS, which includes all populations south of the Eel River. The Northern DPS currently spawns in the Klamath River in California and the Rogue River in Oregon, and is listed as a Species of Concern (69 *Federal Register* [FR] 19975; April 15, 2004). Only the Southern DPS is found in the action area (Figure 4.A.4-1).

NMFS listed the Southern DPS of North American green sturgeon as threatened under the ESA (71 FR 17757; April 7, 2006). NMFS cited concentration of the only known spawning population into a single river (Sacramento River), loss of historical spawning habitat, mounting threats with regard to maintenance of habitat quality and quantity in the Delta and Sacramento River, and an indication of declining abundance based upon salvage data at the State and Federal salvage facilities. The Southern DPS includes all spawning populations of green sturgeon south of the Eel River (exclusive), principally including the Sacramento River green sturgeon spawning population. Included in the listing are the spawning population in the Sacramento River and fish living in the Sacramento River, the Sacramento–San Joaquin River Delta (Delta), and the San Francisco Estuary.

The primary threat to the Southern DPS is the reduction in habitat and spawning area due to dams (such as Keswick, Shasta, Fish Barrier Dam, and Oroville). Spawning is limited to one population in the Sacramento and Feather Rivers, making green sturgeon highly vulnerable to catastrophic events. Continuing threats include migration barriers, insufficient flow, increased water temperatures, juvenile entrainment in water export facilities, nonnative forage species, competitors, predators, poaching (illegal harvest), and pesticides and heavy metals (Biological Review Team 2005). As long-lived, late maturing fish that spawn periodically, green sturgeon are particularly susceptible to threats from illegal fishing. Green sturgeon had previously been caught in the sport and commercial fisheries in Oregon and Washington, and tribal fisheries which target the northern DPS.

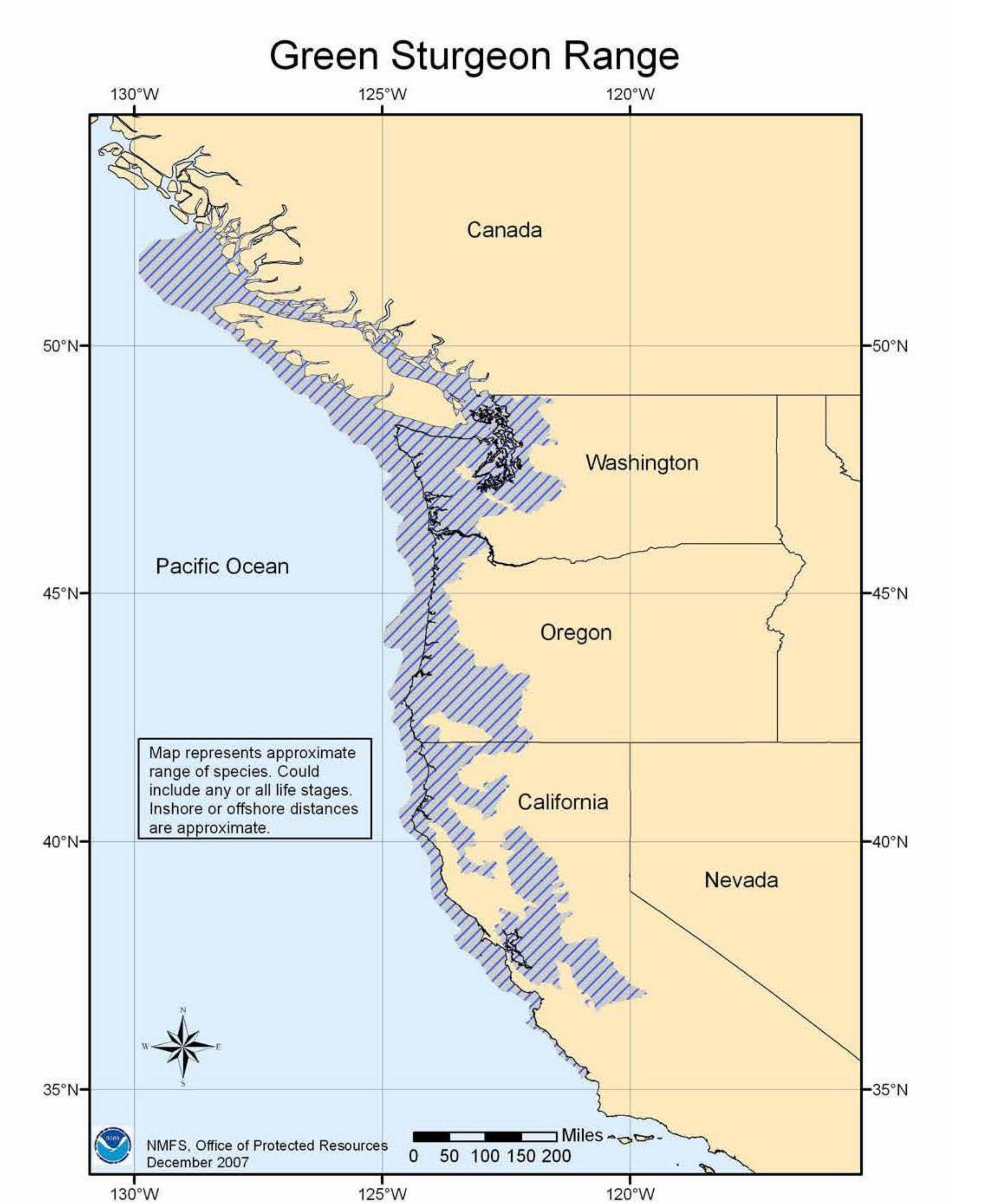

Figure 4.A.4-1. Southern DPS Green Sturgeon Range

On May 21, 2009, NMFS proposed an ESA Section 4(d) rule to apply ESA take prohibitions to the Southern DPS (74 FR 23822). NMFS published the final ESA Section 4(d) rule and protective regulations on June 2, 2010 (75 FR 30714). In California, green sturgeon is a Class 1 Species of Special Concern (qualifying as threatened under the California Endangered Species Act [CESA]) (DFG 2003).

Since the original listing decision, new information has generally reinforced the original reasons for listing Southern DPS, and has reaffirmed NMFS concerns that Southern DPS face substantial threats that challenge their recovery.

4.A.4.3 Critical Habitat

On October 9, 2009, NMFS designated critical habitat for the Southern DPS (74 FR 52300). Critical habitat in marine waters includes areas within the 60-fathom isobath from Monterey Bay to the U.S.-Canada border. Coastal bays and estuaries designated as critical habitat include San Francisco, San Pablo, and Suisun Bays and Humboldt Bay in California; Coos, Winchester, Yaquina, and Nehalem Bays in Oregon; Willapa Bay and Grays Harbor in Washington; and the lower Columbia River Estuary from the mouth to River Kilometer 74. In fresh water, critical habitat includes the mainstem Sacramento River from the Sacramento I-Street Bridge upstream to Keswick Dam (including the Yolo and Sutter Bypasses areas and the lower American River), the Feather River downstream of the Fish Barrier Dam, the Yuba River downstream of the Daguerre Point Dam, and the Delta (Figure 4.A.4-2).

The critical habitat designation identified the following PBFs considered essential for the conservation of the DPS.

1. For freshwater riverine systems:
    a. Food resources. Abundant prey items for larval, juvenile, subadult, and adult life stages.

    b. *Substrate type or size (i.e., structural features of substrates).* Substrates suitable for egg deposition and development (*e.g.*, bedrock sills and shelves, cobble and gravel, or hard clean sand, with interstices or irregular surfaces to ‘‘collect’’ eggs and provide protection from predators, and free of excessive silt and debris that could smother eggs during incubation), larval development (*e.g.*, substrates with interstices or voids providing refuge from predators and from high flow conditions), and subadults and adults (*e.g.*, substrates for holding and spawning).

    c. *Water flow.* A flow regime (*i.e.*, the magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) necessary for normal behavior, growth, and survival of all life stages.

    d. *Water quality.* Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.

    e. *Migratory corridor.* A migratory pathway necessary for the safe and timely passage of Southern DPS within riverine habitats and between riverine and estuarine habitats (*e.g.*, an unobstructed river or dammed river that still allows for safe and timely passage).

f. *Depth.* Deep (≥5 m) holding pools for both upstream and downstream holding of adult or subadult fish, with adequate water quality and flow to maintain the physiological needs of the holding adult or subadult fish.

g. *Sediment quality.* Sediment quality (*i.e.*, chemical characteristics) necessary for normal behavior, growth, and viability of all life stages.

2. For estuarine habitats:

a. Food *resources.* Abundant prey items within estuarine habitats and substrates for juvenile, subadult, and adult life stages.

b. *Water flow.* Within bays and estuaries adjacent to the Sacramento River (*i.e.*, the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds.

c. *Water quality.* Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.

d. *Migratory corridor.* A migratory pathway necessary for the safe and timely passage of Southern DPS within estuarine habitats and between estuarine and riverine or marine habitats.

e. *Depth.* A diversity of depths necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages.

f. *Sediment quality.* Sediment quality (*i.e.*, chemical characteristics) necessary for normal behavior, growth, and viability of all life stages.

3. For nearshore coastal marine areas:

a. *Migratory corridor.* A migratory pathway necessary for the safe and timely passage of Southern DPS within marine and between estuarine and marine habitats.

b. *Water quality.* Nearshore marine waters with adequate dissolved oxygen levels and acceptably low levels of contaminants (*e.g.*, pesticides, organochlorines, elevated levels of heavy metals) that may disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon.

c. *Food resources.* Abundant prey items for subadults and adults, which may in lude benthic invertebrates and fishes.

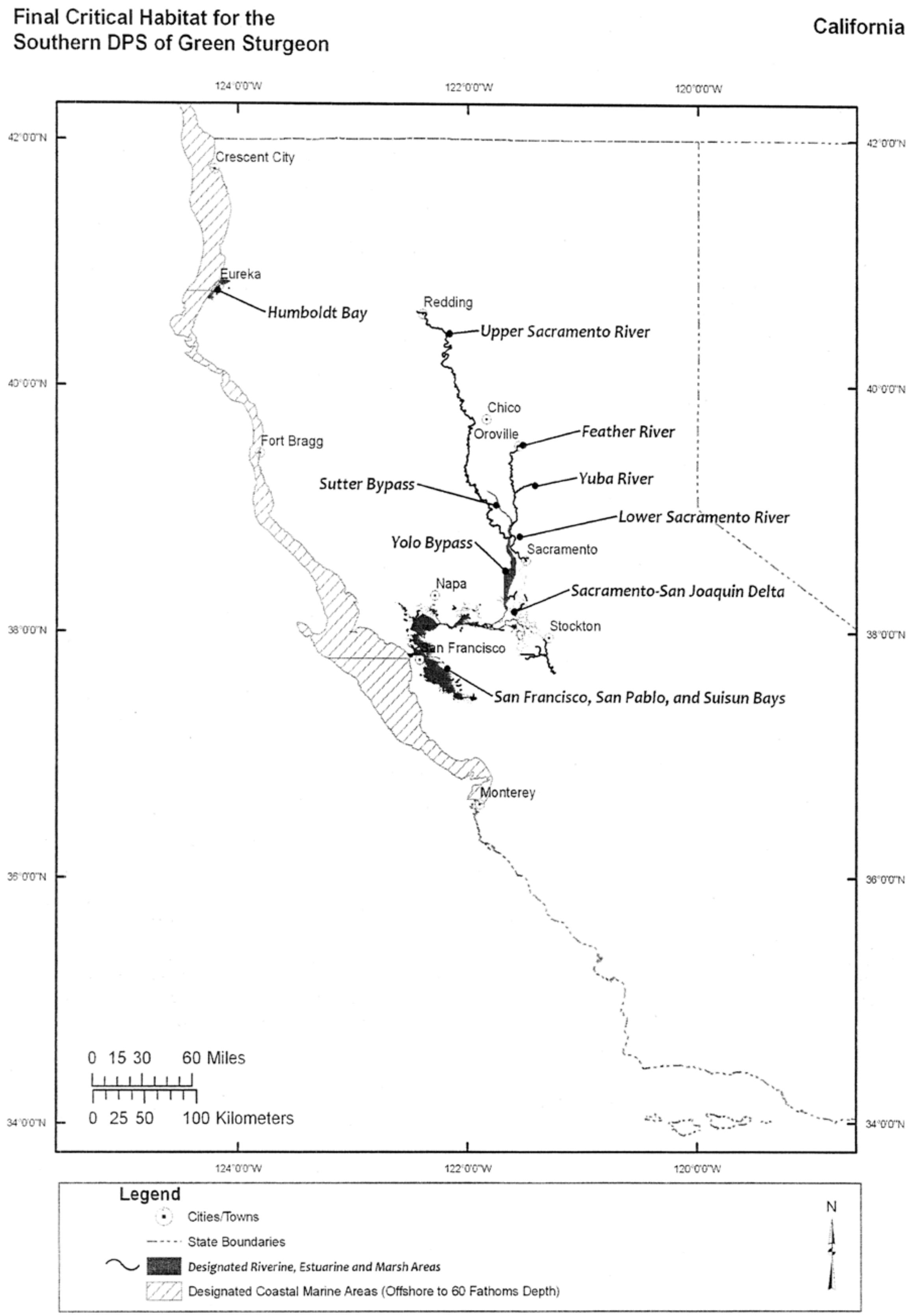

Figure 4.A.4-2. Southern DPS Green Sturgeon Inland Critical Habitat

4.A.4.3.1 Freshwater Riverine Systems

Freshwater habitat of green sturgeon of the Southern DPS varies in function, depending on location in the Sacramento River watershed.

Spawning areas currently are limited to accessible reaches of the Sacramento River upstream of Hamilton City and downstream of ACID Dam (NMFS 2015). From 2008 through 2011, green sturgeon spawning habitat has been identified at seven locations covering a 94 river kilometer reach of the Sacramento River ranging from RK 426 to RK 3325 (Poytress et al. 2012). In addition spawning has been confirmed in the lower Feather River (Seesholtz et al. 2012). Preferred spawning habitats are thought to contain large cobble in deep and cool pools with turbulent water (DFG 2002; Moyle 2002; Adams et al. 2002). Sufficient flows are needed to oxygenate and limit disease and fungal infection of recently laid eggs (Deng et al. 2002; Parsley et al. 2002). In the Sacramento River, spawning appears to be triggered by large increases in water flow (Brown and Michniuk 2007).

Acoustic tagging studies by Erickson et al. (2002) in the Rogue River (Northern DPS green sturgeon) showed adult green sturgeon holding for as long as six months in deep (greater than 5 meters [16 feet]), low-gradient reaches or off-channel sloughs or coves of the river during summer months when water temperatures were between 15 and 23°C (59 and 73.5°F). When ambient temperatures in the river dropped in fall and early winter (less than 10°C [50°F]) and flows increased, fish moved downstream and into the ocean. Water temperatures in spawning and egg incubation areas are critical; temperatures greater than 19°C (66.2°F) are lethal to green sturgeon embryos (Cech et al. 2000; Mayfield and Cech 2004; Van Eenennaam et al. 2005; Allen et al. 2006).

Habitats for migration are downstream of spawning areas and include the mainstem Sacramento River, Delta, Suisun, and San Pablo Bays. These corridors allow the upstream passage of adults and the downstream emigration of juveniles (71 FR 17757; April 7, 2006). Migratory habitat conditions are strongly affected by the presence of barriers and impediments to migration (e.g., dams), unscreened or poorly screened diversions, and degraded water quality. One of the key areas of concern is the Yolo and Sutter bypasses. Adult sturgeon migrating upstream are attracted into the bypasses by high flows, but weirs can act as barriers and block the passage of fish. Fish can also be trapped in the bypasses as floodwaters recede (USFWS 1995, DWR 2005c). Irregularities in the splash basins at the foot of weirs, coupled with multiple road crossings and agricultural impoundments block hydraulic connectivity and can impede fish passage. The result is sturgeon stranding in the bypasses, which results in delayed migration and renders them highly susceptible to poaching, high water temperatures, low DO, and desiccation.

Heublein et al. (2009) found two different patterns of spawning migration and out-migration for green sturgeon in the Sacramento River. Results of this study found six individuals potentially spawned, over-summered, and moved out of the river with the first fall flow event; this pattern is thought to be the common behavior of green sturgeon. Alternatively, nine individuals promptly moved out of the Sacramento River before September 1 without any known flow or temperature cue. Both spawning areas and migratory corridors comprise rearing habitat for juvenile green sturgeon, which feed and grow up to 3 years in fresh water. Stomach contents from adult and juvenile green sturgeon captured in the Delta point to the importance of habitat that supports shrimp, mollusks, amphipods, and small fish (Radtke 1966; Houston 1988; Moyle et al. 1992). Rearing habitat condition and function may be affected by variation in annual and seasonal flow and water temperatures (71 FR 17757; April 7, 2006). Habitats should contain sediment of the appropriate quality and characteristics necessary for normal behavior, growth, and viability of all life stages. Sediments should be free of contaminants, elevated levels of heavy metals, polycyclic aromatic hydrocarbons (PAHs), and organochlorine pesticides that can result in negative effects on any life stage of green sturgeon or their prey. It is thought that bioaccumulation of contaminants from feeding on benthic species may negatively affect the growth, reproductive development, and reproductive success of green sturgeon

4.A.4.3.2 Estuarine Habitats

Estuaries should contain abundant food items including benthic invertebrates and fish. These may include crangonid shrimp, callianassid shrimp, burrowing thalassinidean shrimp, amphipods, isopods, clams, annelid worms, crabs, sand lances, herring eggs, and anchovies. These food items are considered essential for rearing habitat that promotes growth and development of juvenile, subadult, and adult green sturgeon within bays and estuaries.

Within the bays and estuaries adjacent to the Sacramento River system there should be sufficient flow as to allow proper migration cues for adult green sturgeon to move upstream into the Sacramento River and onto the spawning grounds.

To promote the species viability water quality, which includes temperature, salinity, oxygen content, and other chemical characteristics should be adequate in all life stages of Green Sturgeon.

Unobstructed migratory pathways are necessary for the successful and timely passage of adult, sub-adult, and juvenile sturgeon. Green sturgeon should have the ability to freely migrate from the river through the estuarine waterways of the delta and bays and eventually out into the ocean.

Depth of water is important in that a diversity of depths is needed for shelter, foraging, and migration of juvenile, subadult, and adult life stages. Deep holding pools may be important for feeding and energy conservation, or may serve as thermal refugia (Benson *et al.* 2007). Kelly et al. (2007) found that green sturgeon adults and subadults occupied water less than 10 meters deep in San Francisco Bay Estuary, swimming either near the surface or along the bottom. Juveniles within the Sacramento-San Joaquin River Delta have been captured primarily in waters from 3–8 feet deep, which may indicate a preference for shallower water then subadults and adults (Radtke 1966). Sediments should have the same qualities as listed above for Riverine Systems.

4.A.4.3.3 Nearshore Coastal Marine Waters

A migratory pathway is necessary for the safe and timely passage of Southern DPS within marine and between estuarine and marine habitats. Unimpeded passage within coastal marine waters is critical for subadult and adult green sturgeon to access over summering habitats within coastal bays and estuaries and overwintering habitat within coastal waters between Vancouver Island, BC, and southeast Alaska. To summarize, no human induced impediments, either physical, chemical or biological, that may alter the migratory behavior of the fish such that its survival or the overall viability of the species is compromised.

The water quality of coastal marine waters must have adequate dissolved oxygen and must have acceptable low levels of contaminants (see riverine systems) that do not disrupt the normal behavior, growth, and viability of subadult and adult green sturgeon. Based on studies of tagged subadult and adult green sturgeon may need a minimum dissolved oxygen level of at least 6.54 mg O2/l (Kelly et al., 2007; Moser and Lindley 2007).

Green sturgeon spend more than half their lives in coastal marine and estuarine waters, spending from 3–20 years at a time out at sea. Abundant food resources are important to support subadults and adults over long-distance migrations, and may be one of the factors attracting green sturgeon to habitats far to the north. Prey species are likely similar to those in bays and estuaries.

4.A.4.4 Life History

4.A.4.4.1 Immigration and Holding

Adult green sturgeon begin their upstream spawning migrations into the San Francisco Bay in March, reach Knights Landing during April, and spawn between March and July (Heublein 2006). Heublein et al. (2009) found two different patterns of spawning migration and out-migration for green sturgeon in the Sacramento River. Results of this study found six individuals potentially spawned, over-summered, and moved out of the river with the first fall flow event; this pattern is thought to be the common behavior of green sturgeon. Alternatively, nine individuals promptly moved out of the Sacramento River before September 1 without any known flow or temperature cue.

4.A.4.4.2 Spawning

Adult North American green sturgeon are believed to spawn every 3 to 5 years, but can spawn as frequently as every 2 years (NMFS 2005) and reach sexual maturity at an age of 15 to 20 years, with males maturing earlier than females. Adult green sturgeon begin their upstream spawning migrations into the San Francisco Bay in March, reach Knights Landing during April, and spawn between March and July (Heublein 2006). Based on the distribution of sturgeon eggs, larvae, and juveniles in the Sacramento River, CDFW (DFG 2002) concluded that green sturgeon spawn in late spring and early summer upstream of Hamilton City, and possibly to Keswick Dam. Peak spawning is believed to occur between April and June. Females deposit eggs close to the substrate at sites where they quickly sink in between large rock substrate. The large size of green sturgeon eggs relative to other sturgeon indicates that female green sturgeon invest a greater amount of their reproductive energy resources into maternal yolk for nourishment of the embryo, which results in larger larvae (Van Eenennaam et al. 2001). The reserve of maternal yolk and larger larvae could provide an advantage in larval feeding and survival (Van Eenennaam et al. 2001). Compared with other acipenserids, green sturgeon larvae appear more robust and easier to rear (Van Eenennaam et al. 2001).

Similar to winter-run Chinook salmon, the Southern DPS has been relegated to spawning in a single area just below Keswick and Shasta Dams, which have made historical spawning areas inaccessible (Lindley et al. 2004; Adams et al. 2007). Current data and observations document green sturgeon in the Sacramento River as far upstream as Keswick Dam and as far south as the CVP/SWP water export facilities near the southern limit of the Sacramento-San Joaquin Delta.

Spawning in the upper Sacramento River is currently thought to occur from Hamilton City (River Mile [RM] 200) to above Ink's Creek at RM 426 (Poytress et al. 2012). Spawning migrations and spawning by green sturgeon in the upper Sacramento River mainstem have been well documented over the last 15 years (Beamesderfer et al. 2004). Anglers fishing for white sturgeon or salmon commonly report catches of green sturgeon from the Sacramento River at least as far upstream as Hamilton City (Beamesderfer et al. 2004). Eggs, larvae, and post larval green sturgeon are now commonly reported in sampling directed at green sturgeon and other species (Beamesderfer et al. 2004; Brown 2007). Young-of-the-year (yoy) green sturgeon have been observed annually since the late 1980s in fish sampling efforts at Red Bluff Diversion Dam (RBDD) and the Glenn-Colusa Canal (Beamesderfer et al. 2004). Green sturgeon have not been documented in Sacramento River tributaries other than the Feather River system (Beamesderfer et al. 2004, Moyle 2002).

Documented historical and current spawning occurs in the Sacramento River (Adams et al. 2002; Beamesderfer et al. 2004; Adams et al. 2007). Currently, ACID, Keswick, and Shasta dams on the mainstem of the Sacramento River are barriers to the upper river. Although no historical accounts exist for identified green sturgeon spawning occurring above the current dam sites, suitable spawning habitat likely existed. The upstream extent of historical spawning by green sturgeon in the Sacramento River system is unknown. White sturgeon historically ranged into upper portions of the Sacramento system including the Pit River and a substantial number were trapped in and above Lake Shasta when Shasta Dam was closed in 1944 and successfully reproduced until the early 1960s (Beamesderfer et al. 2004). Green sturgeon have not been documented upstream from the Shasta Dam site. According to NMFS (2005), "the BRT considered it possible that the additional habitat behind Shasta Dam in the Pit, McCloud, and Little Sacramento systems would have supported separate populations or at least a single, larger Sacramento River population less vulnerable to catastrophes than one confined to a single mainstem, but the BRT was unable to be specific due to the paucity of historical information" (NMFS 2005).

Historical and recent information confirms that both green and white sturgeons occasionally range into the Feather, Yuba, and Bear rivers but numbers are low (Beamesderfer et al. 2004). It is unknown whether green sturgeon historically spawned in the Feather River either downstream or upstream of Oroville Dam or the Thermalito Afterbay outlet. Spawning is suspected to have occurred in the past due to the continued presence of adult green sturgeon in the river below Fish Barrier Dam. This continued presence of adults below the dam suggests that fish are trying to migrate to upstream spawning areas now blocked by the dam, which was constructed in 1968. Unspecific historical reports of green sturgeon spawning in the Feather River (Wang 1986, USFWS 1995a, DFG 2002, DWR 2007) have not been corroborated by observations of young fish caught in screw traps (Beamesderfer et al. 2004). Spawning has recently been recorded with eggs from three different sturgeon females (Van Eenenaam 2011). In spring 2011, many sturgeon adults were spotted while DIDSON surveys were being conducted (Seesholtz 2011). Significant habitat on the Lower Feather River, while modified, remains accessible downstream from the Thermalito Afterbay outlet (DWR 2005a). Man-made barriers (Sunset Pumps) to upstream movements in the Feather River during low flow years might also limit significant movement of Southern DPS green or white sturgeon into the Feather River to higher flow water years (Beamesderfer et al. 2004).

The current or historical occurrence of green sturgeon in the San Joaquin River has been a source of much speculation. It is unclear whether green sturgeon were historically present, are currently present, or were historically present and have been extirpated from the San Joaquin River (NMFS 2005, Beamesderfer et al. 2007). No juvenile green sturgeon have been documented in the San Joaquin River although no directed sturgeon studies have ever been undertaken in the San Joaquin River (USFWS 1995a, DFG 2002, Adams et al. 2002, Beamesderfer et al. 2004, NMFS 2005). Observations of green sturgeon juveniles or unidentified sturgeon larvae in the San Joaquin River has been limited to the Delta where they could easily, and most likely, have originated from the Sacramento River rather than the San Joaquin River (Beamesderfer et al. 2004). Moyle (2002) suggested that reproduction may have taken place in the San Joaquin River because adults have been captured at Santa Clara Shoal and Brannan Island. However, given the conditions that exist in the San Joaquin River today, they are probably extirpated (Israel and Klimley 2008).

4.A.4.4.3 Egg to Larvae

Adult female green sturgeon produce between 59,000 and 242,000 eggs, depending on body size, with a mean egg diameter of 4.3 millimeters (0.17 inch) (Moyle et al. 1992; Van Eenennaam et al. 2006). Life stages are summarized in Table 4.A.4-1 and occurrence is mapped out in Table 4.A.4-2.

Green sturgeon larvae hatch from after approximately 7 days at a water temperature of 15°C (Van Eenennaam et al. 2001, Deng et al. 2002), which is similar to the rate of white sturgeon development. Newly hatched green sturgeon are approximately 12.5 to 14.5 millimeters (0.5 to 0.57 inch) long and have a large ovoid yolk sac that supplies nutritional energy until exogenous feeding occurs. Green sturgeon larvae do not exhibit the initial pelagic swim–up behavior characteristic of other *Acipenseridae*. Hatchling green sturgeon embryos are weak swimmers and seek nearby (a few cm) cover, and remain under rocks (Deng et al. 2002). Early yolk-sac larvae resemble a “tadpole” with a continuous fin fold both dorsally and ventrally, with well-developed eyes, but a poorly developed mouth and respiratory structures. Green sturgeon are strongly oriented to the river bottom and exhibit nocturnal activity patterns (Cech et al. 2000). After six days, the larvae exhibit nocturnal swim-up activity (Deng et al. 2002). After about 10 days they begin nocturnal downstream migrational movements (Kynard et al. 2005).

Table 4.A.4-1. Southern DPS Green Sturgeon Life Stages in the Action Area.

River	Life Stage	Start Month	End Month	Reference
Upper Sacramento	Migrant	January	December	National Marine Fisheries Service 2009
	Adult Migration	February	June	Heublein *et al.* 2009; Bureau of Reclamation 2008; DFG 2002
	Adult river holding	March	December	Israel and Klimley 2008 (inferred from spawning timing)
	Adult post-spawn emigration	April	January	Heublein *et al.* 2009 (inferred from spawning timing)
	Eggs	March	July	National Marine Fisheries Service 2009; Poytress et al 2009-12
		March	June	Bureau of Reclamation 2008
		April	July	Israel and Klimley 2008
	Larvae, post-larvae	May	October	National Marine Fisheries Service 2009; Poytress et al 2014; DFG 2002
		May	October	Bureau of Reclamation 2008
		May	October	Israel and Klimley 2008
South Delta	Older juvenile >10 months	January	December	National Marine Fisheries Service 2009
Delta	Older juvenile >10 months	January	December	National Marine Fisheries Service 2009
		April	October	National Marine Fisheries Service 2009
Suisun Bay	Older juvenile >10 months	January	December	National Marine Fisheries Service 2009
Feather	Adult immigration	February	June	Seesholtz 2011; Healey and Vincik 2011, (Sac as surrogate)
	Spawning, egg incubation	March	July	Seesholtz 2011, (Sac as surrogate)
	Pre and post spawn holding	April	January	Sac as surrogate; (Israel and Klimley 2008
	Post-spawn emigration	April	January	Sac as surrogate
	Larval to Juvenile rearing & emigration	Year round		Sac as surrogate (NMFS 2009)

Table 4.A.4-2. The Temporal Occurrence of Adult (a) and Juvenile (b) Green Sturgeon in the Sacramento and Feather Rivers. Darker shades indicate months of greatest relative abundance.

Green sturgeon relative abundance	High				Medium				Low			
Sacramento River												
a) Spawning Adults												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Immigration; Hamilton City, Verona[a]												
Spawning, egg incubation; Bend Bridge, RBDD, Hamilton City[b]												
Pre- and post-spawn adult holding; Bend Bridge, RBDD, Hamilton City[c]												
Post-spawn emigration; Bend Bridge, RBDD, Hamilton City[d]												
b) Juvenile emigration												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Larval to Juvenile rearing & emigration[e]												
Feather River												
a) Spawning Adults												
Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Immigration;Feather at Sac confluence[f]												
Spawning, egg incubation; Thermalito Afterbay outlet,Gridley[f]												
Pre- and post-spawn adult holding; Thermalito Afterbay outlet,Gridley[f]												
Post-spawn emigration; Thermalito Afterbay outlet,Gridley[f]												
b) Juvenile emigration												
Larval to Juvenile rearing & emigration; Thermalito Afterbay outlet,Sac confluence[f]												

[a] Miller 1972, DFG 2002;[b] Poytress et al 09-12, Brown 2002, DFG 2002;[c] Isreal and Klimley 2008, inferred from spawn timing;[d] Heublein et al 2009, inferred from spawn timing;[e] Poytress et al 2014, DFG 2002;[f] Sac River timing as a surrogate for relative timing within life-stages in Feather

4.A.4.4.4 Larvae Migration

Juvenile green sturgeon continue to exhibit nocturnal behavior beyond the metamorphosis from larval to juvenile stages. After approximately 10 days, larvae begin feeding and growing rapidly, and young green sturgeon appear to rear for the first 1 to 2 months in the upper Sacramento River between Keswick Dam and Hamilton City (DFG 2002). Length measurements estimate juveniles to be 2 weeks old (24 to 34 millimeters [0.95 to 1.34 inch] fork length) when they are captured at the Red Bluff Diversion Dam (DFG 2002; USFWS 2002), and three weeks old when captured further downstream at the Glenn-Colusa facility (Van Eenennaam et al. 2001). Growth is rapid as juveniles reach up to 30 centimeters (11.8 inches) the first year and over 60 centimeters (24 inches) in the first 2 to 3 years (Nakamoto et al. 1995).

4.A.4.4.5 Esturarine and Delta Behavior

Juveniles spend 1 to 4 years in freshwater and estuarine habitats before they enter the ocean (Nakamoto et al. 1995). According to Heublein et al. (2009), in 2006 all tagged adult green sturgeon emigrated from the Sacramento River prior to September. Lindley et al. (2008) found frequent large-scale migrations of green sturgeon along the Pacific Coast. Kelly et al. (2007) reported that green sturgeon enter the San Francisco Estuary during the spring and remain until fall. Juvenile and adult green sturgeon enter coastal marine waters after making significant long-distance migrations with distinct directionality thought to be related to resource availability.

Stomach contents from adult and juvenile green sturgeon captured in the Sacramento-San Joaquin Delta point to the importance of habitat that supports shrimp, mollusks, amphipods, and small fish (Radtke 1966; Houston 1988; Moyle et al. 1992). Stomachs of green sturgeon caught in Suisun Bay contained *Corophium* sp. (amphipod), *Cragon franciscorum* (bay shrimp), *Neomysis awatchensis* (Opossum shrimp: synonymous with *Neomysis mercedis*) and annelid worms (Ganssle 1966). Stomachs of green sturgeon caught in San Pablo Bay contained *C. franciscorum*, *Macoma* sp. (clam), *Photis californica* (amphipod), *Corophium* sp., *Synidotea laticauda* (isopod), and unidentified crab and fish (Ganssle 1966). Stomachs of green sturgeons caught in Delta contained *Corophium* sp. And *N. awatchensis* (Radtke 1966). As a result of recent changes in the species composition of macroinvertebrates inhabiting the Bay-Delta estuary due to non-native species introductions, the current diet of green sturgeon is likely to differ from that reported in the 1960's.

4.A.4.4.6 Ocean Behavior

In the ocean green sturgeon primarily move northward and commingle with other sturgeon populations, spending much of their lives in the ocean or in Oregon and Washington estuaries (DFG 2002; Kelly et al. 2007).

Green sturgeon are known to range in nearshore marine waters from Mexico to the Bering Sea, with a general tendency to head North after their out-migration from freshwater (NMFS 2005). They are commonly observed in bays and estuaries along the western coast of North America during the late summer and early fall (Emmett et al. 1991; Moyle et al. 1992; Israel et al. 2004; Moser and Lindley 2007; Lindley et al. 2008). Both the Northern DPS green sturgeon and

Southern DPS occur in large numbers in the Columbia River estuary, Willapa Bay, and Grays Harbor, Washington (NMFS 2005).

Subadult and adult sturgeon tagged in San Pablo Bay over summer in bays and estuaries along the coast of California, Oregon, and Washington, between Monterey Bay and Willapa Bay, before moving further north in the fall to overwinter north of Vancouver Island. Individual Southern DPS tagged by the DFG in the San Francisco Estuary have been recaptured off Santa Cruz, California; in Winchester Bay on the southern Oregon coast; at the mouth of the Columbia River; and in Gray's Harbor, Washington (Moyle 2002). Most tags for Southern DPS tagged in the San Francisco Estuary have been returned from outside that estuary (Moyle 2002).

Lindley et al. (2008, 2011) investigated marine migrations of green sturgeon by tagging subadults and adults from northern and Southern DPSs with ultrasonic pinger tags. An array of receivers off the coast of California, Oregon, Washington, British Columbia and Alaska tracked their northern and southern migrations. Most tagged sturgeon moved north along the coast in the fall to spend winters north of Vancouver Island and south of southeast Alaska, and returned in the spring to oversummer in California, Oregon and Washington bays and estuaries. Distribution patterns of fish from different tagging locations varied. Moving north instead of south in the autumn may be advantageous bio-energetically to migrating green sturgeon. The predominate current (Davidson) direction and velocity (10 km d^{-1}) is in the northern direction. This may be advantageous given that average migrations distances are 40 km d^{-1} (Huff et al. 2012; Lindley et al. 2008). Green sturgeon from all spawning populations appear to migrate north as far as Brooks Peninsula but vary in the extent of their southerly spring migrations (Lindley et al. 2008). Marine migrations of green sturgeon may include areas as far south as Monterey Bay and as far north as Brooks Peninsula, Vancouver, BC, but their consistently inhabited range is considerably smaller, ranging North from the vicinity of San Francisco and Monterey Bays and primarily concentrated in the coastal waters of Washington, Oregon and Vancouver Island inside the 200m isobath (Huff et al. 2012). For green sturgeon low temperature may be an important factor limiting the northern extent of their range from extending into the Bering Sea (Huff et al. 2012). Alternative explanations include abundant food and refuge from predators (sharks and pinnipeds) and that dissolved oxygen levels may be too low for green sturgeon in the extreme south (Huff et al. 2012).

Based on their life history, a large percentage of the adult green sturgeon population inhabit the ocean at any given time (Beamesderfer et al. 2007). Green sturgeon typically stay near shore and avoid depths exceeding 100 m (Erickson and Hightower 2007). Relatively large concentrations of sturgeon occur in the Columbia River estuary, Willapa Bay, and Grays Harbor, with smaller aggregations in the San Francisco estuary and other coastal estuaries (Emmett et al. 1991; Moyle et al. 1992; Israel et al. 2004; Moser and Lindley 2007; Lindley et al. 2008). Little is known about juvenile and adult green sturgeon feeding and diet in the ocean. On entering the highly productive ocean environment, green sturgeon grow at a rate of approximately 7 centimeters (2.76 inches) per year until they reach maturity. Male green sturgeon mature at an earlier age and are smaller than females (Van Eenennaam et al. 2006). Green sturgeon spend 3 to 13 years in the ocean before returning to fresh water to spawn.

4.A.4.4.7 Status and Trends

There is relatively little known about the abundance of North American green sturgeon, particularly for those that spawn in the Sacramento River (The Nature Conservancy et al. 2008). In the Sacramento River, the green sturgeon population is believed to have declined over the last two decades, with current spawning run size estimated to be in the hundreds (Biotelemetry Laboratory 2014). In the Feather and Yuba Rivers, green sturgeon sightings are extremely limited. Spawning in these watersheds is rarely recorded, although spawning in the Feather River was documented in 2011 (Seesholtz et al 2012). In the San Joaquin River, the green sturgeon population appears to be extirpated (Figure 4.A.4-3).

Green sturgeon juveniles, subadults, and adults are widely distributed in the Sacramento-San Joaquin Delta and estuary areas including San Pablo Bay (Beamesderfer et al. 2004). The Sacramento-San Joaquin Delta serves as a migratory corridor, feeding area, and juvenile rearing area for North American green sturgeon in the Southern DPS. Adults migrate upstream primarily through the western edge of the Delta into the lower Sacramento River between March and June (Adams et al. 2002). Larvae and post-larvae are present in the lower Sacramento River and North Delta between May and October, primarily in June and July (DFG 2002). Juvenile green sturgeon have been captured in the Delta during all months of the year (Borthwick et al. 1999; DFG 2002). Catches of 1 and 2 year old Southern DPS on the shoals in the lower San Joaquin River, at the CVP/SWP fish salvage facilities, and in Suisun and San Pablo bays indicate that some fish rear in the estuary for at least 2 years (DFG 2002). Larger juvenile and subadult green sturgeon occur throughout the estuary, possibly temporarily, after spending time in the ocean (DFG 2002; Kelly et al. 2007). Figure 4.A.4-3 shows the size distribution of green sturgeon at various life stages observed in sample data from young-of-the-year collected in spring and summer at RBDD in the Sacramento River, juveniles salvaged from CVP/SWP water projects, and subadults sampled by DFG in San Pablo Bay. Adult green sturgeon have been documented in the Yolo Bypass, but these individuals usually end up stranded against the Fremont Weir (Thomas et al. 2013) and if not rescued could have population effects.

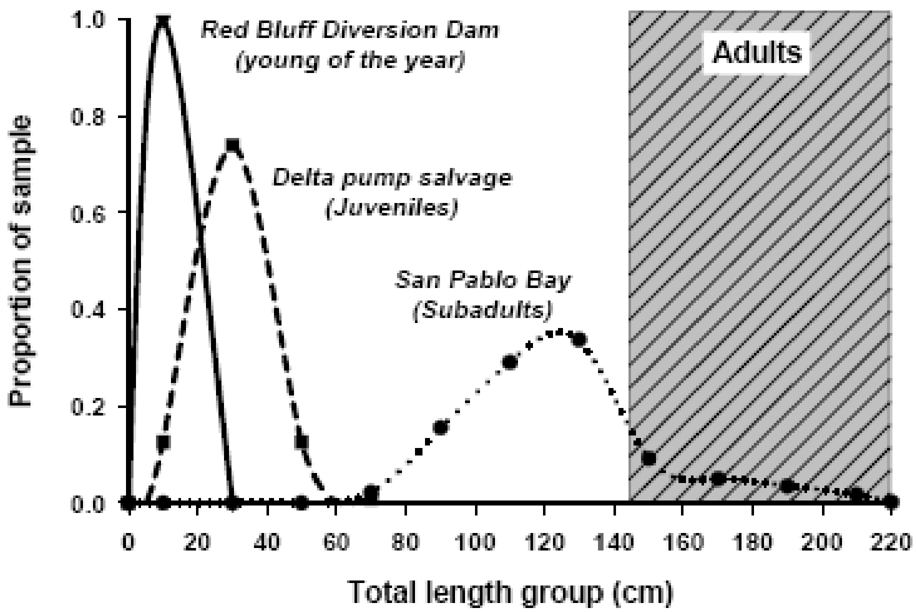

Figure 4.A.4-3. Sizes of Juvenile Green Sturgeon Measured at CVP/SWP Fish Salvage Facilities, 1968–2001 (DFG 2002), Collected in Rotary 1994–2000 (USFWS 2002), and Sampled in Semi-annual San Pablo Bay Sturgeon Stock Assessments (DFG 2002) (Figure from Beamesderfer et al. 2007).

4.A.4.4.8 Abundance

Empirical estimates of green sturgeon abundance are not available for any west coast population including the Sacramento River population. Interpretations of available time series of abundance index data for green sturgeon are confounded by small sample sizes, intermittent reporting, fishery-dependent data, lack of directed sampling, subsamples representing only a portion of the population, and potential confusion with white sturgeon (Adams et al. 2002). This section summarizes the best available data and identifies qualifications to be considered in its application as a description of the current baseline.

The current population status of Southern DPS is unknown (Beamesderfer et al. 2007, Adams et al. 2007). It is believed, based on captures of green sturgeon during surveys for the sympatric white sturgeon in the San Francisco Bay estuary that the population is relatively small (USFWS 1995a), ranging from several hundred to a few thousand adults. Musick et al. (2000) noted that the abundance of North American green sturgeon populations has declined by 88% throughout much of its range. The most consistent sample data for Sacramento green sturgeon is for subadults captured in San Pablo Bay during periodic white sturgeon assessments since 1948. DFG measured and identified 15,901 sturgeon of both species between 1954 and 1991 (USFWS 1995b). California Department of Fish and Wildlife (CDFW) (DFG 2002) estimated that green sturgeon abundance in the Bay-Delta estuary (generally defined as the San Francisco Bay and the Sacramento River-San Joaquin River Delta) ranged from 175 to more than 8,000 adults between 1954 and 2001 with an annual average of 1,509 adults. Using CDFW angler report card reports, the number of green sturgeon caught from 2006 to 2011 ranged from 89 to 311 (Gleason et al. 2008; DuBois et al. 2009, 2010, 2011, 2012). Various attempts have been made to infer green sturgeon abundance based on white sturgeon mark-recapture estimates and relative numbers of white and green sturgeon in the catch (USFWS 1995b, Moyle 2002). However, low catches of green sturgeon preclude estimates or indices of green sturgeon abundance from this data (Schaffter and Kohlhorst 1999, Gingras 2005, as cited in Beamesderfer et al. 2007). It is unclear if the high annual variability in length distributions in these samples (Figure 4.A.4-4) reflects variable recruitment and abundance or is an artifact of small sample sizes, pooling of sample years, or variable distribution patterns between fresh water and ocean portions of the population.

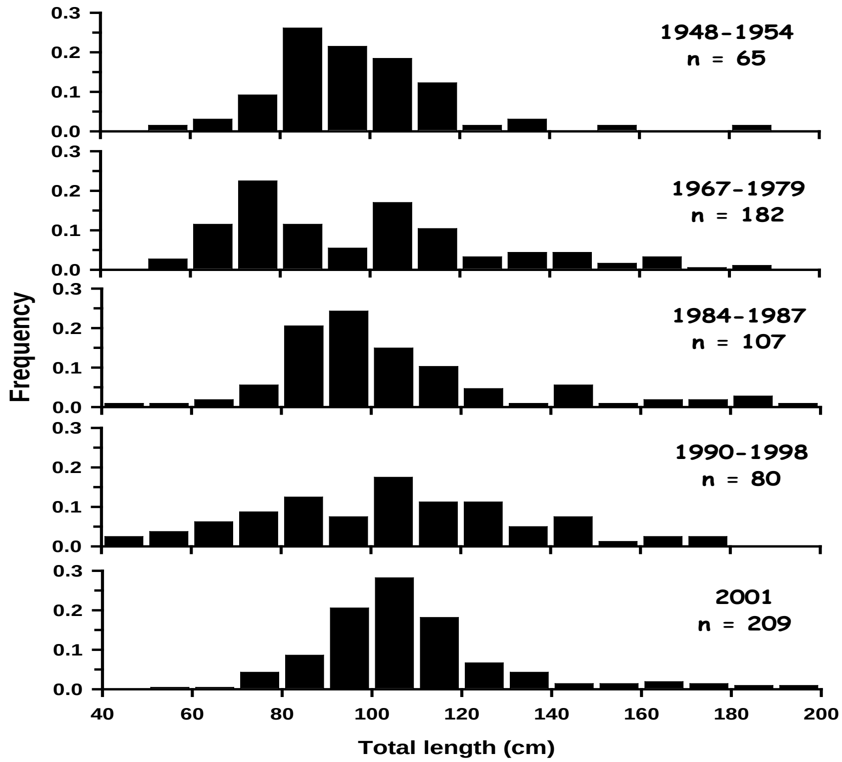

Figure 4.A.4-4. Changes in Length Distribution Over Time Based on Trammel Net Sampling of Subadult Green Sturgeon in San Pablo Bay (DFG 2002) (Figure from Beamesderfer et al. 2007)

Anecdotal information is also available on young-of-the-year green sturgeon from juvenile fish monitoring efforts at RBDD and the Glenn-Colusa Irrigation District pumping facility on the upper Sacramento River. Fish traps have been operated below RBDD and at the Glenn-Colusa Irrigation District (GCID) pumping plant. These facilities report sampling of between zero and 2,068 juvenile green sturgeon per year (Adams et al. 2002).

Approximately 3,000 juvenile green sturgeon have been observed in rotary screw traps operated for juvenile salmon at RBDD from 1994–2000 (Figure 4.A.4-5), through catch of Green Sturgeon was highly variable, not normally distributed and ranged between 0 and 3,701 per year (median = 193) (Poytress et al. 2014). Annual catches of juvenile green sturgeon production have declined over the period from 1995 through 2000 although the relationship of these catches to actual abundance is unknown. Recent data indicate that very little production took place in 2007 and 2008 (13 and 3 larval green sturgeon captured in the RST monitoring sites at RBDD, respectively (Poytress et al. 2014). Larger production was recorded in 2009, 2010, 2011, and none in 2012 (45, 122, 643, and 0 larvae were captured using a benthic D-net; Poytress et al. 2010, Poytress et al. 2011, Poytress et al. 2012, and Poytress et al. 2013). Over 2,000 juvenile green sturgeon have been collected in fyke and rotary screw traps operated at the GCID Diversion from 1986–2003 (Figure 4.A.4-6). Operation of the screw trap at the GCID site began in 1991 and has continued year-around with the exception of 1998. Juvenile green sturgeon at the GCID site were consistently larger in average size, but do not show the same general increase in size over the sampling season as observed at RBDD, which may indicate less favorable growing conditions in the river between RBDD and GCID (DFG 2002). The number captured varied widely (0 to 2,068 per year) with no apparent patterns in abundance between the two sites. Abundance of juveniles peaked during June and July with a slightly earlier peak at the RBDD site (Adams et al. 2002).

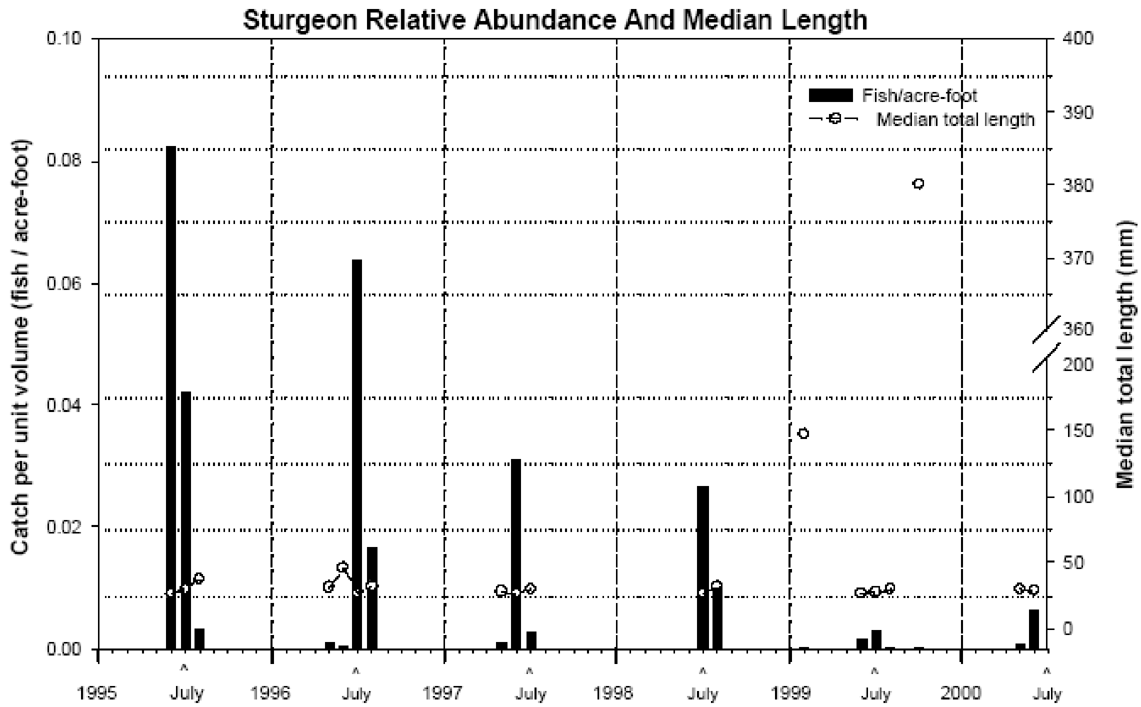

Figure A3. Relative abundance (fish/acre-foot) and median length for juvenile sturgeon (*Acipenser spp.*) captured by rotary-screw traps at Red Bluff Diversion Dam (RK 391), Sacramento River, CA. Data summarized from January 1995 through June 2000. In 1996 and 1997, a total of 124 juvenile sturgeon (*Acipenser spp.*) were grown out and positively identified as green sturgeon (*Acipenser medirostris*).

Figure 4.A.4-5. Green Sturgeon Sample Data from Red Bluff Diversion Dam Rotary Screw Trap Monitoring (USFWS 2002)

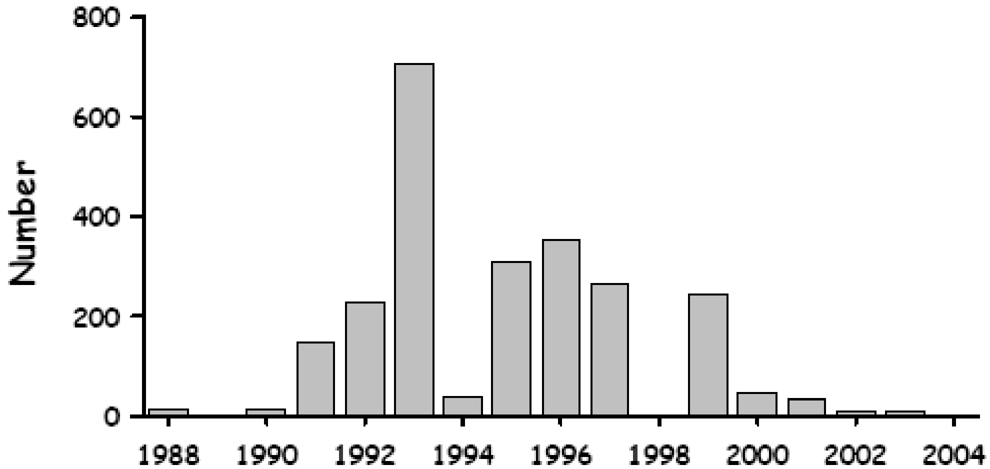

Figure 4.A.4-6. Juvenile Green Sturgeon Collected in Fyke and Rotary Screw Traps Operated at the Glenn-Colusa Irrigation District Diversion from 1986–2003 (Beamesderfer 2005)

Variable numbers of juvenile green sturgeon are observed each year from two south Delta water diversion facilities and provide some of the only information available on the changes in green sturgeon abundance (DFG 2002). When water is exported through the CVP/SWP export facilities, fish become entrained into the diversion. Since 1957, Reclamation has salvaged fish at the Tracy Fish Collection Facility. DFG's Fish Facilities Unit, in cooperation with DWR, began salvaging fish at the Skinner Delta Fish Protective Facility in 1968. The salvaged fish are trucked daily and released at several sites in the western Delta. Salvage of fish at both facilities is conducted 24 hours a day, seven days a week at regular intervals. Entrained fish are subsampled for species composition and numbers.

Numbers of green sturgeon observed at these fish facilities have declined since the 1980s (Figure 4.A.4-7) which contributed to NMFS' decision to list the Southern DPS as a threatened species (71 FR 17757; April 7, 2006). In the Delta, the average number of green sturgeon salvaged per year at the SWP Skinner Fish Facility was 87 individuals between 1981 and 2000, and 20 individuals from 2001 through 2007. From the CVP Tracy Fish Collection Facility, green sturgeon counts averaged 246 individuals per year between 1981 and 2000, and 53 individuals from 2001 through 2007 (M. Donnellan pers.comm.). Patterns were similar between total numbers per year and numbers adjusted for water export volumes which increased during the 1970s and 1980s (Figure 4.A.4-7).

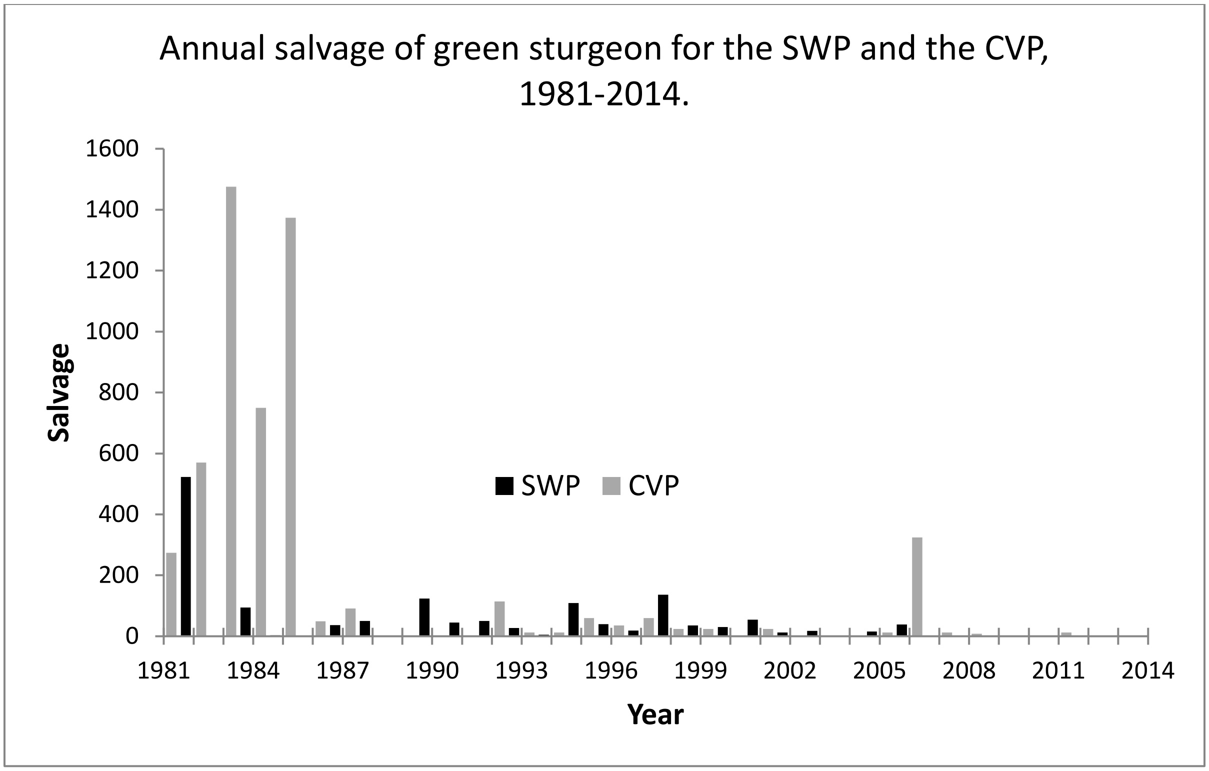

Figure 4.A.4-7. Estimated Annual Salvage of Green Sturgeon at CVP/SWP Fish Facilities in the South Sacramento-San Joaquin River Delta. Green Sturgeon Were not Counted at the Federal Central Valley Project Prior to 1981 (Data from DFW 2015)

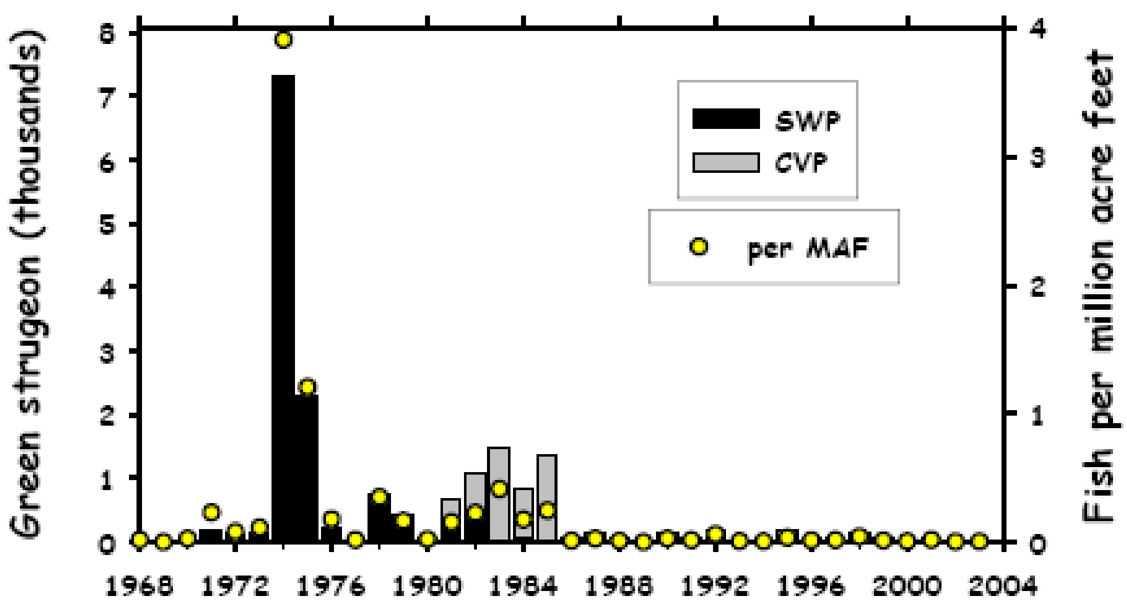

Figure 4.A.4-8. Estimated Annual Salvage of Green Sturgeon at CVP/SWP Fish Facilities in the South Sacramento-San Joaquin River Delta (DFG 2002). Prior to 1981, Green and White Sturgeon Were Counted Together and Reported Simply as Sturgeon at the CVP.

Annual counts of green sturgeon from the CVP/SWP fish facilities are not significantly correlated (Figure 4.A.4-9) (Beamesderfer 2005). Data on green sturgeon are available for both facilities from 1981–2005. Only 1% of the variability in salvage numbers was correlated between facilities (typically $p<0.10$ or $p<0.05$) (Beamesderfer 2005). In 1983, projected salvage at the CVP was 1,475 and only 1 at the SWP. In 1985, projected salvage at the CVP was 1,374 and only 3 at the SWP (Beamesderfer 2005).

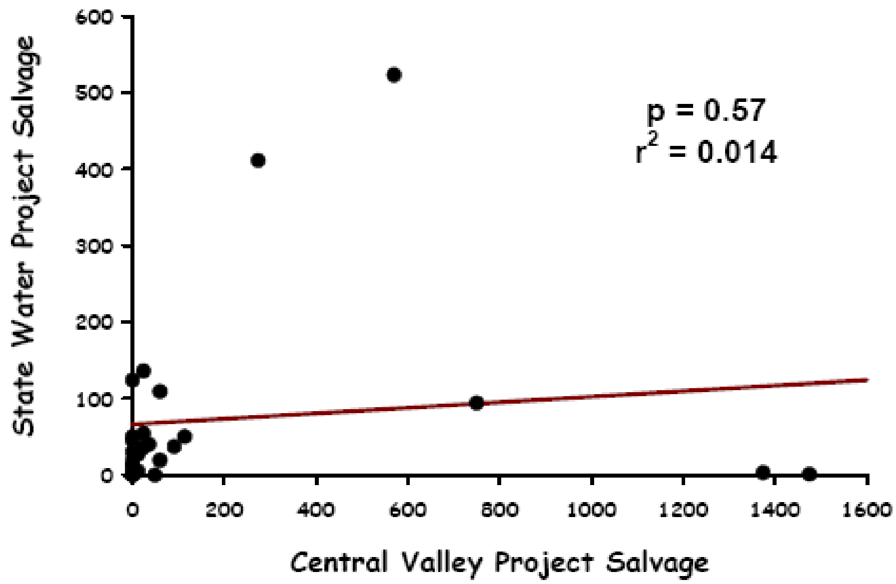

Figure 4.A.4-9. Green Sturgeon Salvage Numbers at State and Federal Facilities are Not Statistically Correlated (Beamesderfer 2005).

4.A.4.5 Threats and Stressors

The discussion below outlines some of the main threats and stressors to green sturgeon.

4.A.4.5.1 Reduced Spawning Habitat

Access to historical spawning habitat has been reduced by construction of migration barriers, such as major dams, that block or impede access to the spawning habitat. Major dams include Keswick Dam on the Sacramento River and Oroville Dam on the Feather River (Lindley et al. 2004; NMFS 2005). The Feather River is likely to have supported significant spawning habitat for the green sturgeon population in the Central Valley before dam construction (DFG 2002). Green sturgeon adults have been observed periodically in the lower Feather River (USFWS 1995a; Beamesderfer et al. 2004). Results of habitat modeling by Mora et al. (2009) suggested there is potential habitat on the Feather River upstream of Oroville Dam that would have been suitable for sturgeon spawning and rearing prior to construction of the dam. This modeling also suggested sufficient conditions are present in some sections of the San Joaquin River downstream of Friant Dam, however, long stretches of the San Joaquin River are de-watered so it is “not surprising that there are no contemporary accounts of green sturgeon in the San Joaquin River” (Mora et al. 2009, p. 45). Additionally is unknown whether green sturgeon ever inhabited the San Joaquin River or its tributaries (Beamesderfer et al. 2004).

4.A.4.5.2 Migration Barriers

Safe and unblocked migratory routes are necessary for passage within riverine habitats and between riverine and estuarine habitats (*e.g*., an unobstructed river or dammed river that still allows for passage). NMFS reports several potential migration barriers, on the Sacramento and Feather Rivers (71 FR 17757; April 7, 2006). As identified in the NMFS BIOP (2009) Red Bluff Diversion Dam (RBDD) was a major fish passage barrier. The gates were permanently raised in 2011, allowing fish passage continuously throughout the year. In 2012, a new screened pumping plant started operation marking a major improvement to fish passage in the Central Valley. In the Central Valley, approximately 4.6% of the total river kilometers have spawning habitat characteristics similar to where Northern DPS green sturgeon spawn, with only 12% of this habitat currently occupied by sturgeon (Neuman et al. 2007). Of the 88% that is unoccupied (approx. 4,000 kilometers [2,485 miles]), 44.2% is currently inaccessible due to dams (Neuman et al. 2007).

4.A.4.5.2.1 Sacramento River Deep Water Ship Channel

Sacramento River water passes through a set of locks at the end of the Sacramento River Deep Water Ship Channel at the connection with the Sacramento River. However, the locks prevent the fish that sense water coming from the Sacramento River from migrating from the Deep Water Ship Channel back to the Sacramento River (DWR 2005).

4.A.4.5.2.2 Fremont Weir

The Fremont Weir is located at the upstream end of the Yolo Bypass, a 40-mile (64-kilometer) long basin that functions as a flood control project on the Sacramento River. Green sturgeon are attracted by high floodwater flows into the Yolo Bypass basin and then concentrate behind Fremont Weir, which they cannot effectively pass (DWR 2005). Green sturgeon that concentrate behind the weir are subject to heavy illegal fishing pressure or become stranded behind the flashboards when high flood flows recede (Healey and Vincik Memo to J. Johnson 2011). Sturgeon can also be attracted to small pulse flows and trapped during the descending hydrograph (Harrell and Sommer 2003). Methods to reduce stranding and increase passage have been investigated by the DWR and CDFW (DWR 2007; Navicky pers. comm.). Thomas et al. (2013) modeled chronic stranding of green sturgeon at flood control structures that could have biologically significant impacts on the viability of Sacramento River green sturgeon population.

4.A.4.5.2.3 ACID Diversion Dam

Green sturgeon adults that migrate upstream in April, May, and June encounter the ACID diversion dam, which completely blocks passage to 5 miles of potential spawning habitat upstream of the diversion dam. However, it is unknown if spawning is occurring in this area (Poytress et al. 2013). Adults that pass upstream of ACID dam before April are forced to wait 6 months until the stop logs are pulled before returning downstream to the ocean. Upstream blockage forces sturgeon to spawn in approximately 12% less habitat between Keswick Dam and RBDD. Newly emerged green sturgeon larvae that hatch upstream of the ACID diversion dam would be forced to hold for 6 months upstream of the dam or pass over it and be subjected to higher velocities and turbulent flow below the dam, thus rendering the larvae and juvenile green sturgeon more susceptible to predation.

4.A.4.5.2.4 Delta Cross Channel (DCC) Gates Operations

It is thought that adult and juvenile green sturgeon use the same migratory routes as Chinook salmon. Delta Cross Channel gate closures occur during the winter and early spring sturgeon migration period (February through May) as required by State Water Resources Control Board (State Water Board) water right Decision 1641 (D-1641). Upstream migrating adult Chinook salmon are known to use the Delta Cross Channel as a migratory pathway when the gates are open (Hallock et al. 1970). When the gates are open, Sacramento River water flows into the central Delta and the Mokelumne and San Joaquin Rivers, providing migration cues. It is possible that attraction to water passing from the Sacramento River into the interior Delta causes delays and straying of green sturgeon, as it does to Chinook salmon (CALFED Bay-Delta Program 2001; McLaughlin and McLain 2004). The Delta Cross Channel completely blocks juvenile and adult sturgeon migration to and from the interior Delta when the gates are closed.

4.A.4.5.2.5 South Delta Temporary Barriers

The South Delta Temporary Barriers Program (TBP) was initiated in 1991. Its objectives are the short-term improvement of water conditions (water quality and elevation) for the south Delta and agricultural diversions, for the improvement of protection for San Joaquin River salmon, and for the development of data for the design of permanent gates. The program involves the seasonal installation of four barriers—one each on Middle River, Grant Line Canal, and Old River and a fish control barrier at the HOR. The barriers are a combination of rock placed into the main channel bed at each location along with overflow weirs and several gated culverts. These barriers are installed in the spring and removed in the fall.

When the barriers are in, green sturgeon within the barriers are trapped in the south Delta, where the habitat is generally regarded as low quality. When the barriers are removed, the green sturgeon are able to migrate out of the south Delta. The TBP continues to be implemented on an annual basis as an interim solution to water levels and circulation until a permanent solution can be implemented.

4.A.4.5.2.6 Suisun Marsh and Salinity Control Gates

DWR operates the Suisun Marsh Salinity Control Gates (SMSCG) to maintain water quality standards set by the SWRCB in D-1641 and the Suisun Marsh Preservation Agreement. The non-operation configuration of the SMSCG from June through August and any period during September through May when the gates are not in operation to meet salinity standards typically consists of the flashboards installed, but the radial gate operation is stopped and held open. Flashboards will be removed if it is determined that salinity conditions at all trigger stations would remain below standards for the remainder of the control season through May 31.

It is possible for young sturgeon to become entrained into Montezuma Slough and Suisun Marsh when the SMSCG is fully operational. Fish may enter Montezuma Slough as they emigrate from the Sacramento River during the fall when the gates are open to draw freshwater into the marsh and then may not be able to move back out when the gates are closed. However, the degree to which movement of green sturgeon is constrained is unknown. In addition, it is possible upstream passage of adults could be influenced as adult green sturgeon may pass through the marsh channels from December through May when their migration into spawning grounds could potentially be delayed. The effects of entrainment on juvenile green sturgeon at Roaring River

Distribution System screen intakes is unknown as screening standards for green sturgeon are currently unidentified.

4.A.4.5.2.7 Feather River

Potential barriers to green sturgeon passage in the Feather River include Shanghai Bend (RM 24.5), a natural geologic feature; an artificial rock weir structure at Sunset Pumps (RM 38.5), and Steep Riffle (RM 61), a natural feature. The extent of these sites as a barrier is not well understood because recently collected anecdotal information and data indicate that sturgeon are found upstream of these potential barriers at the Thermalito Outlet almost yearly (Seesholtz 2011). The rock structure at Sunset Pumps exhibits a 2–3 foot waterfall and a 4-foot wide slot with water velocities estimated at greater than 5 fps while flows are around 2,000 cfs. While it was originally determined that sturgeon likely could not pass this area at low flows (Niggemyer and Duster 2003), recent data from white sturgeon passage studies indicate white sturgeon can pass through velocities up to 8.3 fps (Anderson et al. 2007). Passage of Sunset Pumps by sturgeon during flows around 10,000 cfs is unlikely as velocities within the slot were estimated at around 10–15 fps (Niggemyer and Duster 2003). However, it has been estimated that when flows reach about 15,000 cfs, they over-top the rock structure and passage seems likely. Steep Riffle represented the most reasonable passable potential barrier during low-flow and high-flow conditions. Passage determinations at each of the potential migration barriers in the lower Feather River would continue to be speculative without a greater understanding of sturgeon migration patterns and physiologic limitations (Niggemyer and Duster 2003). Currently, studies are in place to attempt to gather this information in order to better describe the impacts that sturgeon may face in the Feather River.

4.A.4.5.2.7.1 Exposure to Toxins

Exposure of green sturgeon to toxins has been identified as a factor that can lower reproductive success, decrease early life stage survival, and cause abnormal development, even at low concentrations (USFWS 1995a; Environmental Protection Information Center et al. 2001; Klimley 2002). Water discharges containing metals from Iron Mountain Mine, located adjacent to the Sacramento River, have been identified as a possible factor affecting survival of sturgeon downstream of Keswick Dam. In addition, storage limitations and limited availability of dilution flows cause downstream copper and zinc levels to exceed salmonid tolerances. Treatment processes and improved drainage management in recent years have reduced the toxicity of runoff from Iron Mountain Mine to acceptable levels. Although the impact of trace elements on green sturgeon reproduction is not completely understood, negative impacts similar to those of salmonids are suspected (USFWS 1995a; Environmental Protection Information Center et al. 2001; Klimley 2002).

Green sturgeon consume overbite clams (*Potamocorbula amurensis*) and Asian clams (*Corbicula fluminea*), which are known to bioaccumulate selenium rapidly and lose selenium slowly (Linville et al. 2002). Selenium is transferred to the egg yolk where it can cause mortality of larvae. Although chronic and acute exposure to toxics has been identified as a factor adversely affecting various life stages of green sturgeon, the severity, frequency, geographic locations, and population level consequences of exposure to toxics have not been quantified (Linville et al. 2002). However, Linville (2006) observed larvae to have increased skeletal deformities and mortality associated with maternal effects of selenium exposure, while smaller quantities (about

20 milligrams per kilogram [mg/kg]) decreased feeding efficiency and larger quantities (greater than 20 mg/kg) reduced growth rates after four weeks (Lee et al. 2008a).

Methylmercury is another toxic substance that could potentially affect sturgeon development and survival. Between 2002 and 2006, sediment concentrations of methylmercury were highest in the Central Bay, while shallower parts of San Pablo Bay and Suisun Bay also contained levels greater than 0.2 parts per billion (ppb) (San Francisco Estuary Institute 2007). The amount of methylmercury resulting in the death of juvenile green sturgeon ranges between 20 to 40 mg/kg, with greater consumption increasing mortality significantly (Lee et al. 2008b).

4.A.4.5.3 Harvest

As a long-lived, late maturing fish with relatively low fecundity and periodic spawning, the green sturgeon is particularly susceptible to threats from overfishing (Musick 1999). Green sturgeon are regularly caught in the sport, commercial, and tribal fisheries, particularly in Oregon and Washington commercial fisheries (Beamesderfer 2005). Total captures of green sturgeon in the Columbia River Estuary in commercial fisheries between 1985 and 2003 ranged from 46 fish per year to 6,000 (Adams et al. 2007). However, a high proportion of green sturgeon present in the Columbia River, Willapa Bay, and Grays Harbor (as high as 80% in the Columbia River) may be from the Southern DPS (DFG 2002; Israel et al. 2004). Long-term data indicate that harvest for green sturgeon occurs primarily in the Columbia River (51%), coastal trawl fisheries (28%), the Oregon fishery (8%), and the California tribal fishery (8%). Harvest of green sturgeon dropped substantially from over 6,000 from 1985 to 1989 to 512 in 2003 (Adams et al. 2007). This reduction is not due to declining catch-per-effort but is in response to market conditions, regulation changes, and changing fisheries for other species (Adams et al. 2007). Coastal trawl fisheries have declined to low levels, thereby lowering the by-catch of green sturgeon. In 2003, Klamath and Columbia River tribal fisheries accounted for 65% of total catch (Adams et al. 2007). In 2007, California and Washington revised recreational fishing regulations to prohibit retention of green sturgeon, and Oregon prohibited retention of green sturgeon in lower Columbia River recreational fisheries. The retention of green sturgeon in commercial fisheries has been prohibited in the Columbia River since 2006 and statewide in Washington since 2007. California has prohibited commercial fishing for sturgeon since 1917 (Skinner 1962: 84). Green sturgeon are also vulnerable to recreational sport fishing in the Bay-Delta estuary and Sacramento River, as well as other estuaries located in Oregon and Washington. Green sturgeon are primarily captured incidentally in California by sport fishermen targeting the more desirable white sturgeon, particularly in San Pablo and Suisun Bays (Emmett et al. 1991).

To protect spawning green sturgeon, new federal and state regulations, including the take prohibition in the NMFS ESA Section 4(d) rule (75 FR 30714; June 2, 2010), mandate that no green sturgeon can be taken or possessed in California (DFG 2007). If green sturgeon are caught incidentally and released while fishing for white sturgeon, anglers are asked to report it to CDFW on their white sturgeon report card. The level of hooking mortality that results following release of green sturgeon by anglers is unknown. Sport fishing captures have declined through time, but the factors leading to the decline are unknown. CDFW (DFG 2002) indicates that sturgeon are highly vulnerable to the fishery in areas where sturgeon are concentrated, such as the Delta and Suisun and San Pablo Bays in late winter, and the upper Sacramento River during spawning migration. Because many sturgeon in the Columbia River, Willapa Bay, and Grays

Harbor are likely from the Southern DPS, additional harvest closures in these areas would likely benefit the Southern DPS.

Poaching (illegal harvest) of sturgeon is known to occur in the Sacramento River, particularly in areas where sturgeon have been stranded (e.g., Fremont Weir) (Marshall pers. comm.), as well as throughout the Bay-Delta (Schwall pers. comm.). Catches of sturgeon are thought to occur during all years, especially during wet years. Green sturgeon inhabiting the San Joaquin River portion of the Delta experience heavy fishing pressure, particularly from illegal fishing (USFWS 1995a). Areas just downstream of Thermalito Afterbay outlet, Cox's Spillway, and several barriers impeding migration on the Feather River may be areas of high adult mortality from increased fishing effort and poaching. Poaching rates in the rivers and estuary and the impact of poaching on green sturgeon abundance and population dynamics are unknown.

4.A.4.5.4 Reduced Rearing Habitat

Historical reclamation of wetlands and islands have reduced and degraded the availability of suitable in- and off-channel rearing habitat for green sturgeon. Further, channelization and hardening of levees with riprap has reduced in- and off-channel intertidal and subtidal rearing habitat. The resulting changes to river hydraulics, riparian cover, seasonal floodplain inundation, and geomorphology affect important ecosystem functions (Sweeney et al. 2004). The impacts of channelization and riprapping are thought to affect available food resources of larval, post-larval, juvenile, and adult stages of sturgeon, as these life stages are dependent on the food web in freshwater and low-salinity regions of the Delta.

4.A.4.5.5 Increased Water Temperature

Water temperature within suitable tolerances would include: stable water temperatures of 11–17°C (optimal range = 14–16°C) in spawning reaches for egg incubation (March–August) (Van Eenennaam et al. 2005); temperatures less than 20°C for larval development (Werner et al. 2007); and temperatures below 24°C for juveniles (Mayfield and Cech 2004, Allen et al. 2006). Temperatures near the Red Bluff Diversion Dam on the Sacramento River historically occur within optimum ranges for sturgeon reproduction; however, temperatures downstream, especially later in the spawning season, were reported to be frequently above 63°F (17.2°F) (USFWS 1995a). High temperatures in the Sacramento River during the February to June period no longer appear to be a major concern for green sturgeon spawning, egg incubation, and juvenile rearing, as temperatures in the upper Sacramento River are actively managed for winter-run Chinook salmon. The Shasta temperature control device, installed at Shasta Dam in 1998, in combination with improved cold-water pool management and storage in Lake Shasta, have resulted in improved cool water stream conditions in the upper Sacramento River.

Water temperatures in the upper anadromous reach of the Feather River (between Fish Barrier Dam and Thermalito Outlet) appear adequate for spawning and egg incubation and, in some years, water temperatures downstream of the Thermalito Outlet are also adequate for spring spawning and egg incubation. Prior to the construction of the Oroville Dam, water temperatures in the Feather River at Oroville averaged 65–71°F from June through August for the period of 1958–1968 (DWR 2004). After Oroville Dam construction, water temperatures in the Feather River at the Thermalito Afterbay averaged 60–65°F from June through August for the period of

1993–2002 (DWR 2004). In addition, modeling results indicate that under existing conditions, water temperatures several miles downstream of the Thermalito Outlet would average 66°F or less in 80% of all days in July (DWR 2005a). Based on this information, post-Oroville Dam water temperatures may be cooler in the lower Feather River during the summer months than historical river temperatures (DWR 2005a). In the Sacramento River, green sturgeon spawn in the spring and summer. Historically, temperatures associated with late spring and summer spawning were found in reaches of the Sacramento and Feather Rivers above impassable barriers. Most anecdotal observations of Southern DPS in the Feather River come from the pool below the Thermalito Outlet (DWR 2007). These observations suggest Southern DPS are selecting the habitat found at the outlet for holding (and possible spawning during some years) over the cooler upstream reach, possibly due to conditions associated with the augmented flows below the outlet. Water temperatures necessary for spawning and egg incubation do not persist below the Thermalito Outlet during late spring and summer. Therefore, late spring and summer spawning may not be supported in the Feather River. NMFS states "An effective population of spawning green sturgeon (i.e., a population that is contributing offspring to the next generation) no longer exists in the Feather River and was likely lost due to … thermal barriers associated with the Thermalito Afterbay Facility" (71 FR 17757, 17762; April 7, 2006). Spring-run Chinook salmon regularly hold below and pass upstream of the Thermalito Outlet (DWR 2005b) suggesting that the Thermalito Afterbay Outlet does not represent a complete thermal barrier to coldwater species. However, the de-coupling of potential spawning habitat (below the Thermalito Outlet) and late spring and summer water temperatures necessary for successful spawning and egg incubation may limit green sturgeon spawning in the Feather River to a narrow window in the spring.

The lack of flow in the San Joaquin River from dam and diversion operations and agricultural return flows contribute to higher temperatures in the mainstem San Joaquin River, offering less water to keep temperatures cool for sturgeon, particularly during late summer and fall, although white sturgeon have been observed spawning in the San Joaquin even in dry water years (Jackson and Van Eenennaam 2013). Though these effects are difficult to measure, temperatures in the lower San Joaquin River continually exceed preferred temperatures for sturgeon migration and development during spring months. Temperatures at Stevenson on the San Joaquin River near the Merced River confluence recorded on May 31 (spawning typically occurs from April to June) between 2000 and 2004 ranged from 77 to 82°F (25 to 27.8°C) (DWR 2007).

Juvenile sturgeon are also exposed to increased water temperatures in the Delta during the late spring and summer, although temperature in the Delta is mostly controlled by ambient air temperatures.

4.A.4.5.6 Nonnative Species

Green sturgeon have most likely been impacted by non-native invasive species introductions resulting in changes in trophic interactions in the Delta. Many of the recent introductions of invertebrates have greatly affected the benthic fauna in the Delta. DFG (2002) reviewed many of the recent non-native invasive species introductions and the potential consequences to green sturgeon. Most notable species responsible for altering the trophic system of the Sacramento-San Joaquin Delta include the overbite clam, the Chinese mitten crab, the introduced mysid shrimp *Acanthomysis bowmani*, and another introduced crustaceans, *Gammarus* sp.

Introductions of invasive plant species such as the water hyacinth (*Eichhornia crassipes*) and *Egeria densa* have altered nearshore and shallow water habitat by raising temperatures and inhibiting access to shallow water habitat. *Egeria* forms thick “walls” along the margins of channels in the Delta. This growth prevents juvenile native fish from accessing their preferred shallow water habitat along the channel’s edge. Water hyacinth creates dense floating mats that can impede river flows and alter the aquatic environment beneath the mats. Dissolved oxygen levels beneath the mats often drop below optimal levels for fish due to the increased amount of decaying vegetative matter produced from the overlying mat. Like *Egeria*, water hyacinth is often associated with the margins of the Delta waterways in its initial colonization, but can eventually cover the entire channel if conditions permit. This level of infestation can produce barriers to anadromous fish migrations within the Delta. The introduction and spread of *Egeria* and water hyacinth have created the need for aquatic weed control programs that utilize herbicides targeting these species.

Recent stomach content analysis of white sturgeon from the San Francisco Bay estuary indicates that the invasive overbite clam (*Potamocorbula amurensis*) may now be a major component of the white sturgeon diet and possibly green sturgeon diets, and unopened clams were often observed throughout the alimentary canal (Kogut 2008). Kogut’s study found that at least 91% of clams that passed through sturgeon digestive tracts were alive. Green sturgeon could be affected in a similar manner. This suggests sturgeon are potential vehicles for transport of adult overbite clams and also raise concern about the effect of this invasive clam on sturgeon nutrition and contaminant exposure. Consumption of *Potamocorbula* and *Corbicula*, is of particular concern because of the high bioaccumulation rates of these clams (Linville et al. 2002). Although Chinese mitten crabs may be eaten by adult green sturgeon, it is unlikely that they are a major prey item. The Chinese mitten crab population in the Delta has undergone a substantial decline since 2002 (Hieb 2012); therefore, it has not been a major factor affecting green sturgeon during this period.

4.A.4.5.7 Dredging

Hydraulic dredging to allow commercial and recreational vessel traffic is a common practice in the Sacramento and San Joaquin Rivers, navigation channels in the Delta, and Suisun, San Pablo, and San Francisco Bays. Such dredging operations pose risks to bottom-oriented fish such as green sturgeon. Studies by Buell (1992) reported approximately 2,000 sturgeon entrained in the removal of one million tons of sand from the bottom of the Columbia River at depths of 60 to 80 feet (18 to 24 meters). In addition, dredging operations can decrease the abundance of locally available prey species, and contribute to resuspension of toxics such as ammonia[4], hydrogen sulfide, and copper during dredging and dredge spoil disposal, and alter bathymetry and water movement patterns (NMFS 2006).

[4] Ammonia in water generally forms some amount of ammonium. Therefore, the use of the term *ammonia* implies that both ammonia and ammonium may be present.

4.A.4.5.8 Reduction in Turbidity

Turbidity levels in the Delta have declined over the past few decades (Jassby et al. 2002), but little is known about the potential effects of reduced turbidity on green sturgeon.

4.A.4.5.9 Entrainment

Larval sturgeon are susceptible to entrainment from nonproject (not part of CVP or SWP) water diversion facilities because of their migratory behavior and habitat selection in the rivers and Delta. The overall impact of entrainment of fish populations is typically unknown (Moyle and Israel 2005); however, there is enough descriptive information to predict where green sturgeon may be entrained. Herren and Kawasaki (2001) documented 431 nonproject diversions on the Sacramento River between Sacramento and Shasta Dam. Entrainment information regarding larval and post-larval individual green sturgeon is unreliable because entrainment at these diversions has not been monitored and field identification of green sturgeon larvae is difficult. USFWS staff are working on identification techniques and are optimistic that green sturgeon greater than 40 millimeters (1.6 inch) can be identified in the field (Poytress 2006). Sturgeon collected at the Glenn-Colusa Irrigation District diversion located on the upper Sacramento River are not identified to species, but are assumed to primarily consist of green sturgeon because white sturgeon are known to spawn primarily downstream (Schaffter 1997). Although screens at the Glenn-Colusa Irrigation District diversion satisfy both the NMFS and CDFW screening criteria for salmonids, the effectiveness of these criteria is unknown for sturgeon. Low numbers of green sturgeon (less than 1% of total present February to June) have also been identified and entrained at the Red Bluff Research Pumping Plant (Borthwick et al. 1999).

In the Feather River, there are eight large diversions greater than 10 cubic feet per second (cfs) and approximately 60 small diversions between 1 and 10 cfs between the Thermalito Afterbay outlet and the confluence with the Sacramento River (USFWS 1995a). Based on potential entrainment problems of green sturgeon elsewhere in the Central Valley and the presence of multiple screened and unscreened diversions on the Feather River, it is thought that operation of unscreened water diversions on the Feather River are a possible threat to juvenile green sturgeon.

Presumably, juvenile green sturgeon become less susceptible to entrainment as they grow and their swimming ability and capacity to escape diversions improves. The majority of North American green sturgeon captured in the Delta are between 200 and 500 millimeters (7.9 and 19.7 inches) long (DFG 2002). Herren and Kawasaki (2001) inventoried water diversions in the Delta and counted 2,209 diversions of various types, only 0.7% of which were screened. The majority of these diversions were between 12 and 24 inches (305 and 610 mm) in diameter. The vulnerability of juvenile green sturgeon to entrainment at these unscreened diversions is largely unknown, although in two multiyear studies (Nobriga et al. 2004; Pickard et al. 1982) no green sturgeon were caught In a recent study Mussen et. al. (2014) found that juvenile green sturgeon are potentially vulnerable to unscreened water diversions, showing at fairly high rates (26-61%) of entrainment in laboratory studies.

The largest diversions in the Delta are the CVP/SWP facilities, located in the southern Delta, where a low number of juvenile green sturgeon have been recorded as part of fish salvage monitoring (DFG 2002).

The Tracy Fish Collection Facility (TFCF), at the intake to the DMC, is designed to intercept fish before they are entrained into the DMC by the Tracy Pumping Plant. Fish are collected and transported by tanker truck to release sites away from the pumps. Adult green sturgeon are rarely observed at the TFCF. Green sturgeon salvage counts averaged 246 individuals per year between 1981 and 2000, and 34 individuals per year between 2001 and 2011 (Donnellan pers. comm.). This reduction in salvage is consistent with a significant reduction in white sturgeon take at the salvage facilities in the same periods (NMFS 2005).

The Skinner Fish Protection Facility (SFPF) located between Banks and CCF, intercepts fish, which are collected and transported by tanker truck to downstream release sites. This facility uses behavioral barriers to guide targeted fish into holding tanks for subsequent transport by truck to release sites within the Delta. The average number of green sturgeon taken per year at the SWP Skinner Fish Facility was 87 individuals between 1981 and 2000, and 10 individuals from 2001 through 2014 (California Department of Fish and Wildlife 2015). The number of green sturgeon has been low since 2008, with 22 green sturgeon salvaged from 2008 to 2012; none were salvaged for 5 years out of the 7.

4.A.4.5.10 Low Flows

In its final rule listing the Southern DPS, NMFS states that “CDFG (1992) and FWS (1995) found a strong correlation between mean daily freshwater outflow (April to July) and white sturgeon year class strength in the Sacramento-San Joaquin Estuary (these studies primarily involve the more abundant white sturgeon; however, the threats to green sturgeon are thought to be similar), indicating that insufficient flow rates are likely to pose a significant threat to green sturgeon.” (71 FR 17757, 17763; April 7, 2006). NMFS (2009) states, “An adequate flow regime (i.e., magnitude, frequency, duration, seasonality, and rate-of-change of fresh water discharge over time) is necessary for normal behavior, growth, and survival of all life stages in the upper Sacramento River”. It is envisioned that a flow regime of this type would contain sufficient flow magnitude to induce spawning, emigration, and maintain water temperatures within optimal range for egg, larval, and juvenile development (52–66°F) (Cech et al. 2000, Mayfield and Cech 2004, Van Eenennaam et al. 2005, Allen et al. 2006). Flows need to be adequate to reduce incidences of fungal infection and flush fine sediment from substrate.

High temperatures caused by lower flows in rivers and the Delta may have a negative effect on sturgeon populations. DFG (1992) and USFWS (1995) found a strong correlation between mean daily temperature (April to July) and white sturgeon year-class strength from the Sacramento River. The Shasta Temperature Control Device began operating in 1997, but storage limitations may limit the ability of Shasta Dam releases to regulate temperatures during drier water years. DFG (1992) and USFWS (1995) also found a strong correlation between mean daily freshwater outflow from the Sacramento-San Joaquin watershed and year-class strength in the estuary. It should be noted that flow and temperature are correlated, and the DFG and USFWS studies were conducted prior to temperature control device installation on Shasta Dam; therefore, it is difficult to quantify flow effects on juvenile production independent of temperature.

In the Feather River under low flow conditions (~2,000 cfs), the Sunset pumps are most likely a barrier to green sturgeon passage (Niggemyer and Duster 2003). In some years, water temperatures downstream of the Thermalito Outlet are inadequate for spawning and egg incubation but are not likely a physical barrier for adult migration into the upper reach, which has been suggested as a reason why green sturgeon are not found in the river during low flow years (DWR 2007).

The lack of flow in the San Joaquin River from dam and diversion operations and agricultural return flows contribute to higher temperatures in the mainstem San Joaquin River, offering less water to keep temperatures cool for sturgeon, particularly during late summer and fall. Whether direct or indirect, the effects of flow on green sturgeon are not well understood but likely play an important role in population performance, which is why lows flows are documented as a potential threat in NMFS' 2002 and 2005 status reviews (Adams et al. 2002; NMFS 2005) and the NMFS' proposed and final rules for listing the Southern DPS (70 FR 17386, April 6, 2005; 71 FR 17757, April 7, 2006).

Within bays and estuaries adjacent to the Sacramento River (i.e., the Sacramento-San Joaquin Delta and the Suisun, San Pablo, and San Francisco bays), sufficient flow into the bay and estuary to allow adults to successfully orient to the incoming flow and migrate upstream to spawning grounds is required. Sufficient flows are needed to attract adult green sturgeon to the Sacramento River from the bay and to initiate the upstream spawning migration into the upper river. Currently, flows provide the necessary attraction to green sturgeon to enter the Sacramento River. Nevertheless, these flows are substantially less than what would have been available historically to stimulate the spawning migration.

4.A.4.5.11 Predation

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

4.A.4.6 Green Sturgeon Viable Salmonid Population (VSP) Parameters

The VSP concept measures population performance in term of four key parameters: abundance, population growth rate, spatial structure, and diversity. Although the VSP concept was developed for Pacific salmonids, the underlying parameters are general principles of conservation biology and can therefore be applied more broadly to green sturgeon.

4.A.4.6.1 Abundance

Abundance is examined at the population level, and therefore the size of population is what is really being measured. Two ways have been used to infer historical abundance and population trends in green sturgeon first by examining the number salvaged at the state and federal pumping facilities (Figure 4.A.4-7 and Figure 4.A.4-8), and second by incidental catch of green sturgeon in CDFW's white sturgeon sampling and tagging program. Biases in the data are problematic in that salvage is a measure of how the facilities entrain green sturgeon and can be confounded by dispersal patterns, collection nuances due to delta flow dynamics, and changes in configuration and operation of the facilities over time. Catches of green sturgeon in the white sturgeon sampling program are inherent with variability due to low incidence of green sturgeon in catches coupled with variable effort, and catchability, which leads to high probable error in estimates of green sturgeon abundance based on catch of white sturgeon. Only recently has more rigorous scientific inquiry begun with (Israel and May 2010) and (Mora unpublished data).

Salvage data from the State and Federal fish facilities can infer abundance has declined over the years (Figure 4.A.4-7 and Figure 4.A.4-8) and there is a moderate negative correlation at a significant level between year and salvage at each of the facilities since 1981 (Federal RHO = -.5748, $p < .001$; State RHO = -.5166, $p < .002$) (Figure 4.A.4-9).

More robust estimates of Green Sturgeon abundance are being developed by the University of California, Davis, using acoustic telemetry surveys to locate green sturgeon in the Sacramento River. Results of these surveys indicate an average annual spawning run of 272 fish (Mora unpublished data). This estimate does not include the number of spawning adults in the lower Feather River, where green sturgeon spawning was recently confirmed. This estimate is preliminary and involves a number of untested assumptions regarding sampling efficiency, discrimination between green and white sturgeon, and spawner residence time. Although caution must be taken in using this estimate to infer the spawning run size for the Sacramento River until further analyses are completed, this preliminary estimate provides reasonable order-of-magnitude numbers for recovery planning purposes until such time as new information is developed (NMFS SOS Draft for Green Sturgeon).

4.A.4.6.2 Productivity

Productivity (i.e. population growth rate) should address whether the population able to maintain its present status (i.e., is the population growth rate approximately 1.0), whether the population has the ability to grow i.e. the population at carrying capacity, or the habitat is able carry higher abundances. Levels of understanding around these factors are poorly understood for Southern DPS green sturgeon. Larval abundance as derived from count data at RBDD and GCID shows high variance between years, but also highlights years that are clearly successful in producing larval green sturgeon. An example of this is occurred in 2011 when 3700 larvae were captured (Poytress *et al.* 2012). For comparison, counts from other years were an order of magnitude lower. Some concern exists over whether the temperature regime that is maintained in the Upper Sacramento River for winter-run Chinook is too cold for optimal green sturgeon hatching success and optimal larval growth (Poytress et al. 2013). These data are not standardized between years, and there are questions about sampling methodology, so the data may not be purely representative of each year's productivity. In characterizing green sturgeon year class strength, it appears to be episodic with the a few successful spawning events driving abundance (NMFS 2010). The variability in the data makes it unclear whether the population is able to maintain its current level or attain higher abundance than present. Because of the paucity of data, other indicators such as cohort replacement ratios and spawner abundance trends cannot be calculated. The long lifespan of the species and long age to maturity makes trend detection dependent upon data sets spanning decades, something that is currently lacking. Acoustic telemetry work begun by Ethan Mora (UC Davis) on the Sacramento River and by Alicia Seesholtz (DWR) on the Feather River, as well as larval and juvenile studies begun by Bill Poytress (USFWS) may eventually produce sufficient data to allow the calculation of productivity metrics.

4.A.4.6.3 Spatial Structure

Green sturgeon range from Ensenada, Mexico to the Bering Sea, Alaska (Colway and Stevenson 2007; Moyle 2002). Green sturgeon spawn in two California basins: the Sacramento and Klamath Rivers. During the late summer and early fall, subadults and nonspawning adult green sturgeon frequently can be found aggregating in estuaries along the Pacific coast (Emmett 1991, Moser and Lindley 2007). These reproducing populations are genetically distinct and occupy the Southern (Sacramento) and Northern (Klamath) DPS (Adams et al. 2002; Israel et al. 2004).

1. A Northern DPS consisting of populations originating from coastal watersheds northward of and including the Eel River (*i.e.,* Klamath, Trinity, and Rogue Rivers).
2. A Southern DPS consisting of populations originating from coastal watersheds south of the Eel River.

The Southern and Northern DPS co-occur throughout much of their coastal range including bays and estuaries in Oregon and Washington. Israel *et al.* (2009) found that green sturgeon within the SF Estuary and Sacramento River are almost entirely Southern DPS. Additional data collected from acoustic tagging studies give high certainty to what Israel found genetically.

Within inland waters (i.e., upstream [east] of the Golden Gate Bridge) green sturgeon are known to range throughout the estuary and the delta and range up the Sacramento, Feather, and Yuba Rivers. Within the Sacramento River, Keswick Dam (RK 486), located represents the highest point that would be accessible to green sturgeon, but ACID dam (RK 480) blocks access to the top 6 kilometers of remaining habitat. Limited larval sampling by USFWS at 16 and 56 kilometers below Keswick captured no larvae. Habitat usage has been confirmed to the confluence with Ink Creek (59 kilometers below Keswick), which was confirmed as a spawning site in 2011 (Poytress *et al.* 2012). In the Feather River, DWR staff have observed green sturgeon as high as the Fish Barrier Dam. Spawning has recently been recorded with eggs from three different sturgeon females (Van Eenenaam 2011). In spring 2011, many sturgeon adults were spotted while DIDSON surveys were being conducted (Seesholtz 2011). Significant habitat on the Lower Feather River, while modified, remains accessible downstream from the Thermalito Afterbay outlet (DWR 2005a). Green sturgeon have been documented up to Daguerre Point Dam on the Yuba River (Bergman *et al.* 2011). Green Sturgeon cannot pass through the fish ladder at Daguerre Point Dam, although potential spawning habitat does exist upstream of the dam. Although no historical accounts exist for green sturgeon spawning above the current dam sites, suitable spawning habitat likely existed. The upstream extent of historical spawning by green sturgeon in the Sacramento River system is unknown. It is unknown whether green sturgeon historically spawned in the Feather River either downstream or upstream of current Oroville Dam or the Thermalito Afterbay outlet. Spawning is suspected to have occurred in the past due to the continued presence of adult green sturgeon in the river below Oroville Dam. The current or historical occurrence of green sturgeon in the San Joaquin River has been a source of much speculation. It is unclear whether green sturgeon are currently present or were historically present and have been extirpated from the San Joaquin River (NMFS 2005, Beamesderfer et al. 2007). No juvenile green sturgeon have been documented to occur in the San Joaquin River, although no directed sturgeon studies have ever been undertaken in the San Joaquin River (USFWS 1995a, DFG 2002, Adams et al. 2002, Beamesderfer et al. 2004, NMFS 2005).

Mora *et al.* (2009) analyzed and characterized known green sturgeon habitat and used that characterization to identify potential green sturgeon habitat within the Sacramento and San Joaquin River basins that now lies behind impassable dams. This study concluded that about 9% of historically available habitat is now blocked by impassible dams, but more importantly, this blocked habitat was likely high quality for spawning.

Additional studies by UC Davis (Mora et al. 2015) have revealed that green sturgeon spawning sites are concentrated in just a handful of locations. Mora found that on the Sacramento River, just 3 sites accounted for over 50% of the green sturgeon documented in June of 2010, 2011, and 2012. All were presumed to be at these locations to spawn. This is a critical point about the application of the spatial structure VSP parameter, which is largely concerned with the spawning habitat spatial structure. Given a high concentration of individuals at just a few spawning sites, extinction risk due to stochastic events would be expected to be increased.

Current scientific understanding indicates that Southern DPS green sturgeon is a single, independent population, which principally spawns in the main stem Sacramento River, and breeds opportunistically in the Feather River and possibly the Yuba River. The species is highly vulnerable to poaching and catastrophic events due to concentrated spawning in a few locations.

4.A.4.6.4 Diversity

Diversity, as defined in the VSP concept in (McElhany *et al.* 2000), includes genetic traits such as DNA sequence variation, and other traits that are influenced by both genetics and the environment, such as ocean behavior, age at maturity, and fecundity. Variation is important because it allows the species to utilize a wider array of environments, it insulates the species from short-term spatial and temporal changes in the environment, and it provides the raw material that is necessary for adaptation to changing environmental conditions.

While recognition that diversity is essential to the viability of the species, the specifics of these traits within green sturgeon are not recognized well enough to know whether Southern DPS are buffered against long-term extinction risk. Given that abundance estimates for Southern DPS are low, larger numbers of individuals within the population should offer greater diversity and therefore greater viability. Focus should be directed on trying to increase the number of individuals and seek to establish a second breeding population outside the Sacramento River, with the Feather River being best positioned, and to a lesser extent, the Yuba River. Highly altered environments within the Central Valley could influence basic diversity principles such as run timing and behavior (see stressors).

4.A.4.6.5 Conclusion

Southern DPS viability is inhibited by small population size, lack of multiple populations, and the constriction of spawning sites to a few locations. The probability of extinction is thought to be moderate because there is so much uncertainty regarding the scope of threats and viability of population indices (National Marine Fisheries Service 2010). McElhany et al. (2000) defined viability as "independent population having a negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year timeframe". The extinction risk facing Southern DPS is not negligible over the long-term (approximately100 year) time horizon; therefore it can be concluded that the DPS is not viable. Population viability analysis done by Thomas et al. (2013) in relation to stranding at weirs supports this conclusion. Many assumptions were made in his model that need verification, but it was alarming to note that over a 50-year time period the DPS declined under all scenarios where stranding events were recurrent over the lifespan of a green sturgeon.

Having only one population is problematic in that an ESU represented by a single population at moderate risk of extinction is at high risk of extinction over the long run (Lindley et al. 2007). This concern applies to any DPS or ESU represented by a single population, although NMFS concluded, after weighing all available information, that the extinction risk is moderate (National Marine Fisheries Service 2010).

4.A.4.6.6 Relevant Conservation Efforts

The Anadromous Fish Restoration Program of the Central Valley Project Improvement Act contains a goal of supporting efforts that lead to doubling the natural production of anadromous fish in the Central Valley on a sustainable, long-term basis, at levels not less than twice the average levels attained during the period of 1967 to 1991. Although most efforts of the Anadromous Fish Restoration Program have focused on Chinook salmon because of their listing history and status, sturgeon may receive some unknown amount of incidental benefit from these restoration efforts. For example, the acquisition of water for flow enhancement on tributaries to the Sacramento River, fish screening for the protection of Chinook salmon and Central Valley steelhead, spawning gravel augmentation, or riparian revegetation and instream restoration projects would likely have some ancillary benefits to sturgeon. The Anadromous Fish Restoration Program has also invested in a green sturgeon research project that has helped improve our understanding of the life history requirements and temporal patterns of the Southern DPS.

Many beneficial actions have originated from and been funded by the CALFED Bay-Delta Program (CALFED), including such projects as floodplain and instream restoration, riparian habitat protection, fish screening and passage projects, research on nonnative invasive species and contaminants, restoration methods, watershed stewardship, and education and outreach programs. In its proposed rule for listing ESUs of West Coast salmonids, NMFS reviewed the details of the Central Valley Project Improvement Act and CALFED programs and potential benefits for anadromous fish, particularly Chinook salmon and Central Valley steelhead (69 FR 33102; June 14, 2004). Projects potentially benefiting sturgeon primarily consist of fish screen evaluation and construction projects, restoration evaluation and enhancement activities, and contaminant studies. Two evaluation projects specifically addressed green sturgeon, while the remaining projects primarily address listed salmonids and fishes of the area in general. The new information developed through these research investigations will be used to enhance the understanding of the risk factors affecting population dynamics and recovery, thereby improving the ability to develop effective management measures.

The Delta Regional Ecosystem Restoration Implementation Plan (DRERIP) was formed to guide the implementation of CALFED Ecosystem Restoration Plan elements in the Delta. The DRERIP team has created a suite of ecosystem and species conceptual models, including green sturgeon (Israel and Klimley 2008), that document existing scientific knowledge of Delta ecosystems. The DRERIP team is in the process of using these conceptual models to assess the suitability of actions proposed in the Ecosystem Restoration Plan for implementation.

In response to concerns about passage impediment to green sturgeon and other migratory species, operations of the Red Bluff Diversion Dam have been ceased and a new water pumping facility with a state-of-the –art fish screen has been constructed. The project now provides a reliable water supply for high-value crops in Tehama, Glenn, Colusa, and northern Yolo Counties while providing year-round unimpeded fish passage.

The combination of increased law enforcement and new sport fishing regulations adopted over the past several years specifically to protect sturgeon and reduce their harvest is expected to further reduce illegal fishing practices as well as the effects of incidental harvest of green sturgeon by recreational anglers throughout the range of the species. Mitigation under the Delta Fish Agreement has increased the number of wardens enforcing harvest regulations for steelhead and other fish in the Delta and upstream tributaries by creating the Delta Bay Enhanced Enforcement Program.

4.A.4.7 Recovery Goals

On November 12, 2009, NMFS announced its intent to develop a recovery plan for the Southern DPS and has requested information from the public (74 FR 58245). An outline for the recovery plan was published in December 2010 (NMFS 2010), but the plan itself has not yet been completed.

Key recovery needs and implementation measures identified for the Southern DPS include the following:

- Additional spawning and egg/larval habitat
    - Restore access to suitable habitat
    - Improve potential habitat
    - Establish additional spawning populations
    - Ensure adequate spatial separation of spawning populations
    - Ensure all spawning populations are of sufficient size to meet genetic diversity criteria
- Research/Monitoring
    - Determine current and future population abundance and distribution of all life stages
    - Obtain data needed for population viability assessment
    - Determine fisheries-specific discard mortality rates and effects of capture

- Identify feeding habitats and prey resources

4.A.4.8 References

4.A.4.8.1 Written References

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4.A.5 Riparian Brush Rabbit (*Sylvilagus bachmani riparius*)

4.A.5.1 Legal Status and Distribution

Riparian brush rabbit was listed as endangered under the Endangered Species Act (ESA) on February 23, 2000 (65 *Federal Register* [FR] 8881). It is also listed as endangered under the California ESA.

One of eight subspecies of brush rabbit in California, the riparian brush rabbit occupies a range that is disjunct from other brush rabbits, near sea level on the northwestern floor of the San Joaquin Valley (U.S. Fish and Wildlife Service 1998). Its historical distribution may have extended along portions of the San Joaquin River and its tributaries on the valley floor from at least Stanislaus County to the Sacramento–San Joaquin River Delta (Delta) (Orr 1935 in U.S. Fish and Wildlife Service 1998). Populations are known to have historically occurred in riparian forests on the valley floor along the San Joaquin and Stanislaus Rivers and some tributaries of the San Joaquin River (U.S. Fish and Wildlife Service 1998). One population estimate within this historical range was about 110,000 individuals (U.S. Fish and Wildlife Service 1998).

The dramatic decline of the riparian brush rabbit began in the 1940s with the building of dams constructed for irrigation and flood control on the major rivers of the Central Valley. Protection from flooding resulted in conversion of floodplains to croplands and the consequent reduction and fragmentation of remaining riparian communities. By the mid-1980s, the riparian forest within the species' former range had been reduced to a few small and widely scattered fragments totaling about 5,189 acres (2,100 hectares) (U.S. Fish and Wildlife Service 1998).

Remaining populations of riparian brush rabbits occur in only two locations in San Joaquin County. One population is at an approximately 258-acre (104-hectare) patch in Caswell Memorial State Park on the Stanislaus River immediately southeast of the action area. The other population is located at several small, isolated or semi-isolated patches immediately west and southwest of Lathrop, totaling approximately 270 acres (109 hectares) along Paradise Cut and Tom Paine Slough and channels of the San Joaquin River in the south Delta (Kelly 2015; Kelly et al. 2011; Williams et al. 2002). In addition, a captive breeding program has established a population on the Faith Ranch, which is owned by the wine-making Gallo family (U.S. Fish and Wildlife Service 2007c).

4.A.5.2 Life History and Habitat Requirements

The following are important components of riparian brush rabbit habitat.

- Large patches of dense brush composed of riparian vegetation such as blackberry (*Rubus* spp.), California wild rose (*Rosa californica*), and low-growing willows (*Salix* spp.), or other dense shrub species.

- Ecotonal edges of brushy species to grasses and herbaceous forbs.

- Scaffolding plants (dead or alive) for blackberry and rose to grow tall enough to withstand flood events.

- A tree overstory that is not closed, if present.
- High-ground refugia from flooding (Kelly et al. 2011).

Brush rabbits move through the dense brush and thickets by creating tunnels through the vegetation. Generally, riparian forests that support a closed overstory canopy lack sufficient understory shrubs to support riparian brush rabbits (U.S. Fish and Wildlife Service 1998). Small herbaceous openings in proximity to cover are also required for foraging, and higher-elevation areas are required to sustain populations during floods (U.S. Fish and Wildlife Service 1998).

Sites inhabited by riparian brush rabbits usually have a mix of wild roses, blackberries, coyote bush (*Baccharis pilularis*), and grape vines (*Vitis californica*), with high volumes of roses and coyote bush in comparison to uninhabited sites (Williams and Basey 1986; U.S. Fish and Wildlife Service 1998; Kelly et al. 2011). Williams and Basey (1986) also note that brush rabbit sites support significantly more ground litter and surface area of roses and significantly fewer willows than sites occupied by desert cottontails. This condition may indicate the presence of higher-elevation areas that are not flooded regularly or heavily, an important element of brush rabbit habitat (Williams and Basey 1986). Herbaceous forbs, such as mugwort (*Artemesia douglasiana),* stinging nettle (*Urtica dioica),* and gumplant (*Grindelia camporum),* at the edge of the brush/thicket habitat has been found to be an important habitat component for riparian brush rabbit (Kelly et al. 2011). Mugwort provides cover, food, and is flood tolerant. Gumplant forms dense stands and thus provides important cover from predators while the rabbit forages.

The average home range for riparian brush rabbit varies from year to year but is within the range of 3.1 to 7.4 acres (1.3 to 3 hectares). The average core use area is typically less than half of the home range area (1.2 to 1.9 acres [0.5 to 0.8 hectares]) (California Department of Fish and Game 2005; Chapman 1974). Home ranges generally conform to the size of the available brushy habitat (U.S. Fish and Wildlife Service 1998). Individuals are intolerant of each other when they come too close, but there is no well-defined territoriality. Young are more tolerant of approach by another rabbit than are adults (Chapman 1974; U.S. Fish and Wildlife Service 1998).

Riparian brush rabbits feed at the edges of shrub cover rather than in large openings. Their diet consists of herbaceous vegetation, such as grasses, sedges, clover, forbs and buds, bark, and leaves of woody plants. Grasses and other herbs are the most important food for brush rabbits, but shrubs such as California wild rose, coyote bush, and blackberry also are eaten. When available, green cow clover (*Trifolium wormskioldii*) is preferred over all other foods (Orr 1940; Larsen 1993; U.S. Fish and Wildlife Service 1998; Sandoval et al. 2006).

Riparian brush rabbits typically remain hidden under protective shrub cover. They seldom venture more than 1 meter from cover. They often remain motionless while searching for signs of danger before moving short distances. When pursued, they leap back into the cover of shrubs instead of heading into open ground (Chapman 1974). Williams (1988) reported that they will generally not cross large, open areas, and hence are unable to disperse beyond the dense brush of the riparian forest.

4.A.5.3 Reasons for Decline

The primary threats to the survival of riparian brush rabbit are the limited extent of its existing habitat, extremely low numbers of individual animals, and few extant populations. The small sizes of its remaining populations, the localization of the behavior of the subspecies, and the highly limited and fragmented nature of remaining habitat restrict natural dispersal and put the species at risk from a variety of environmental factors. The existing population sizes do not meet the minimum population sizes that Thomas (1990) suggests are required to assure the medium- to long-term persistence of birds or mammals (i.e., the geometric mean of population size should be 1,000 for species with normally varying numbers and about 10,000 for species exhibiting a high variability in population size). Therefore, the species is considered at a high risk of imminent extinction from several consequent threats related to population genetics, demographics, and environmental stochasticity (U.S. Fish and Wildlife Service 1998).

Flooding is a key issue for riparian brush rabbits and thought to be responsible for major population declines. Riparian brush rabbits are closely tied to brushy cover and will generally not cross large, open areas. Thus, they are unable to disperse beyond the dense brush, making them susceptible to mortality during flood events (Williams 1988; U.S. Fish and Wildlife Service 1998).

Periodic flooding still occurs along all major rivers in the Central Valley (Kindle 1984). With behavioral restrictions on its freedom of movement (low mobility) and the shortage of habitat that is suitably protected from frequent floods downstream of Caswell Memorial State Park, there is little chance that individuals escaping drowning or predation will be able to meet mates or reproduce (U.S. Fish and Wildlife Service 1998).

Wildfire also poses a major threat. Long-term fire suppression of Caswell Memorial State Park, combined with prolonged drought, has caused the buildup of high fuel loads from dead leaves, woody debris, and senescent flammable shrubs. The dense, brushy habitat to which the rabbits are restricted is thus highly susceptible to catastrophic wildfire that would cause both high mortality and destruction of habitat. Recovery of the riparian brush rabbit population from such a devastating event would be improbable (U.S. Fish and Wildlife Service 1998).

Like most rabbits, the riparian brush rabbit is subject to a variety of common contagious, and generally fatal, diseases that could be transmitted easily to riparian brush rabbits from neighboring populations of desert cottontails. For these small remnant brush rabbit populations, this kind of epidemic could quickly eliminate the entire population (Williams 1988; U.S. Fish and Wildlife Service 1998).

A wide variety of aerial and terrestrial predators prey on riparian brush rabbit, including various raptors, coyote (*Canis latrans*), gray fox (*Urocyon cinereoargenteus*), bobcat (*Lynx rufus*), long-tailed weasel (*Mustela frenata*), mink (*Neovison vison*), raccoon (*Procyon lotor*), snakes, feral dogs (*Canis lupus familiaris*), and feral cats (*Felis catus*) (Kelly et al. 2011). A robust population of the riparian brush rabbit should be able to withstand predation, but habitat adjacent to residential properties or along public roads or waterways, or subject to human disturbance, can exacerbate predation risk (Kelly et al. 2011). The black rat (*Rattus rattus*) is an exotic invasive species that may be a threat to riparian brush rabbit populations by preying on offspring and competing for resources. Black rats appear to be ubiquitous in riparian natural communities in the Central Valley (Kelly et al. 2011).

4.A.5.4 Status of the Species in the Action Area/Environmental Baseline

The south Delta population (Paradise Cut and Tom Paine Slough) of riparian brush rabbit is located south of the action area, near Mossdale. This area is on private land, and watercourses are managed for flood control, not wildlife management. Surveys conducted by the Endangered Species Recovery Program under contract with the California Department of Water Resources have not resulted in additional occurrences of riparian brush rabbit in the action area; however, surveys are incomplete because of lack of property access.

4.A.5.5 Critical Habitat

Critical habitat has not been designated for riparian brush rabbit.

4.A.5.6 Suitable Habitat Definition

As described above in Section 4.A.5.2, *Life History and Habitat Requirements*, and below in Section 4.A.5.7, *Head of Old River Gate Habitat Assessment*, suitable riparian habitat for riparian brush rabbit consists of large patches (at least 0.05 acre) of brushy understory shrub layer of valley riparian forests. Most occupied sites are in riparian settings with an open overstory canopy or savannah-like settings that support patches of low-growing wild rose, wild grape, blackberry, and coyote bush, where the brush rabbits move through the dense brush and thickets by creating tunnels through the vegetation. Riparian forests that support a closed overstory canopy generally lack sufficient understory shrubs to support riparian brush rabbits (Williams 1988; U.S. Fish and Wildlife Service 1998). Suitable grassland habitat consists of grassy patches very near to dense brush, which provide foraging opportunities near cover (Kelly et al. 2011). Riparian brush rabbit suitable habitat is geographically constrained to the mainstem of the San Joaquin Old River from Highway 4 south to the southern edge of the action area (legal Delta), on the intersection of Old River and Highway 4 south to the confluence with the mainstem of the San Joaquin River, Thomas Paine Slough, and Paradise Cut.

4.A.5.7 Head of Old River Gate Habitat Assessment

4.A.5.7.1 Habitat Assessment

A habitat assessment for the riparian brush rabbit was completed on December 18, 2015 to inform a comprehensive biological assessment. The habitat assessment was completed within and in the vicinity of the Head of Old River Gate construction footprint (see Figure 6.2-2). To enable the U.S. Fish and Wildlife Service to evaluate the project's impacts, the following information was collected.

4.A.5.7.1.1 Head of Old River Gate Project Description and Map

A description of the Head of Old River Gate (HORGate) construction project can be found in Section 3.2.8, *Head of Old River Gate*. Figure 6.2-1 shows the location of the HORgate construction area within the Delta and with respect to the location of riparian bursh rabbit occurrences. Figure 6.2-2 shows the just the vicinity of the HOR gate construction area and the associated spoils area. Photos of the proposed HOR gate construction site taken during the habitat assessment are provided in Attachment 4.A.1, *Riparian Brush Rabbit Habitat Assessment Photo Log*.

4.A.5.7.1.2 Hydrology

Among the threats to riparian brush rabbit is flooding and the complete inundation of habitat.

1. The proposed construction site for the HOR gate is on the Old River, just downstream of the confluence between the San Joaquin and Old Rivers. If Riparian Brush Rabbits were displaced from floods upstream of the proposed HOR gate construction site (there is no RBR habitat available at the HOR gate), on the San Joaquin River, there is suitable riparian brush rabbit habitat downstream on the San Joaquin River; however, there is no suitable riparian brush rabbit habitat on Old River for several miles downstream of the San Joaquin and Old River confluence where the HOR gate is located. The San Joaquin River is approximately 200 feet wide downstream of the confluence with Old River and Old River is approximately 150 feet wide downstream of the confluence with San Joaquin River, in the vicinity of the HOR gate.

2. No federal and state water flood control, storage, delivery, and export programs may affect riparian bursh rabbit habitat at the proposed HOR gate construction site because no appropriate habitat for riparian brush rabbit exists at the site. No existing regulatory measures to protect threatened or endangered fish will conflict with efforts to protect riparian brush rabbit habitat from flood or desiccation at the site because no habitat for riparian brush rabbit exists there. There are no flood and restoration easements in the project area.

3. No 100, 500, and 1,000 year floodplain will be affected by the project. No quantifiable changes in flood flows would result from the action.

4.A.5.7.1.3 Soils and geomorphology

Frequently flood-scoured and silt deposit areas may have been preferred browsing sites historically, due to the colonization of grasses and forbs to these areas. There are no flood-scoured and silt deposit areas in the proposed HOR gate construction footprint.

4.A.5.7.1.4 Vegetation: diversity, distribution, structure

Riparian forests are structurally and floristically complex vegetation communities. These forests occur in many different forms throughout the Central Valley. There is no riparian forest in the proposed HOR gate project activity area. The following list includes many of the plants which characterize RBR habitat; each species below is quantified for the construction site:

1. Overstory

    - Platanus racemosa (California sycamore) - NONE

    - Populus fremontii (Fremont cottonwood) - NONE

    - Quercus lobata (valley oak) - NONE

- Salix spp. (willow spp.) - NONE

2. Intermediate Layer

- Acer negundo subsp. californicum (box elder) - NONE
- Fraxinus latifolia (Oregon ash) - NONE
- Salix spp. (willow spp.) - few, small (Salix exigua)
- Sambucus spp. (elderberry) – NONE

3. Vines (lianas) growing through various layers

- Aristolochia californica (Dutchman's pipe vine) - NONE
- Clematis spp. (Wild clematis) - NONE
- Vitis californica (wild grape) - NONE

4. Undergrowth

- Artemisia douglasiana (Douglas' sagewort) – small patches
- Baccharis pilularis (coyote brush) - NONE
- Rhus diversiloba (poison oak) - NONE
- Rosa californica (California wild rose) - NONE
- Rubus spp. (blackberry) - NONE
- Urtica dioica (stinging nettle) - NONE

4.A.5.7.1.5 Distribution

There are no clumps of dense continuous vegetation that are 460 square meters (4,951.398 square feet) or greater. There is no *Rubus spp.* (blackberry) or *Rosa californica* (California wild rose) at the site.

4.A.5.7.1.6 Structure - Succession

There is no old growth overstory at the site. The only undergrowth shrub species at the site are small, discontinous patches of *Artemisia douglasiana* (Douglas' sagewort). By definition, there is no undergrowth vegetation because there is no overstory.

4.A.5.7.1.7 Other species

The only related species observed at the site were Jackrabbits, *Lepus californicus*.

4.A.5.7.1.8 Trapping

Trapping was not completed, or recommended for the proposed HOR gate project site, or immediately adjacent to the project site because it was determined that no habitat appropriate for riparian brush rabbit existed there.

4.A.6 San Joaquin Kit Fox (*Vulpes macrotis mutica*)

4.A.6.1 Legal Status and Distribution

San Joaquin kit fox was listed as endangered under the Federal Endangered Species Act (ESA) on March 11, 1967 (32 *Federal Register* [FR] 4001). It was listed as threatened species under the California ESA in 1971. In 2010, the U.S. Fish and Wildlife Service (USFWS) completed a 5-year review for this species, and determined that the kit fox continues to meet the definition of endangered.

San Joaquin kit fox historically occurred in alkali scrub/shrub and arid grasslands throughout the level terrain of the San Joaquin Valley floor from southern Kern County north to Tracy in San Joaquin County, and up into more gradual slopes of the surrounding foothills and adjoining valleys of the interior Coast Range (U.S. Fish and Wildlife Service 2010: 1)

By 1998, when the *Recovery Plan for Upland Species of the San Joaquin Valley* (U.S. Fish and Wildlife Service 1998) was completed, local surveys, research projects, and incidental sightings indicated that kit fox inhabited a portion, but not all, of the areas of suitable habitat remaining in the San Joaquin Valley and lower foothills of the coastal ranges, Sierra Nevada, and Tehachapi Mountains. The boundaries of the kit fox's range still extended from southern Kern County north to Contra Costa, Alameda, and San Joaquin Counties on the west, and to the La Grange area, Stanislaus County, on the east side of the Valley. The largest extant populations were known from western Kern County on and around the Elk Hills area and Buena Vista Valley, and the nearby Carrizo Plain Natural Area where relatively level terrain is separated by narrow rugged ranges (U.S. Fish and Wildlife Service 1998:124-125, 2010:11).

Currently, the entire range of the kit fox appears to be similar to what it was at the time of the 1998 Recovery Plan; however, population structure has become more fragmented, and at least some of the resident satellite subpopulations, such as those at Camp Roberts, Fort Hunter Liggett, Pixley National Wildlife Refuge (NWR), and the San Luis NWR, have apparently been locally extirpated, and portions of the range now appear to be frequented by dispersers rather than resident animals (U.S. Fish and Wildlife Service 2010:15).

4.A.6.2 Life History and Habitat Requirements

Natural habitats for San Joaquin kit fox include alkali sink, alkali flat, and grasslands (U.S. Fish and Wildlife Service 2010:19–20). Agricultural lands do not provide long-term suitable habitat for kit fox for a variety of reasons. Lands with row crops are subjected to weekly inundation during irrigation, which impedes kit fox foraging and precludes the establishment, maintenance, and use of earthen dens. Prey abundance is relatively low in row crops, and when land is converted to agricultural uses, prey diversity is reduced, prey species composition changes, and favored prey species such as kangaroo rats disappear. Although kit fox may enter the margins of row crops and farther into orchards at night from natural lands, researchers found no evidence that kit fox were able to use farmland, even when it was the predominant available habitat (U.S. Fish and Wildlife Service 2010:21).

In the northern part of the range, San Joaquin kit fox is associated primarily with foothill annual grasslands (Swick 1973; Hall 1983; Bell 1994) and sometimes with valley oak savanna and alkali grasslands (Bell 1994). In the central and southern portions of the range, kit foxes are also found in remnant patches of native valley floor scrubland (e.g., valley sink scrub, valley saltbush scrub, upper Sonoran subshrub, interior Coast Range saltbush scrub), as well as grazed grasslands, agricultural lands, petroleum fields, and some urban areas (U.S. Fish and Wildlife Service 1998).

Dens are typically in relatively flat terrain or in gently sloping hills, in washes, drainages, and roadside berms. Occupied habitats are usually associated with loose-textured soils to facilitate den construction (Grinnell et al. 1937; Egoscue 1962; Morrell 1972). Shallow soils with close proximity to bedrock, soils with high water tables, and impenetrable hardpan layers are generally avoided (Morrell 1972; O'Farrell and Gilbertson 1979; O'Farrell et al. 1980; McCue et al. 1981). However, kit foxes will also modify burrows dug by other animals, such as California ground squirrel (*Otospermophilus beecheyi*, formerly *Spermophilus beecheyi*). Frequently in the northern end of their range, dens may be found in soils with high clay content (Orloff et al. 1986).

The breeding season begins during September and October when adult females begin to clean and enlarge natal or pupping dens. Mating and conception occur between late December and March, and litters of two to six pups are born between late February and late March. (U.S. Fish and Wildlife Service 1998:126.)

The home ranges of San Joaquin kit foxes are extensive and vary by location. Home range size is thought to be related to prey abundance, and studies have shown that mean home range size varies from 1,072 to 5,782 acres. San Joaquin kit foxes appear to disperse readily, with dispersal distances varying greatly (1.1 to 50 miles; these were observed in studies from relatively large areas with little development). Successful dispersal appears to be a key factor for the recovery and survival of kit fox, partly because kit fox populations are becoming more fragmented, and successful dispersal among subpopulations helps to maintain genetic diversity, save declining populations, and prevent extinction (U.S. Fish and Wildlife Service 2010:6, 107–108).

San Joaquin kit fox diet varies geographically, seasonally, and annually based on variation in abundance of potential prey (U.S. Fish and Wildlife Service 1998). In the southern and central portions of their range, kangaroo rats, pocket mice, white-footed mice (*Peromyscus* spp.), and other nocturnal rodents are key prey items. California ground squirrels, black-tailed hares (*Lepus californicus*), San Joaquin antelope squirrels (*Ammospermophilus nelsoni*), desert cottontails (*Sylvilagus audubonii*), ground-nesting birds, and insects are also taken (Jensen 1972; Archon 1992; Utah Division of Wildlife Resources 2010). In the northern part of their range, kit foxes most frequently consume California ground squirrels (Orloff et al. 1986). Cottontails, black-tailed hares, pocket mice, and kangaroo rats are also eaten (Hall 1983).

4.A.6.3 Reasons for Decline

Habitat loss and fragmentation from urbanization and agricultural expansion are the principal factors in the decline of the San Joaquin kit fox in the San Joaquin Valley (Laughrin 1970; Jensen 1972; Morrell 1975; Knapp 1978). By 1979, an estimated 6.7% of the San Joaquin Valley floor's original native habitat south of Stanislaus County remained untilled and undeveloped (U.S. Fish and Wildlife Service 1983). Cypher et al. (2013) estimated that only 4,267 square kilometers of high suitable habitat and 5,569 square kilometers of medium suitable habitat remain, with much of the habitat highly fragmented. The majority of these habitat areas were located in the southern portion of the kit fox range, with 67 and 35% of this high and medium suitable habitat occurring in Kern and San Luis Obispo Counties, respectively. In the northern range, continued urbanization, primarily in Contra Costa and Alameda Counties, water storage and conveyance projects, road construction, energy development, and other activities continue to reduce and fragment remaining grassland habitats. These land conversions contribute to kit fox declines through displacement, isolation of remaining populations, creation of barriers to movement, mortality, and a reduction of prey populations (U.S. Fish and Wildlife Service 1998).

Although livestock grazing is not necessarily detrimental and, in fact, may be beneficial (Morrell 1975; Orloff et al. 1986), intensive overgrazing that destroys shrub cover and reduces prey abundance may be detrimental (O'Farrell et al. 1980; O'Farrell and McCue 1981; U.S. Fish and Wildlife Service 1983).

The use of pesticides and rodenticides also threatens kit foxes. Ground squirrel control programs in the 1970s severely reduced California ground squirrel populations in Contra Costa County and are thought to have contributed to kit fox declines in the northern range (Bell et al. 1994; U.S. Fish and Wildlife Service 1998). Kit fox is also susceptible to secondary poisoning from rodenticides (Standley et al. 1992).

Predation of San Joaquin kit foxes by coyotes (*Canis latrans*), bobcats (*Lynx rufus*), and nonnative red fox (*Vulpes vulpes*) has also contributed to the decline of San Joaquin kit fox. Coyotes and red foxes also compete with kit foxes for the same prey (U.S. Fish and Wildlife Service 2010: 7-8).

4.A.6.4 Status of the Species in the Action Area/Environmental Baseline

Available occurrence data indicates that the density of the San Joaquin kit fox population north of Santa Nella is very low; kit fox in the Northern Range have either experienced extirpation or have fallen below detectable numbers (Clark, et al 2007). The population density north of I-580 along the east coast range foothills is extremely low, if the species has not been extirpated from that area altogether. Orloff et al. (1986) found kit fox in Alameda and San Joaquin counties, but were unable to document the presence of kit foxes in Contra Costa County (Smith, et al 2006).

From 1991 to 1992, Bell and Ralls observed kit foxes at 3 sites in Contra Costa County, and 1 site in San Joaquin County, and a possible kit fox track was recorded at one site that encompassed both Alameda and San Joaquin counties. However, subsequent work in Alameda and Contra Costa counties with baited camera stations on public land and spotlight surveys on roads through potential kit fox habitat found no evidence of kit fox presence, even in areas where they had been documented earlier (Smith, et al 2006).

Smith et. al. (2006) surveyed 213 km within 24 properties in Alameda, Contra Costa, and San Joaquin counties using trained scat detection dogs, a proven effective survey technique for San Joaquin kit fox. Additionally, aircraft surveys were conducted to locate dens. No evidence for kit fox was found in the northern range. The study concluded that kit fox occur in the northern range in extremely low densities or only intermittently, if they have not been extirpated (Smith et al 2006). Currently, kit fox observations in the Northern Range are rare and no populations are known to occur there (Cypher et al 2013).

In February 2003, the Endangered Species Recovery Program surveyed DWR's property using scat detection dogs, including DWR land north of the intake channel, around Clifton Court Forebay, around Banks PP, and along the California Aqueduct to the south extent of Bethany Reservoir. No kit fox sign was observed and no kit fox scats were found.

In 1992 and 1993, DWR staff surveyed a 500 foot corridor from Clifton Court Forebay and Old River and along the South Bay Aqueduct to the city of Fremont. Several hundred burrows large enough to be classified as potential kit fox dens were identified. Using track medium, the burrows were monitored for 3 consecutive days. No kit fox tracks were observed at any of the burrows or anywhere in the alignment, and no other sign of kit fox were observed. (Bradbury, unpubl data).

In 1994, DWR and CDFG completed spotlight and camera surveys around Clifton Court Forebay, along the Banks Pumping Plant intake channel, along the length of the California Aqueduct to Patterson, CA, and along the length of the South Bay Aqueduct through Livermore. Additionally, because culverts are often used as artificial dens, every culvert along the California Aqueduct and Southbay Aqueduct in those same areas were searched for kit fox; culverts occur approximately every 1/10 mile. No San Joaquin kit fox were observed or photographed (Bradbury unpubl data). In Kern County, San Joaquin kit fox are readily observed and photographed along the California Aqueduct, and often use culverts for artificial dens (Bradbury pers obs 1989-2013).

There are limited records of San Joaquin Kit Fox in the CNDDB for the species' northern range, and only 28 records of the species north of I-580/205, which span almost 50 years; many are questionable in reliability relative to location accuracy and identification. Clark et al. 2007 analyzed CNDDB records of San Joaquin kit foxes and their results indicate that many of the records may be misidentification of coyote pups. Most of the records from the northern range are more than 30 years old and were apparently re-creations of recalled occurrences, and at least some have factual errors.

An example of a likely factual error is record #561 from 1987, which states that the fox was observed near a wind generator, but there have been no wind generators in the area delineated for the occurrence. Additionally, only 2 records are of kit fox in agricultural areas (based on occurrence delineation and description of habitat):

1. "1 juvenile kit fox observed during daylight in Jun 1991" in an agricultural field north of the town of Byron (record #575); it is unlikely that a juvenile kit fox would be away from its den at such a young age, especially during the day;
2. One along an Old River levee in 1991 (record #60), based on a print on a track pad; it is unlikely a kit fox would be in a riparian zone almost 3 miles from suitable grassland habitat. Neither record is confirmed by follow-up surveys.

Based on the description of the sighting on number 1, and the location and basis for number 2, both records have a high potential to be identification error.

There are just 5 records for kit fox north of I-580/205 in the last 20 years, although there have been numerous surveys completed during that time. Two records are based on tracks, with no apparent confirmation through follow-up surveys.

Only one record is of kit fox in an area consistent with the project location and habitat type: record #34 adjacent to the Tracy Pumping Plan intake. This record well indicates the likelihood of mistaken records in the CNDDB from observers unfamiliar with the species:

- Observer indicates there were 40 dens in what is approximately a 3 acre area, including approximately 10 “recent dens.”
- Observer notes hearing a “yip”, indicating a kit fox was present.
- Observer concludes that the small area supports a small population of kit fox, for several years.
- Observer cites observations of kit fox by Western Area Power Administration employees.

What the observer is describing is a cluster of holes created by a colony of California ground squirrels, with potentially a coyote or red fox in the area, based on the following.

- The observer is obviously counting holes, not dens. Ten “recent [kit fox] dens” in an area that size is highly unlikely; kit fox are not colonial and dens are spread among very large areas.
- An observer familiar with the species would know that kit fox have a very distinct “roop” call; a “yip” is more characteristic of a red fox or coyote.
- Kit fox are not communal like ground squirrels; the small area would not support a “small population” of kit fox.
- Non-biologists regularly mistake red foxes and young coyotes for kit foxes (pers ob). Red foxes and coyotes are much more likely to be active during the day than kit fox, when workers are likely to see them. Biologists without sufficient experience with kit foxes will also sometimes mistake coyote pups with kit foxes, as coyote pups can look remarkably similar to adult kit foxes (Clark et al. 2007).

On February 4, 2016, DWR staff with kit fox life history expertise surveyed the site; there were approximately 30 burrow holes, and 6 showed signs of recent excavation, but all were too small for kit fox use and were obviously ground squirrel burrows. Canid scats was observed at two locations in the immediate area but were too large for kit fox, and were identified as red fox scat. The conclusion based on the above analysis is that the record is unreliable.

On June 30, 2016, California Department of Fish and Wildlife indicated that some experts believe San Joaquin kit fox may still occur in the action area (pers. comm. Brooke Jacobs).

4.A.6.5 Critical Habitat

Critical habitat has not been designated for San Joaquin kit fox.

4.A.6.6 Suitable Habitat Definition

As described above in Section 4.A.6.2, *Life History and Habitat Requirements*, and below in Section 4.A.6.7, *Species Habitat Suitability Model*, suitable habitat for San Joaquin kit fox in the action area consists of grasslands, vernal pool complex, and alkali seasonal wetland complex with burrows in the area shown on Figure 6.3-1.

4.A.6.7 Species Habitat Suitability Model

A habitat suitability model was not used to assess effects on San Joaquin kit fox. The USFWS and California Department of Fish and Wildlife visited the area around Clifton Court Forebay during the summer of 2016, and determined that the area mapped as California tiger salamander habitat in this region corresponds with the area that provided suitable habitat characteristics for San Joaquin kit fox. Figure 6.3-1 depicts this area.

4.A.7 California Least Tern (*Sternula antillarum browni*)

4.A.7.1 Legal Status and Distribution

The California least tern is listed as endangered under the state and Federal endangered species acts. The species was listed by the California Fish and Game Commission pursuant to California's Endangered Species Act (ESA) (Fish and Game Code, Sections 2050 *et seq.)* on June 27, 1971, and by the U.S. Fish and Wildlife Service (USFWS) pursuant to the Federal ESA on October 13, 1970 (35 *Federal Register* [FR] 8491). The California least tern is also designated as a state Fully Protected species. Critical habitat has not been designated for this species.

The historical breeding range of the California least tern extends along the Pacific Coast from approximately Moss Landing to the southern tip of Baja California (Grinnell and Miller 1944). However, since about 1970, colonies have been reported north to San Francisco Bay (U.S. Fish and Wildlife Service 2006d). The nesting range in California is somewhat discontinuous as a result of the availability of suitable estuarine shorelines, where California least terns often establish breeding colonies. Marschalek (2006) identified six geographic population clusters along the Pacific Coast in California, including San Diego, Camp Pendleton, Los Angeles/Orange County, Ventura County, San Luis Obispo/Monterey County, and San Francisco Bay. The majority of the California population is concentrated in three counties: San Diego, Orange, and Los Angeles.

Statewide surveys in 2010 estimated a minimum of 6,437 breeding pairs, with about 85% of the breeding colonies occurring in southern California and only a small percentage (6.3% or 406 breeding pairs) occurring in the San Francisco Bay Area (Marschalek 2011). Statewide, the growth of the breeding population has been dramatic since state and Federal listing of the California least tern, from only several pairs in the late 1960s to a current minimum of 6,437 pairs (Marschalek 2011). Marschalek (2011) reported on monitoring activities at six active breeding colonies in the San Francisco Bay Area in 2010, with a total number of breeding pairs estimated at approximately 406.

4.A.7.2 Life History and Habitat Requirements

California least terns nest in loose colonies on barren or sparsely vegetated sandy or gravelly substrates above the high tide line along the coastline and in lagoons and bays of the California coast. Colonies are always near water that provides foraging opportunities. Foraging typically occurs in shallow estuaries or lagoons (Thompson et al. 1997; U.S. Fish and Wildlife Service 2006d).

California least terns are migratory and are present at nesting areas from mid-April to late September (Anderson and Rigney 1980; Patton 2002). Courtship generally occurs during April and May and usually takes place away from the nesting area on exposed tidal flats or beaches. Nesting begins by mid-May (Massey 1981). Clutch size ranges from one to four eggs but usually consists of two or three eggs, with a single brood raised each year. Incubation is usually 20 to 25 days, and young are fledged by 28 days. The young will continue to depend on adults for an additional 2 weeks (Rigney and Granholm 2005). Wintering areas are largely unknown, but are suspected to be along the Pacific Coast of Central and South America (Massey 1977). In the San Francisco Bay Area and Suisun Bay, nesting colonies are typically located in abandoned salt ponds and along estuarine shores, often using artificially or incidentally created habitat (Rigney and Granholm 2005; Marschalek 2008). Foraging occurs in the bay or large river estuaries.

California least terns select nesting colony sites that are free of human or predatory disturbance and are located in proximity to a foraging area. The availability of such sites is a limiting factor for the species. California least terns roost on the ground. Nest sites are shallow depressions without nesting material, typically in barren sandy or gravelly substrate. Prior to egg-laying, adults generally roost away from nest sites, from 0.25 mile at coastal sites to several miles at estuarine sites. This behavior is thought to be a form of predator avoidance (U.S. Fish and Wildlife Service 2006d).

California least terns are very gregarious and nest, feed, roost, and migrate in colonies. They are highly sensitive to nest disturbance and will readily abandon nest sites if disturbed (Davis 1974, as cited in Rigney and Granholm 2005).

The California least tern feeds in shallow estuaries and lagoons for small fish, including anchovies (*Engraulis* spp.), silversides (*Atherinops* spp.), and shiner surfperch (*Cymatogaster aggregata*) (Rigney and Granholm 2005). It hovers above the water, then plunges but does not completely submerge. It will also forage in the shallow tidal zone of the open ocean and in bays (Rigney and Granholm 2005).

4.A.7.3 Reasons for Decline

The loss, degradation, and disturbance of suitable coastal strand and estuarine shoreline habitat is the primary reason for the historical reduction of California least tern populations. Most extant colonies occur on small patches of degraded nesting habitat surrounded on all sides by human activities. The majority of colony sites are in areas that were incidentally created during development projects. Further expansion and recovery of the California least tern population may require the creation or restoration of nesting habitat (U.S. Fish and Wildlife Service 2006d).

Human disturbance was noted as early as the mid-1920s as a factor in causing colony abandonment and population declines (Rigney and Granholm 2005), and is still considered a major threat to remaining colonies (Garrett and Dunn 1981; Marschalek 2009). There is no suitable natural habitat in California that is free of development, military, or recreation-related human disturbances; thus, opportunities for the species to develop new breeding territories are mostly restricted to artificially or incidentally created habitat. Fencing has been used to prohibit entry into colony sites, but this also restricts the movement of birds. Lack of fencing or damage to existing fencing has led to nesting failures (U.S. Fish and Wildlife Service 2006d).

Predation is regarded as the most significant threat to existing colonies. Marschalek (2011) reports 47 vertebrate and invertebrate predators or suspected predators of California least tern colonies in 2010. Most depredated tern chicks were taken by gull-billed terns (*Gelochelidon nilotica,* formerly *Sterna nilotica*). Common ravens (*Corvus corax*), coyotes (*Canis latrans*), and American crows (*Corvus brachyrhynchos*) had the highest depredation rate of eggs while peregrine falcons (*Falco peregrinus*) and unknown avian species had the highest depredation rate of fledglings and adults. Marschalek (2011) calculated that 1,007 eggs, 340 chicks, 161 fledglings, and 115 to 129 adults were lost to predation events in 2010.

4.A.7.4 Status of the Species in the Action Area/Environmental Baseline

Recently, seven California least tern nesting sites have been reported from the vicinity of the Delta, two of which (Montezuma Hills and Pittsburg Power Plant) are in the action area (Marschalek 2011). California least terns have nested at the Montezuma Wetlands on the eastern edge of Suisun Marsh near Collinsville since 2006. This colony site was unintentionally created as part of a wetlands restoration project that requires increasing the elevation of certain areas prior to flooding (Marschalek 2008). A pile of sand and shells, formed during excavation of the wetland restoration site, attracted terns to the site, which to date has prevented completion of the restoration project. Marschalek (2011) reports 23 breeding pairs (0.036%), 17 nests, and at least five fledglings from this breeding colony in 2010. California least terns also recently began nesting at the Pittsburg Power Plant in Pittsburg, although with less success. In 2010, Marschalek (2011) documented no breeding pairs at this site. This was the third time in the last 4 years that least terns did not nest at this site.

Two additional locations were recently reported from just outside the action area, including Napa Sonoma Marsh Wildlife Area—Green Island Unit on the Napa River east of the San Pablo Bay National Wildlife Refuge and northwest of American Canyon, where 47 breeding pairs and 47 nests producing 85 fledglings were reported in 2010 (Marschalek 2011); and along a gravel road between two treatment ponds at the Sacramento Regional Wastewater Treatment Plant (Bufferlands) east of I-5, where a single successful nest was documented in 2010 (Marschalek 2011) and in 2016.

There is one record of a California least tern foraging in the Clifton Court Forebay from 1994 (Yee et al. 1995). However, California least tern is not expected to be foraging at the forebay because it is 20 miles from the nearest nesting site (Pittsburg), and the typical foraging habitat for California least tern is within 2 miles of their colonies (Atwood and Minsky 1983).

4.A.7.5 Critical Habitat

Critical habitat has not been designated for California least tern.

4.A.7.6 Suitable Habitat Definition

As described above in Section 4.A.7.2, *Life History and Habitat Requirements*, and below in Section 4.A.7.7, *Species Habitat Suitability Model*, suitable foraging habitat for California least tern includes all of the tidal perennial aquatic natural community within the action area. Suitable nesting habitat includes barren or sparsely vegetated gravelly substrates which are unlikely to occur within the Delta due to its highly altered landscape.

4.A.7.7 Species Habitat Suitability Model

4.A.7.7.1 GIS Model Data Sources

The California least tern model uses vegetation types and associations from the following data sets: composite vegetation layer (Hickson and Keeler-Wolf 2007 [Delta]; Boul and Keeler-Wolf 2008 [Suisun Marsh]; TAIC 2008 [Yolo Basin]), aerial photography (U.S. Department of Agriculture 2005), and land use survey of the Sacramento–San Joaquin River Delta and Suisun Marsh area (Version 3) (California Department of Water Resources 2007). Using these data sets, the model maps the distribution of suitable riparian brush rabbit habitat in the action area. Vegetation types were assigned based on the species requirements as described above and the assumptions described below.

4.A.7.7.2 Breeding and Foraging Habitat Model Description

Modelled foraging habitat includes all areas mapped as tidal perennial aquatic. Nesting habitat (barren or sparsely vegetated sandy or gravelly substrates above the high tide line along the coastline) is not mapped but has potential to occur in along the perimeter of large water bodies. However, the potential for occurrence of the necessary substrate in the Delta is very unlikely due to the highly modified nature of the Delta.

4.A.7.7.3 Assumptions

- **Assumption:** California least tern habitat in the action area is geographically constrained to areas described in Section 4.A.7.7.2, *Breeding and Foraging Habitat Model Description.*

  Rationale: As evidenced by recent breeding occurrences at the Montezuma Wetlands, adjacent to the Grizzly Island Wildlife Area, the Pittsburg Power Plant in the City of Pittsburg, and the Bufferlands associated with the Sacramento County Wastewater Treatment Plant in the City of Elk Grove, the California least tern has potential to nest in shoreline habitat adjacent to large permanent water bodies within the action area. It is assumed that continued range expansion could occur in association with suitable tidal perennial aquatic habitat throughout the action area. Although most of the shoreline habitat has been modified or is artificial, nesting colonies are often in artificially or incidentally created habitat (Rigney and Granholm 2005; Marschalek 2008) such as gravel roads, debris piles, and other conditions that mimic a natural sandy or gravelly substrate. It is assumed that foraging can occur in large river estuaries, such as the Sacramento and San Joaquin Rivers, and other tidal perennial aquatic habitat throughout the action area. However, because little if any natural nesting habitat occurs and future breeding occurrences may occur incidentally around these water bodies, it is not possible to accurately determine locations of suitable breeding habitat. Therefore, it is assumed that breeding sites could occur in the future adjacent to tidal perennial aquatic habitat.

4.A.8 Western Yellow-billed Cuckoo (Coccyzus americanus occidentalis)

4.A.8.1 Legal Status and Distribution

The Western distinct population segment (DPS) of the yellow-billed cuckoo was listed as threatened under the Federal Endangered Species Act (ESA) on October 2, 2014 (79 *Federal Register* [FR] 59991–60038). Western yellow-billed cuckoo is also listed as an endangered species under the California ESA.

The historical distribution of yellow-billed cuckoo extended throughout the Central Valley, where Belding (1890) considered the species common. In the mid-1940s, Grinnell and Miller (1944) still considered the Central Valley distribution to extend from Bakersfield to Redding.

Currently, the only known populations of breeding western yellow-billed cuckoo are in several disjunct locations in California, Arizona, and western New Mexico (Halterman 1991; Johnson et al. 2007; Dettling et al. 2015; Stanek 2014; Parametrix Inc. and Southern Sierra Research Station 2015). Yellow-billed cuckoos winter in South America from Venezuela to Argentina (Hughes 1999; Sechrist et al. 2012) after a southern migration that extends from August to October (Laymon 1998). They migrate north and arrive at California breeding grounds between May and July, but primarily in June (Gaines and Laymon 1984; Hughes 1999; 78 FR 61621).

Studies conducted in 1986 and 1987 indicate that there were approximately 31 to 42 pairs in California (Laymon and Halterman 1987) at that time. Although a few occurrences have been detected elsewhere recently, including near the Eel River, the only locations in California that currently sustain breeding populations include the Colorado River system in southern California, the South Fork Kern River east of Bakersfield, and isolated sites along the Sacramento River in northern California (Laymon and Halterman 1989; Laymon 1998; Halterman 2001; Hammond 2011; Dettling et al. 2014; Stanek 2014; Parametrix Inc. and Southern Sierra Research Station 2015).

4.A.8.2 Life History and Habitat Requirements

The western yellow-billed cuckoo is a riparian obligate species. Its primary habitat association is willow-cottonwood riparian forest, but other tree species such as white alder (*Alnus rhombifolia*) and box elder (*Acer negundo*) may be an important habitat element in some areas, including occupied sites along the Sacramento River (Laymon 1998). Nests are primarily in willow (*Salix* spp.) trees; however, other tree species are occasionally used, including Fremont cottonwood (*Populus fremontii*) and alder. Along the Sacramento River, orchards of English walnut (*Juglans regia*), prune, and almond trees have also been reportedly used for nesting (Laymon 1980). Occupied habitat in Butte County was described by Halterman (1991) as great valley cottonwood riparian forest and great valley mixed riparian forest, including willows, box elder, and white alder. Potential habitat also occurs in valley marshland with willow riparian corridors, such as that found in the Llano Seco area of Butte County.

On the Santa Ana River, nest site height in willow trees averaged 14 feet, but on the Sacramento River, a nest in a cottonwood tree was reported at 100 feet and canopy cover is typically dense (averaging 96.8% at the nest). Patch size was found to be the most important habitat variable to predict presence of western yellow-billed cuckoos on the Sacramento River (Girvetz and Greco

2009). Large patch sizes (20 to 40 hectares, with a minimum width of 100 meters) are typically required for cuckoo occupancy (Laymon 1998; Riparian Habitat Joint Venture 2004).

Although western yellow-billed cuckoos nest primarily in willow trees, Fremont cottonwood trees are important foraging habitat, particularly as a source of insect prey. All studies indicate a highly significant association with relatively expansive stands of mature cottonwood-willow forests; however, western yellow-billed cuckoos will occasionally occupy a variety of marginal habitats, particularly at the edges of their range (Laymon 1998). Continuing habitat succession has also been identified as important in sustaining breeding populations (Laymon 1998). Meandering streams that allow for constant erosional and depositional processes create habitat for new rapidly growing young stands of willow, which create preferred nesting habitat conditions for western yellow-billed cuckoo. Lateral channel migration and point bar deposition that create new floodplains and channel bend cut-offs that create floodplain lakes are important processes that create viable western yellow-billed cuckoo habitat (Greco 2013).

A habitat model developed by Gaines (1974) for the yellow-billed cuckoo in the Sacramento Valley includes the following elements: patch size of at least 25 acres, at least 330 feet wide and 990 feet long, within 330 feet of surface water, and dominated by cottonwood/willow gallery forest with a high-humidity microclimate. Laymon and Halterman (1989) further refined the model by classifying habitat patch sizes for suitability. A willow-cottonwood forest patch greater than 1,980 feet wide and greater than 200 acres (81 hectares) is classified as optimum habitat; a patch 660 to 1,980 feet wide and 102.5 to 200 acres (41.5 to 81 hectares) is suitable; a patch 330 to 660 feet wide and 50 to 100 acres (20 to 40 hectares) is marginal, and smaller patches are unsuitable. The Riparian Habitat Joint Venture recommends restoring habitat in 25 locations to support 625 pairs (25 pairs per location) (Riparian Habitat Joint Venture 2004). Predictions suggest that a minimum of at least 25 pairs in a subpopulation, with interchange with other subpopulations, should be relatively safe from extirpation (Riparian Habitat Joint Venture 2004). To achieve this goal for the Sacramento Valley, it would be necessary to establish or preserve at least 6,070 hectares (15,000 acres) of optimum and suitable habitat. As of 1998, only 2,367 hectares (5,850 acres) of habitat were considered suitable (Laymon 1998).

Limited information is available on home range and territory size. Territory size at the South Fork Kern River ranged from 20 to 100 acres (8 to 40 hectares) (Laymon 1998), and on the Colorado River as small as 10 acres (4 hectares) (Laymon and Halterman 1989). Patch size, type and quality of habitat, and prey abundance largely determine the size of territories (Halterman 1991). Laymon and Halterman (1989) concluded that sites greater than 200 acres in extent and wider than 1,950 feet were optimal and sites 101 to 200 acres in extent and wider than 650 feet were suitable.

4.A.8.3 Reasons for Decline

Historical declines of the western yellow-billed cuckoo are attributed to the removal of riparian forests in California for agricultural and urban expansion. Habitat loss and degradation continue to be the most significant threats to remaining populations. Habitat loss continues as a result of bank stabilization and flood control projects, urbanization along edges of watercourses, agricultural activities, and river management that alter flow and sediment regimes. Nesting cuckoos are also sensitive to habitat fragmentation that reduces patch size (Hughes 1999).

Pesticide use associated with agricultural practices may affect behavior and cause death or potentially affect prey populations (Hughes 1999). Predation is a significant source of nest failures, which have been recorded at 80% in some areas (Hughes 1999). Fragmentation of occupied habitats could make nest sites more accessible and more vulnerable to predation. Nestlings and eggs are vulnerable to predation by snakes, small mammals, and birds.

4.A.8.4 Status of the Species in the Action Area/Environmental Baseline

Although there are only two historical records in the vicinity of the action area (California Department of Fish and Wildlife 2013), the species is known to have been historically common in riparian habitat throughout the Central Valley, from Kern County north to Redding (Laymon 1998). In 2013, there were two unconfirmed audible occurrences along the American River Parkway approximately five miles from the action area. These two occurrences were less than five miles apart along the river and heard on the same day (EBird 2015). In 2015 there was a confirmed visual occurrence along the American River located in proximity to both the 2013 occurrences and approximately five miles from the action area (EBird 2015).

There are no recently confirmed western yellow-billed cuckoo breeding locations in the action area. In summer 2009, the California Department of Water Resources detected one and possibly two yellow-billed cuckoos in a remnant patch of riparian forest in the vicinity of Delta Meadows (Delta Habitat Conservation and Conveyance Program 2011). Breeding status was not confirmed. The two historic sightings and the two recent sightings of yellow-billed cuckoo in the vicinity of the action area are presumed to be migrating birds.

Most riparian corridors in the action area do not support sufficiently large riparian patches or the natural, geomorphic processes that provide suitable cuckoo breeding habitat (Greco 2013). The species likely continues to migrate along the Sacramento River and other drainages to northern breeding sites in the Sutter Basin and Butte County. There are several remnant riparian patches in the vicinity of Mandeville and Medford Islands that provide riparian vegetation suitable for cuckoos, but do not provide sufficiently large patch size to support breeding cuckoos. Thre have been very few occurrences of western yellow-billed cuckoo in the action area, and birds found were migrating through.

4.A.8.5 Critical Habitat

Designation of critical habitat for the Western DPS of yellow-billed cuckoo was published in the *Federal Register* on August 15, 2014 (57 FR 48547-48652). There is no designated critical habitat for the Western DPS of yellow-billed cuckoo in the action area.

4.A.8.6 Suitable Habitat Definition

The western yellow-billed cuckoo habitat model described in Section 4.A.8.7.2, *Habitat Model Description*, uses existing, alliance-level vegetation data to identify suitable migratory habitat for western yellow-billed cuckoo. Suitable habitat for western yellow-billed cuckoo consists of riparian forest; no minimum patch size or minimum vegetation stature has been established for migratory use.

4.A.8.7 Species Habitat Suitability Model

4.A.8.7.1 GIS Model Data Sources

The western yellow-billed cuckoo habitat model uses vegetation types and associations from the following data sets: composite vegetation layer (Hickson and Keeler-Wolf 2007 [Delta]; Boul and Keeler-Wolf 2008 [Suisun Marsh]; TAIC 2008 [Yolo Basin]), and aerial photography (U.S. Department of Agriculture 2005, 2010). Using these data sets, the model maps the distribution of suitable western yellow-billed cuckoo migratory habitat in the action area. Vegetation types were assigned based on the species requirements as described above and the assumptions described below.

4.A.8.7.2 Habitat Model Description

The migratory habitat model in the Delta include the following valley/foothill riparian vegetation types from the composite vegetation layer.

- Fremont cottonwood (*Populus fremontii*)
- White alder (*Alnus rhombifolia*)
- Oregon ash (*Fraxinus latifolia*)
- Box elder (*Acer negundo*)
- Hinds' walnut (*Juglans hindsii*)
- Black willow (*Salix gooddingii*)
- Arroyo willow (*Salix lasiolepis*)
- Shining willow (*Salix lucida*)
- Narrow-leaf willow (*Salix exigua*)
- *Alnus rhombifolia/Salix exigua* (*Rosa californica*)
- *Alnus rhombifolia/Cornus sericea*
- *Acer negundo–Salix gooddingii*
- *Salix gooddingii–Populus fremontii* (*Quercus lobata–Salix exigua–Rubus discolor*)
- *Salix gooddingii/Rubus discolor*
- *Salix lasiolepis*–mixed brambles (*Rosa californica–Vitis californica–Rubus discolor*)
- *Salix exigua*–(*Salix lasiolepis–Rubus discolor–Rosa californica*)

- *Salix gooddingii*/wetland herbs
- *Salix gooddingii–Quercus lobata*/wetland herbs
- *Salix lasiolepis–Cornus sericea/Schoenoplectus*[5] spp. (*Phragmites australis–Typha* spp.) complex unit
- *Cornus sericea–Salix exigua*
- *Cornus sericea–Salix lasiolepis*/(*Phragmites australis*)
- *Quercus lobata/Rosa californica* (*Rubus discolor–Salix lasiolepis/Carex* spp.)
- *Quercus lobata–Acer negundo*
- *Quercus lobata – Alnus rhombifolia* (*Salix lasiolepis–Populus fremontii–Quercus agrifolia*)

In 2011, and again in 2012, the species habitat models were updated to include previously unmapped portions of the action area. For most areas newly mapped, vegetation data were not available at the alliance level as in the rest of the action area and so most of the new analysis areas were mapped at the natural community level.

4.A.8.7.3 Assumptions

- **Assumptions:** Western yellow-billed cuckoo habitat is restricted to the vegetation types described in Section 4.A.8.7.2, *Habitat Model Description*.

  Rationale: The western yellow-billed cuckoo is a riparian obligate species. Its primary habitat association is willow-cottonwood riparian forest, but other species such as alder (*Alnus rhombifolia*) and box elder (*Acer negundo*) may be an important habitat element in some areas, including occupied sites along the Sacramento River (Laymon 1998).

[5] Formerly known as *Scirpus*.

4.A.9 Giant Garter Snake (*Thamnophis gigas*)

4.A.9.1 Legal Status and Distribution

Giant garter snake was listed as threatened under the Federal Endangered Species Act (ESA) on October 20, 1993 (58 *Federal Register* [FR] 54033). Giant garter snake is also listed as threatened under the California ESA. The *Draft Recovery Plan for the Giant Garter Snake* was completed in 1999 (U.S. Fish and Wildlife Service 1999b) and a 5-year review was completed in 2012 (U.S. Fish and Wildlife Service 2012). The U.S. Fish and Wildlife Service (USFWS) prepared a revised draft recovery plan for the giant garter snake, published in 2015 (USFWS 2015).

Occurrence records indicate that giant garter snakes are distributed in 13 unique population clusters coinciding with historical flood basins, marshes, wetlands, and tributary streams of the Central Valley (Hansen and Brode 1980; Brode and Hansen 1992; U.S. Fish and Wildlife Service 1999b). These populations are isolated, without protected dispersal corridors to other adjacent populations. USFWS recognizes these 13 extant populations (58 FR 54053) as including Butte Basin, Colusa Basin, Sutter Basin, American Basin, Yolo Basin-Willow Slough, Yolo Basin-Liberty Farms, Sacramento Basin, Badger Creek-Willow Creek, Coldani Marsh, East Stockton Diverting Canal and Duck Creek, North and South Grassland, Mendota, and Burrel-Lanare. These populations extend from Fresno north to Chico and include portions of 11 counties: Butte, Colusa, Glenn, Fresno, Merced, Sacramento, San Joaquin, Solano, Stanislaus, Sutter, and Yolo (U.S. Fish and Wildlife Service 1999b:9, 11–12).

4.A.9.2 Life History and Habitat Requirements

Giant garter snake resides in marshes, ponds, sloughs, small lakes, low-gradient streams, and other waterways, and in agricultural wetlands, including irrigation and drainage canals, rice fields, and the adjacent uplands (58 FR 54053). It resides in small mammal burrows and soil crevices located above prevailing flood elevations throughout its winter dormancy period (U.S. Fish and Wildlife Service 2006b). Burrows are typically located in sunny exposures along south- and west-facing slopes. Data based on radiotelemetry studies show that home range varies by location, with median home range estimates varying between 23 acres (range [10.3 to 203 acres], n = 8) (9 hectares, range = 4.2 to 82 hectares) in a semi-native perennial marsh system and 131 acres (range [3.2 to 2,792 acres], n = 29) (53 hectares, range = 1.3 to 1130 hectares) in a managed refuge (U.S. Fish and Wildlife Service 1999b).

The species requires the following habitat elements.

- Adequate water during the snake's active season (early spring through mid-fall) to provide food and cover.

- Emergent, herbaceous wetland vegetation, such as cattails (*Typha* spp.) and bulrushes (*Schoenoplectus*, formerly *Scirpus*), accompanied by vegetated banks for escape cover and foraging habitat during the active season.

- Basking habitat of grassy banks and openings in waterside vegetation.

- High-elevation uplands for cover and refuge from floodwaters during the snake's dormant season in the winter (U.S. Fish and Wildlife Service 1999b:22).

Because of lack of habitat and emergent vegetation cover, giant garter snakes generally are not present in larger rivers and wetlands with sand, gravel, or rock substrates. In addition, the major rivers within the species' range have been highly channelized, removing oxbows and backwater areas that probably at one time provided suitable habitat. Riparian woodlands do not generally provide suitable habitat because most have excessive shade, lack of basking sites, and absence of prey populations. Giant garter snakes are also absent from most permanent waters that support established populations of predatory game fishes and from most sites that undergo routine dredging, mechanical or chemical weed control, or compaction of bank soils (Brode 1988; U.S. Fish and Wildlife Service 1999b, 2006b).

Changing agricultural regimes, development, and other shifts in land use create an ever-changing mosaic of available habitat. Giant garter snakes move around in response to these changes in order to find suitable sources of food, cover, and prey. Connectivity between regions is therefore extremely important for providing access to available habitat and for genetic interchange. In an agricultural setting, giant garter snakes rely largely on the network of canals and ditches that provide irrigation and drainage to provide this connectivity (Jones & Stokes 2005).

In the Central Valley, rice fields have become important habitat for giant garter snakes. Irrigation water typically enters the rice fields during April along canals and ditches. Giant garter snakes use these canals and their banks as permanent habitat for both spring and summer active behavior and winter hibernation. Where these canals are not regularly maintained, lush aquatic, emergent, and streamside vegetation develops prior to the spring emergence of giant garter snakes. This vegetation, in combination with cracks and holes in the soil, provides much-needed shelter and cover during spring emergence and throughout the remainder of the summer active period (Hansen 1998).

Rice is planted during spring, after the winter fallow fields have been cultivated and flooded with several inches of standing water. In some cases, giant garter snakes move from the canals and ditches into these rice fields soon after the rice plants emerge above the water's surface, and they continue to use the fields until the water is drained during late summer or fall (Hansen and Brode 1993). It appears that the majority of giant garter snakes move back into the canals and ditches as the rice fields are drained; a few may overwinter in the fallow fields, where they hibernate in burrows in the small berms separating the rice checks (low dikes) (Hansen 2008, 2011).

While in the rice fields, the snakes forage in the shallow, warm water for small fish and the tadpoles of bullfrogs and tree frogs. For shelter and basking sites, giant garter snakes use the rice plants, vegetated berms dividing the rice checks, and vegetated field margins. Gravid (pregnant) females may be observed in the rice fields during summer, and at least some giant garter snakes are born there (Hansen and Brode 1993; Hansen 2008).

Water is drained from the rice fields during late summer or fall by a network of drainage ditches. These ditches are sometimes routed alongside irrigation canals and are often separated from the irrigation canals by narrow vegetated berms that may provide additional shelter. Remnants of old sloughs also may remain within rice-growing regions, where they serve as drains or irrigation canals. Giant garter snakes may use vegetated portions along any of these waterways as permanent habitat. Studies indicate that despite the presence of ditches or drains, giant garter snakes will generally abandon aquatic habitat that is not accompanied by adjacent shallow-water wetlands (Wylie and Amarello 2008; Hansen 2007; Jones & Stokes 2008), underscoring the important role that rice plays in this species' life history.

4.A.9.3 Status of the Species in the Action Area/Environmental Baseline

The action area is in the Mid-Valley Recovery Unit identified in the draft recovery plan (U.S. Fish and Wildlife Service 1999b), and three of the 13 giant garter snake populations identified by USFWS are located in the action area along the periphery of the Delta, including the Yolo Basin-Willow Slough, Yolo Basin-Liberty Farms, and Coldani Marsh-White Slough populations (U.S. Fish and Wildlife Service 1999b). The rarity and isolation of giant garter snake from within the remainder the Delta suggest the lack of other extant populations in the area. Although giant garter snakes may have occupied this region at one time, longstanding reclamation of wetlands for intense agricultural applications has eliminated most suitable habitat (Hansen 1986) and prevented the re-establishment of viable giant garter snake breeding populations in the Delta, other than the three populations noted. Recent observations in the central Delta (e.g., Sherman Island) could be of snakes that occasionally move into the central Delta by 'washing-down' from known populations, such as Liberty Island or Coldani Marsh/White Slough, and that these occurrences do not represent local breeding populations (California Department of Fish and Wildlife 2013; Hansen 2011; Vinnedge Environmental 2013): USFWS and CSFW giant garter snake experts now believe recent sightings in the Central Delta may represent an extant population that lives in emergent vegetation along river edges; snakes have typically been found in this area along levess roads away from typically used habitat, so the status of snakes found in the Central Delta remains unknown.

4.A.9.4 Reasons for Decline

Habitat loss and fragmentation, flood control activities, changes in agricultural and land management practices, predation from introduced and native species, parasites, and water pollution are the main causes for the decline of giant garter snake. Conversion of Central Valley wetlands for agriculture and urban uses has resulted in the loss of as much as 95% of historical habitat for giant garter snake (Wylie et al. 1997). In areas where giant garter snake has adapted to agriculture, maintenance activities such as vegetation and rodent control, bankside grading or dredging, and discharge of contaminates, threaten their survival (Hansen and Brode 1980, 1993; U.S. Fish and Wildlife Service 1999b; Wylie et al. 2004). In developed areas, threats of vehicular mortality also are increased. Paved roads likely have a higher rate of mortalities than dirt or gravel roads due to increased traffic and traveling speeds. The loss of wetland habitat is compounded by elimination or compaction of adjacent upland and associated bankside vegetation cover, as well as water fouling; these conditions are often associated with cattle grazing (Thelander 1994). Although irrigated pastures may provide the summer water that giant garter snakes require, high stocking rates may degrade habitat by removing protective plant cover and underground and aquatic retreats such as rodent and crayfish burrows (Hansen 1986; U.S. Fish and Wildlife Service 1999b; Szaro et al. 1985). However, cattle grazing may provide an important function in controlling invasive vegetation that can compromise the overall value of wetland habitat.

Giant garter snakes are also threatened by the introduction of exotic species such as bullfrogs (Dickert 2003; U.S. Geological Survey 2004). Large vertebrates, including raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), red foxes (*Vulpes vulpes*), gray foxes (*Urocyon cinereoargenteus*), river otters (*Lutra canadensis*), opossums (*Didelphis virginiana*), northern harriers (*Circus cyaneus*), hawks (*Buteo* spp.), herons (*Ardea herodias, Nycticorax nycticorax*), egrets (*Ardea alba*, *Egretta thula*), and American bitterns (*Botaurus lentiginosus*) also prey on giant garter snakes (U.S. Fish and Wildlife Service 1999b). In areas near urban development, giant garter snakes may also fall prey to domestic or feral house cats. In permanent waterways, introduced predatory game fishes, such as bass (*Micropterus* spp.), sunfish (*Lepomis* spp.), and channel catfish (*Ictalurus* spp.), prey on giant garter snakes and compete with them for smaller prey (58 FR 54053; Hansen 2008).

Selenium contamination and impaired water quality have been identified as a threat to giant garter snakes, particularly in the southern portion of their range including Kesterson National Wildlife Area (U.S. Fish and Wildlife Service 1999b; Ohlendorf et al. 1988; Saiki and May 1988; Saiki et al. 1991).

4.A.9.5 Critical Habitat

Critical habitat has not been designated for giant garter snake.

4.A.9.6 Suitable Habitat Definition

Suitable habitat is described by USFWS in the 2015 Draft Recovery Plan (U.S. Fish and Wildlife Service 2015), including:

4.A.9.6.1 Aquatic Component

The giant garter snake has been recognized as requiring aquatic habitat since it was first described, and has been consistently observed and captured in association with aquatic habitats since accounts of the snake were first published. The aquatic component of the giant garter snake habitat has been regarded as a steadfast requirement for the survival of the snake, and researchers acknowledge the following qualitative requirements of ideal aquatic habitat for the giant garter snake (U.S. Fish and Wildlife Service 2015):

1. Water present from March through November.
2. Slow moving or static water flow with mud substrate.
3. Presence of emergent and bankside vegetation that provides cover from predators and may serve in thermoregulation.
4. The absence of a continuous canopy of riparian vegetation.
5. Available prey in the form of small amphibians and small fish.
6. Thermoregulation (basking) sites with supportive vegetation such as folded tule clumps immediately adjacent to escape cover.
7. The absence of large predatory fish.

8. Absence of recurrent flooding, or where flooding is probable the presence of upland refugia.

4.A.9.6.2 Upland Component

Although the giant garter snake is predominately an aquatic species, incidental observations and radio telemetry studies have shown that the snake can be found in upland areas near the aquatic habitat component during the active spring and summer seasons. Upland habitat (land that is not typically inundated during the active season and is adjacent to the aquatic habitat of the giant garter snake) is used for basking to regulate body temperature, for cover, and as a retreat into mammal burrows and crevices in the soil during ecdysis (shedding of skin) or to avoid predation. Giant garter snakes have been observed using burrows for refuge in the summer as much as 50 meters (164 feet) away from the marsh edge. Important qualities of upland habitat have been found by researchers (U.S. Fish and Wildlife Service 2015 to include:

1. Availability of bankside vegetative cover, typically tule (*Scirpus* sp.) or cattail (*Typha* sp.), for screening from predators.
2. Availability of more permanent shelter, such as bankside cracks or crevices, holes, or small mammal burrows.
3. Free of poor grazing management practices (such as overgrazed areas).

4.A.9.6.3 Upland Winter Refugia Component

During the colder winter months, giant garter snakes spend their time in a lethargic state. During this period, giant garter snakes over-winter in locations such as mammal burrows along canal banks and marsh locations, or riprap along a railroad grade near a marsh or roads. Giant garter snakes typically do not over-winter where flooding occurs in channels with rapidly moving water, such as the Sutter Bypass. Over-wintering snakes use burrows as far as 200 to 250 meters (656 to 820 feet) from the edge of summer aquatic habitat (U.S. Fish and Wildlife Service 2015), but are typically found within 200 feet of aquatic habitat, therefore USFWS typically considers uplands within 200 feet of aquatic habitat to be habitat for giant garter snake.

4.A.9.7 Species Habitat Suitability Model

During design and assessment of the proposed action, the habitat suitability model for the giant garter snake went through several cycles of review and revision. This led to a rather complex model that incorporates a wide variety of data sources, as detailed below in Section 4.A.9.7.3, 2011 and 2012 Updates to Giant Garter Snake Habitat Suitability Model, and Section 4.A.9.7.4, 2015 Updates to Giant Garter Snake Habitat Suitability Model. The latest changes to the model were made in the summer of 2015 in response to agency comments and included the addition of the verified wetland delineation (California Department of Water Resources 2015) to identify modeled aquatic habitat and the removal of occurrence data as a means by which upland habitat is qualified. For the portions of the action area not covered by the wetland delineation, the original aquatic habitat model remains. The model is further described below in Section 4.A.9.7.1, GIS Model Data Sources.

4.A.9.7.1 GIS Model Data Sources

The giant garter snake model uses vegetation types and associations from the following data sets: composite vegetation layer (Hickson and Keeler-Wolf 2007 [Delta]; TAIC 2008 [Yolo Basin]), California Department of Water Resources 2007 land use survey of the Delta area-version 3, land use survey of the Delta and Suisun Marsh area - version 3 (California Department of Water Resources 2007), and the USGS-National Hydrography Dataset, 1:24,000 (U.S. Geological Survey 1999). Using these data sets, the model maps the distribution of suitable giant garter snake habitat in the action area. Vegetation types and spatial buffers were assigned based on the species' requirements as described above and the assumptions described below.

4.A.9.7.2 Habitat Model Description

The model includes the following aquatic cover categories and associated types.

- Tidal aquatic habitat
    - Tidal freshwater perennial aquatic–all types
    - Tidal freshwater emergent wetland–all types
- Nontidal aquatic habitat
    - Nontidal freshwater emergent wetland–all types
    - Nontidal freshwater perennial aquatic–all types
    - Managed wetland (all except Suisun)
    - Bulrush–cattail freshwater marsh, NFD super alliance (all except Suisun)
- Agriculture
    - Rice
    - Wild rice

Modeled upland overwintering and movement habitat for giant garter snakes includes the following terrestrial land cover types immediately adjacent to and within 200 feet (61 meters) of the aquatic habitat types previously listed.

- Agriculture

  - Native vegetation[6]
  - Non-irrigated mixed pasture
  - Non-irrigated native pasture
- Alkali seasonal wetland complex
  - Alkali heath (*Frankenia salina*)
  - *Allenrolfea occidentalis* mapping unit
  - Alkaline vegetation mapping unit
  - Creeping wild ryegrass (*Leymus triticoides*)
  - *Distichlis spicata*–annual grasses
  - Distichlis spicata–Juncus balticus
  - *Distichlis spicata–Salicornia virginica* (formerly *Sarcocornia*)
  - *Frankenia salina–Distichlis spicata*
  - *Juncus balticus*-meadow vegetation
  - Pickleweed (*Salicornia virginica*)
  - *Salicornia virginica–Cotula coronopifolia*
  - *Salicornia virginica–Distichlis spicata*
  - Salt scalds and associated sparse vegetation
  - Saltgrass (*Distichlis spicata*)
  - *Suaeda moquinii*–(*Lasthenia californica*) mapping unit
- Developed
  - Levee rock riprap
  - Unclassified
- Grassland

[6] Native vegetation is a land use designation within the DWR crop type dataset (2007). For the purposes of incorporating native vegetation classes into the correct species models and, when applicable, assigning habitat foraging values, the management on these lands most resembles that of non-irrigated pasture or annual grassland.

- *Bromus diandrus–Bromus hordeaceus*
- California annual grasslands-herbaceous
- Degraded vernal pool complex–California annual grasslands–herbaceous
- Degraded vernal pool complex–Italian ryegrass (*Lolium multiflorum*)
- Italian ryegrass (*Lolium multiflorum*)
- *Lolium multiflorum–Convolvulus arvensis*
- Ruderal herbaceous grasses & forbs
- Upland annual grasslands & forbs formation
- Unclassified

- Inland dune scrub
    - *Lotus scoparius*-Antioch Dunes
    - *Lupinus albifrons*-Antioch Dunes
- Managed wetland
    - Barren gravel and sand bars
    - Bulrush–cattail fresh water marsh NFD super alliance
    - *Crypsis* spp.–wetland grasses–wetland forbs NFD super alliance
    - Intermittently flooded perennial forbs
    - Intermittently or temporarily flooded undifferentiated annual grasses and forbs–4
    - *Lepidium latifolium–Salicornia virginica–Distichlis spicata*
    - Managed alkali wetland (*Crypsis*)
    - Managed annual wetland vegetation (nonspecific grasses & forbs)
    - Perennial pepperweed (*Lepidium latifolium*)
    - Poison hemlock (*Conium maculatum*)
    - *Polygonum amphibium*
    - Rabbitsfoot grass (*Polypogon maritimus*)

- *Schoenoplectus* (formerly *Scirpus*) spp. in managed wetlands
- Seasonally flooded undifferentiated annual grasses and forbs
- Shallow flooding with minimal vegetation at time of photography
- Smartweed *Polygonum* spp. –mixed forbs
- Temporarily flooded grasslands

- Other natural seasonal wetland
    - Degraded vernal pool complex-vernal pools
    - *Juncus bufonius* (salt grasses)
    - Santa Barbara sedge (*Carex barbarae*)
    - Seasonally flooded grasslands
    - Temporarily flooded perennial forbs
    - Vernal pools
- Valley/foothill riparian
    - *Acacia–robinia*
    - *Acer negundo-Salix gooddingii*
    - *Alnus rhombifolia/Cornus sericea*
    - *Alnus rhombifolia/Salix exigua* (*Rosa californica*)
    - Arroyo willow (*Salix lasiolepis*)
    - *Baccharis pilularis*/annual grasses & herbs
    - Black willow (*Salix gooddingii*)
    - Black willow (*Salix gooddingii*)–valley oak (*Quercus lobata*) restoration
    - Blackberry (*Rubus discolor*)
    - Blackberry NFD super alliance
    - Box elder (*Acer negundo*)
    - Buttonbush (*Cephalanthus occidentalis*)

- California dogwood (*Cornus sericea*)
- California wild rose (*Rosa californica*)
- Coast live oak (*Quercus agrifolia*)
- *Cornus sericea–Salix exigua*
- *Cornus sericea–Salix lasiolepis*/(*Phragmites australis*)
- Coyote bush (*Baccharis pilularis*)
- Fremont cottonwood–valley oak–willow (ash–sycamore) riparian forest NFD association
- Fremont cottonwood (*Populus fremontii*)
- Giant cane (*Arundo donax*)
- Hinds' walnut (*Juglans hindsii*)
- Horsetail (*Equisetum* spp.)
- Intermittently flooded to saturated deciduous shrubland
- Intermittently or temporarily flooded deciduous shrublands
- Mexican elderberry (*Sambucus mexicana*)
- Mixed Fremont cottonwood–willow spp. NFD alliance
- Mixed willow super alliance
- Narrow-leaf willow (*Salix exigua*)
- Oregon ash (*Fraxinus latifolia*)
- Pampas grass (*Cortaderia selloana–C. jubata*)
- *Quercus lobata–Acer negundo*
- *Quercus lobata–Alnus rhombifolia* (*Salix lasiolepis–Populus fremontii–Quercus agrifolia*)
- *Quercus lobata–Fraxinus latifolia*
- *Quercus lobata/Rosa californica* (*Rubus discolor–Salix lasiolepis/Carex* spp.)
- Restoration sites

- *Salix exigua*–(*Salix lasiolepis*–*Rubus discolor*–*Rosa californica*)
- *Salix gooddingii*–*Populus fremontii*–(*Quercus lobata*–*Salix exigua*–*Rubus discolor*)
- *Salix gooddingii*–*Quercus lobata*/wetland herbs
- *Salix gooddingii/Rubus discolor*
- *Salix gooddingii*/wetland herbs
- *Salix lasiolepis*–(*Cornus sericea*)/*Schoenoplectus* spp.–(*Phragmites australis*–*Typha* spp.) complex unit
- *Salix lasiolepis*–mixed brambles (*Rosa californica*–*Vitis californica*–*Rubus discolor*)
- Shining willow (*Salix lucida*)
- Temporarily or seasonally flooded–deciduous forests
- Tobacco brush (*Nicotiana glauca*) mapping unit
- Valley oak (*Quercus lobata*)
- Valley oak (*Quercus lobata*) restoration
- Valley oak alliance–riparian
- White alder (*Alnus rhombifolia*)
- White alder (*Alnus rhombifolia*) –arroyo willow (*Salix lasiolepis*) restoration
- Unclassified

- Vernal Pool Complex
  - *Allenrolfea occidentalis* mapping unit
  - California annual grasslands–herbaceous
  - *Distichlis spicata*–annual grasses
  - Italian ryegrass (*Lolium multiflorum*)
  - Mixed *Schoenoplectus* mapping unit
  - Ruderal herbaceous grasses & forbs
  - Salt scalds and associated sparse vegetation
  - Saltgrass (*Distichlis spicata*)

- Seasonally flooded grasslands
- *Suaeda moquinii*–(*Lasthenia californica*) mapping unit
- Vernal pools

4.A.9.7.3 2011 and 2012 Updates to Giant Garter Snake Habitat Suitability Model

In 2011, and again in 2012, the species habitat models were updated to include previously unmapped portions of the action area. For most areas newly mapped, vegetation data were not available at the alliance level as in the rest of the action area and so most of the new analysis areas were mapped at the natural community level. Additional detail regarding crop types was available for cultivated lands and was incorporated into the mapping. For the giant garter snake, in the new analysis areas, the following natural communities are assumed to provide the listed habitat type (Department of Water Resources 2007).

- Agriculture
  - Rice (aquatic nontidal)
- Managed wetland (all except Suisun)
  - Bulrush–cattail freshwater marsh, NFD super alliance (all except Suisun) (aquatic nontidal)
- Nontidal freshwater perennial emergent wetland
  - Nontidal perennial aquatic–water (aquatic nontidal)

In the areas of additional analysis, the following tidal aquatic natural communities were assumed to provide giant garter snake aquatic habitat.

- Tidal freshwater emergent wetland (aquatic tidal)
- Tidal perennial aquatic
  - Tidal perennial aquatic–water (all except Suisun) (aquatic tidal)

In the areas of additional analysis, the following upland natural communities within 200 feet of aquatic habitat were assumed to provide giant garter snake upland habitat.

- Agriculture
  - Cultivated annual graminoid (upland)
  - Pasture (upland)
- Grasslands

  - Pasture (upland)
  - Upland annual grasslands & forbs formation (upland)
- Managed wetlands
  - *Crypsis* spp.-wetland grasses-wetland forbs NFD super alliance (upland)
  - Vernal pools (Upland)
- Other seasonal wetlands (upland)
- Vernal pool complex (upland)

4.A.9.7.4 2015 Updates to Giant Garter Snake Habitat Suitability Model

Since the last update in 2012, the model has gone through several additional changes which are described below.

- Rice patches were removed from Bouldin Island; rice is no longer grown in this region and this area is now categorized as “grain and hay” per the verified wetland delineation (California Department of Water Resources 2015) and a conversation between Mike Bradbury and the owner’s group (Bradbury, Mike, pers. comm., 2015).
- The November 2014 crop type layer replaced the June 2013 layer; the new layer provided more detail regarding the irrigation status of pasturelands (i.e., irrigated versus nonirrigated). This change had no effect on the giant garter snake impacts analysis, it was simply done so that all models are using the most up-to-date information.
- Where there was overlap with the former aquatic model, the verified wetland delineation (California Department of Water Resources 2015) data replaced the tidal and nontidal aquatic habitat model. The nontidal and tidal aquatic portions of the former model remain in areas outside of the wetland delineation area.
- The process of replacing the former aquatic portion of the model with the new wetland delineation data resulted in small “slivers” of land without coverage by the habitat model. This is because the wetland delineation data was more accurate than the previous tidal and nontidal model (i.e., the spatial extent of the wetland data was smaller and did not overlap 100% with the former model).
- These slivers described above were manually classified as either upland or determined to not be suitable habitat using aerial photography. Most of the slivers were classified as upland. A separate data layer of these slivers has been maintained to allow for review.
- The wetland delineation data included 13 types of wetland, 7 of which were considered giant garter snake habitat. Table 4.A.9-1 below presents which wetland types are considered habitat.

- The aquatic habitat was buffered by 1,000 feet and the agricultural ditches that were within the buffer were added to the model as aquatic habitat, replacing the “linear” portion of the model.

- The uplands portion of the model was not modified, however, uplands habitat was added to the model where there were slivers of land that were reclassified from aquatic to upland based on the new wetland delineation data. The upland habitat may have small changes as it is based on suitable land cover types within 200 feet of aquatic habitat and the above changes to the aquatic habitat could effect changes on the upland habitat.

Table 4.A.9-1. Wetland Types and Assumed Habitat Quality Values for the Revised Giant Garter Snake Aquatic Model.

Wetland or Water Group Type	Cowardin Class	Suitable Giant Garter Snake Habitat (Yes/No)	Rationale for Habitat Quality Value
Agricultural Ditch	R4	Yes	Some will be high value habitat, some will be moderate or low; but because only those agricultural ditches within a given distance of suitable/beneficial upland habitat are selected, a moderate value is reasonable.
Alkaline Wetland	PEM/PSS	No	Giant garter snakes are not known to occur in vernal pools or alkali seasonal wetlands.
Clifton Court Forebay	R1UB	No	Giant garter snakes are not known from Clifton Court Forebay.
Conveyance Channel	R1UB	No	Giant garter snakes are not known to occur within the conveyance channel on the western edge of Clifton Court Forebay.
Depression	PUB	Yes	Open water infested with predatory, non-native fish; small amount of emergent wetland.
Emergent Wetland	PEM	Yes	Emergent, herbaceous wetland vegetation, such as cattails and bulrushes provide foraging habitat during the active season
Forest	PFO	No	The small number of wetlands in this type are in the Coumnes-Mokelumne area and because they are surrounded by forest/riparian areas are not considered habitat.
Lake	L1UB	Yes	Open water infested with predatory, non-native fish; small amount of emergent wetland.
Natural Channel	R4	Yes	Does not have permanent water, forested up to the edge of the aquatic habitat.
Scrub-Shrub	PSS	No	Scrub shrub is an alkali seasonal wetland type and alkali wetland types are not known to support giant garter snake in the action area, west of Clifton Court Forebay.
Seasonal Wetland	PEM	No	Because of their seasonality and poor vegetation quality, seasonal wetlands are not considered habitat. Surrounding uplands and ag ditches would be the primary habitat in these regions.
Tidal Channel	R1UB/R1UB V	Yes	Open-water, high flows, high density of predatory, invasive fish; emergent wetland habitat is the high value habitat and tidal channels are just providing movement habitat.
Vernal Pool	PEM2	No	Giant garter snakes are not known to occur in vernal pools or alkali seasonal wetlands in the action area.

4.A.9.7.5 Assumptions

Giant garter snakes inhabit marshes, ponds, sloughs, small lakes, low-gradient streams and other waterways, and agricultural wetlands, including irrigation and drainage canals, rice fields, and the adjacent uplands (U.S. Fish and Wildlife Service 2006b). In the Sacramento Valley, their habitat requirements include adequate water during the snake's active season (early spring through mid-fall) to provide food and cover, and emergent herbaceous wetland vegetation for escape cover and foraging habitat during the active season.

- **Assumption:** Suisun Marsh does not support potentially occupied giant garter snake habitat.

  Rationale: Suisun Marsh lies outside of the acknowledged range of the species (U.S. Fish and Wildlife Service 1999).

- **Assumption:** Giant garter snakes could potentially use any watercourse within 1,000 feet of aquatic habitat, perennial marsh, or flooded rice field in the action area, except in Suisun Marsh.

  Rationale: Watercourses, perennial marsh, and flooded rice fields are most likely consistently inundated during most of the snake's active season and are therefore available for breeding, foraging, or movement.

- **Assumption:** Tidal perennial aquatic habitat suitable for giant garter snake consists of those areas within 20 feet (6 meters) of bank margins.

  Rationale: In tidal perennial aquatic features (e.g., the Sacramento and San Joaquin Rivers and tidal zones in the central Delta), giant garter snakes are limited to shallow, near-shore habitats providing vegetative cover, foraging, thermoregulating opportunities, and refuge from predatory fishes. Accordingly, tidal perennial aquatic features are buffered internally by 20 feet (6 meters) to capture the near-shore habitat and exclude the relatively deep water areas that are considered unsuitable.

- **Assumption:** Potentially occupied giant garter snake upland habitat consists of the vegetation types listed in Section 4.A.9.7.2, *Habitat Model Description,* and upland habitat values are consistent with the designated value rankings for each vegetation type listed.

  Rationale: Giant garter snakes require basking habitat of grassy banks and openings in waterside vegetation. They also require uplands for cover and refuge from floodwaters during the snake's dormant season in the winter (U.S. Fish and Wildlife Service 2006b). Riparian woodlands are unlikely to provide suitable habitat as a result of excessive shade, lack of basking sites, and absence of prey populations (U.S. Fish and Wildlife Service 2006b). However, giant garter snakes can potentially occur along watercourses with willow-dominated riparian or riparian scrub habitats, particularly where emergent herbaceous wetland vegetation is present, because of the relatively low overstory structure and intermittent occurrence of the riparian vegetation. Vegetation types that are relatively open are most likely to provide basking sites and burrows, and are likely to have the highest habitat value for giant garter snakes.

- **Assumption:** Potentially occupied giant garter snake upland habitat consists of appropriate land cover types within 200 feet (61 meters) of modeled aquatic habitat

  Rationale: Giant garter snakes use grassy stream banks and upland habitats adjacent to perennial watercourses or wetlands as overwintering and movement habitat.

4.A.9.7.6 Model Limitations

Suitable upland overwintering habitat is overestimated in areas subject to prolonged inundation by flood events such as that which occurs in the Yolo Bypass. Periodic inundation influences suitability for use as overwintering habitat and, depending on the frequency of inundation, could create a biological sink as snakes reestablish overwintering patterns in the inundation zone during nonflood years and then are displaced from or killed at overwintering sites during an inundation event. Because there is little research on this topic, the Yolo Bypass is included as potential overwintering habitat for giant garter snake; however, it is likely that either the bypass is not used for this purpose because of the current frequency and extent of flooding or that it represents a site where snakes are periodically displaced during the inactive season when inundation occurs.

Most historical and recent occurrences of the giant garter snake in the action area have been reported from areas outside of the central Delta, including portions of the Yolo Basin and at Coldani Marsh/White Slough along the eastern edge of the action area (California Department of Fish and Wildlife 2013; Hansen 2007, 2009, 2011; Wylie and Amarello 2008). These areas are also consistent with the USFWS' description of extant populations within the action area and Yolo Basin (U.S. Fish and Wildlife Service 1999). Additional relatively recent occurrences extend north of Coldani Marsh/White Slough to Stone Lakes and east of the Mokelumne and Sacramento Rivers. The northern and eastern portions of the action area are known to support extant populations and are where recent and historical records suggest a greater likelihood of undiscovered extant populations to occur as described above.

Scattered records from the central Delta suggest that giant garter snakes may have occupied this region at one time, but longstanding reclamation of wetlands for intense agricultural applications has eliminated most suitable habitat (Hansen 1986). Historical and recent surveys conducted in the Delta have failed to identify any extant population clusters in the region (Hansen 1986; Patterson 2005; California Department of Water Resources 2006), including 2009 surveys conducted by DWR (Hansen 2011). The action area is within the Delta Basin Recovery Unit for giant garter snake (USFWS 2015). Occurrences for giant garter snake in the action area are shown on Figure 6.6-1.

4.A.10 California Red-legged Frog (*Rana draytonii*)

4.A.10.1 Legal Status and Distribution

California red-legged frog was Federally listed as threatened pursuant to the Federal Endangered Species Act (ESA) in 1996 (61 *Federal Register* [FR] 25813). A recovery plan was prepared for this species by the U.S. Fish and Wildlife Service (USFWS) in 2002 (U.S. Fish and Wildlife Service 2002a), and a 5-year review was initiated in 2011 (76 FR 30377). California red-legged frog is also considered a species of special concern by the California Department of Fish and Wildlife.

The historical range of the California red-legged frog generally extends south along the coast from the vicinity of Point Reyes National Seashore, Marin County, California, and inland from the vicinity of Redding, Shasta County, California, southward along the interior Coast Ranges and Sierra Nevada foothills to northwestern Baja California, Mexico (U.S. Fish and Wildlife Service 2007b). Although there are a few historical records from several Central Valley locales (Jennings and Hayes 1994), Fellers (2005) considers persistent occupancy in the lowlands of the Central Valley unlikely due to extensive annual flooding.

The current range is generally characterized based on the current known distribution. USFWS (2007b) notes that while the California red-legged frog is still locally abundant in portions of the San Francisco Bay area and the central coast, only isolated populations have been documented elsewhere within the species' historical range, including the Sierra Nevada, northern Coast Ranges, and northern Transverse Ranges.

4.A.10.2 Life History and Habitat Requirements

Storer (1925) and Hayes and Jennings (1988) describe aquatic breeding habitat requirements for California red-legged frog as cold water pond habitats (including stream pools) with emergent and submergent vegetation, providing suitable cover for young and adults and ensuring successful reproduction. Optimal habitats are described as deep-water ponds or pools at least 2.3 feet deep along low-gradient streams with dense stands of overhanging willows and a fringe of cattails between the willow roots and overhanging willow limbs. Hayes and Jennings (1988) also note that California red-legged frogs may prefer pools along intermittent streams rather than backwater pools along perennial streams, possibly for predator avoidance, particularly bullfrogs (*Lithobates catesbeianus*). California red-legged frog uses a variety of aquatic habitats that meet these requirements including permanent and ephemeral ponds, perennial and intermittent streams, seasonal wetlands, springs, seeps, marshes, dune ponds, lagoons, and human-made aquatic features (U.S. Fish and Wildlife Service 2007b).

In addition to aquatic breeding habitat, California red-legged frog also requires upland nonbreeding habitat for cover, aestivation, and migration and other movements. Nonbreeding cover habitat may include nearly any area within 1 to 2 miles of a breeding site that stays moist and cool through the summer, and can include vegetated areas with coyote bush (*Baccharis pilularis*), California blackberry thickets (*Rubus ursinus*), and root masses associated with willows (*Salix* spp.) and California bay trees (*Umbellularia californica*) (Fellers and Kleeman 2007). Potential cover habitat includes all aquatic, riparian, and upland areas that provide cover, such as animal burrows, boulders or rocks, organic debris such as downed trees or logs, and industrial debris; agricultural features such as drains, watering troughs, spring boxes, abandoned sheds, or hay stacks may also be used (61 FR 25813).

Juvenile frogs are active diurnally and nocturnally, while adult frogs are primarily nocturnal (Hayes and Tenant 1985). California red-legged frogs are most likely to make overland movements through upland habitats at night during wet weather (U.S. Fish and Wildlife Service 2002a; Bulger et al. 2003; Fellers and Kleeman 2007). During the course of a wet season, movements up to 1 mile are possible (U.S. Fish and Wildlife Service 2002a). During dry weather, the subspecies tends to remain very close to a water source and are typically within about 200 feet of water (U.S. Fish and Wildlife Service 2002a; Bulger et al. 2003; Fellers and Kleeman 2007). California red-legged frogs have been known to disperse distances up to 1.8 miles from the breeding site to sites within the stream system (U.S. Fish and Wildlife Service 2002a; Fellers and Kleeman 2007).

Breeding occurs between late November and late April (Jennings and Hayes 1994) and most frogs lay their eggs in March (U.S. Fish and Wildlife Service 2002a). Males move to breeding sites 2 to 4 weeks before females arrive (Storer 1925). Eggs hatch in 20 to 22 days, depending on water temperature (U.S. Fish and Wildlife Service 2002a). Thereafter, tadpoles require 11 to 20 weeks to complete metamorphosis (Storer 1925).

4.A.10.3 Reasons for Decline

USFWS (2002a) estimates that the species has lost approximately 70% of its former range, with severe declines occurring primarily in the Central Valley and southern California (Jennings and Hayes 1994). Sizable populations continue to exist only in coastal drainages and associated pond habitats between Point Reyes and Santa Barbara (Jennings and Hayes 1994).

The principal factors contributing to the decline of the California red-legged frog are loss of habitat due to urban development, conversion of native habitats to agricultural lands, introduction of nonnative predators, and pesticide use (Fisher and Shaffer 1996; Hobbs and Mooney 1998; Davidson et al. 2002).

Habitat loss, degradation, and fragmentation are significant factors in declining populations of California red-legged frogs. Conversion of lands to agricultural and urban uses, overgrazing, mining, recreation, and timber harvesting have all contributed to habitat losses and disturbances. Urbanization often fragments habitat and creates barriers to dispersal (U.S. Fish and Wildlife Service 2002a). Road densities generally increase as a consequence of urbanization. Roads can create significant barriers to frog dispersal (Reh and Seitz 1990) and reduce population densities due to mortality caused by automobile strikes (Fahrig et al. 1995; Yolo County Habitat Conservation Plan/Natural Community Conservation Plan Joint Powers Agency 2009).

The conversion of natural lands to agricultural uses, such as stands of monotypic row crops, can alter habitats to the extent that they become uninhabitable for California red-legged frogs (U.S. Fish and Wildlife Service 2002a). Fisher and Shaffer (1996) suggest that intense farming in the San Joaquin Valley has resulted in drastic declines in California red-legged frog populations, due to a lack of suitable habitat. Pesticides, herbicides, and other agrochemicals are known to be toxic to various life stages of ranid frogs (Hayes and Jennings 1986). Pesticide drift has also been suggested as a potential cause of declining populations of four species of ranids in California, including California red-legged frogs (Davidson et al. 2002).

Exotic predatory fish and bullfrogs also pose significant threats to California red-legged frogs. Hayes and Jennings (1986) noted that locations in which exotic fish were present contained few California red-legged frogs. Bullfrogs have been implicated in the decline of the subspecies in several studies (Fisher and Shaffer 1996; Kiesecker and Blaustein 1998; Lawler et al. 1999), and Moyle (1973) indicated that bullfrogs might have been the most important factor in the extirpation of California red-legged frogs from the Central Valley floor. Bullfrogs depredate and out-compete California red-legged frogs due to their larger size, more varied diet, and longer breeding season (Hayes and Jennings 1986; Yolo County Habitat Conservation Plan/Natural Community Conservation Plan Joint Powers Agency 2009).

4.A.10.4 Status of the Species in the Action Area/Environmental Baseline

In the action area, California red-legged frog has been detected only in aquatic habitats within the grassland landscape west and southwest of Clifton Court Forebay and in the vicinity of Brentwood and Marsh Creek along the west-central edge of the action area, and in some upland sites in the vicinity of Suisun Marsh. These areas are within the easternmost edge of the current range of California red-legged frog within the Coast Ranges. While there are several recent detections of the species in the Sierra Nevada foothills, California red-legged frog is not known to occur in the agricultural habitats of the Central Valley. The California Natural Diversity Database contains records for several extant occurrences along Marsh Creek and Clifton Court Forebay and the western edge of the Suisun Marsh (California Department of Fish and Wildlife 2013). Occupied habitats are characterized by grassland foothills with stock ponds and slow-moving perennial drainages. The species is not known to occur, nor is it expected to occur, elsewhere in the action area.

4.A.10.5 Critical Habitat

Final designation of critical habitat for California red-legged frog was published in the *Federal Register* on March 17, 2010 (75 FR 12816–12959). There is no designated critical habitat for California red-legged frog in the action area. Critical habitat unit ALA-2 is located west of Clifton Court Forebay in the vicinity of the action area.

4.A.10.6 Suitable Habitat Definition

As described above in Section 4.A.10.2, *Life History and Habitat Requirements*, and below in Section 4.A.10.7, *Species Habitat Suitability Model*, suitable aquatic breeding habitat for California red-legged frog in the action area consists of perennial and intermittent streams, managed wetland, freshwater emergent wetland, and perennial aquatic natural communities (e.g., ponds). Other aquatic habitats that are suitable, though may not be present in the action area, include seasonal wetlands, springs, seeps, marshes, dune ponds, lagoons, and human-made aquatic features (U.S. Fish and Wildlife Service 2007b). Upland cover and dispersal habitat include almost any areas within 1 to 2 miles of breeding habitat but within the action area would be limited to annual grasslands, alkali seasonal wetland complex, vernal pool complex, and valley/foothill riparian.

4.A.10.7 Species Habitat Suitability Model

4.A.10.7.1 GIS Model Data Sources

The California red-legged frog model uses vegetation types and associations from the following data sets: composite vegetation layer (Hickson and Keeler-Wolf 2007 [Delta], Boul and Keeler-Wolf 2008 [Suisun Marsh], TAIC 2008 [Yolo Basin]), aerial photography (U.S. Department of Agriculture 2005), and land use survey of the Sacramento–San Joaquin River Delta (Delta), Suisun Marsh area-version 3 (California Department of Water Resources 2007) and the National Hydrography Dataset (U.S. Geological Survey 1999). Using these data sets, the model maps the distribution of suitable California red-legged frog habitat in the action area according to the species' two primary life requisites: aquatic breeding habitat and upland cover and dispersal habitat. Vegetation types were assigned to a suitability category based on the species requirements as described above and the assumptions described below.

4.A.10.7.2 Aquatic Habitat Model Description

Aquatic habitat for the California red-legged frog includes the following land cover types and conditions in the area south and west of SR 4 from Antioch (Bypass Road to Balfour Road to Brentwood Boulevard) to Byron Highway; then south and west along the county line to Byron Highway; then west of Byron Highway to I-205, north of I-205 to I-580, and west of I-580. Habitat also occurs along the western edge of Suisun Marsh, west of I-680. Habitat in the California Aqueduct and the Delta Mendota Canal is not included the model.

- Perennial and intermittent streams
- Aquatic habitat types from the composite vegetation layer
    - Managed wetland
        - *Schoenoplectus* (formerly known as *Scirpus*) spp. in managed wetlands
        - Polygonum amphibium
    - Nontidal freshwater perennial emergent
        - Broad-leaf cattail (*Typha latifolia*)
        - American bulrush (*Schoenoplectus americanus*)
        - Mixed *Schoenoplectus* mapping unit
        - *Schoenoplectus acutus* pure
        - *Schoenoplectus acutus* (*Typha latifolia*)–*Phragmites australis*

- Tidal freshwater emergent wetland
    - Mixed *Schoenoplectus* mapping unit
    - Mixed *Schoenoplectus*/floating aquatics (*Hydrocotyle–Eichhornia*) complex
    - Mixed *Schoenoplectus*/submerged aquatics (*Egeria–Cabomba–Myriophyllum* spp.) complex
    - Hardstem bulrush (*Schoenoplectus acutus*)
    - *Schoenoplectus acutus* pure
    - *Schoenoplectus acutus–Typha angustifolia*
    - *Schoenoplectus acutus–Typha latifolia*
    - *Schoenoplectus acutus*–(*Typha latifolia*)–*Phragmites australis*
    - California bulrush (*Schoenoplectus californicus*)
    - *Schoenoplectus californicus–Eichhornia crassipes*
    - *Schoenoplectus californicus–Schoenoplectus acutus*
    - American bulrush (*Schoenoplectus americanus*)
    - Narrow-leaf cattail (*Typha angustifolia*)
    - *Typha angustifolia–Distichlis spicata*
- Perennial aquatic
    - Floating primrose (*Ludwigia peploides*)
    - *Ludwigia peploides*
    - Generic floating aquatics
    - Water hyacinth (*Eichhornia crassipes*)
    - Pondweed (*Potamogeton* spp.)
    - Milfoil–waterweed (generic submerged aquatics)
    - Brazilian waterweed (*Egeria–Myriophyllum*) submerged
    - *Hydrocotyle ranunculoides*
    - Algae

- Water

4.A.10.7.3 Assumptions

- **Assumption:** California red-legged frog habitat in the action area is geographically constrained to areas described in Section 4.A.10.7.2, *Aquatic Habitat Model Description*.

  Rationale: In the action area, the California red-legged frog has been detected only in aquatic habitats in the grassland landscape west of Clifton Court Forebay, near Brentwood and Marsh Creek along the west-central edge of the action area, and along the western edge of Suisun Marsh, west of I-680. These areas represent the easternmost edge of the current range of California red-legged frog in the Coast Ranges. The species is not known to occur, nor is it expected to occur, elsewhere in the action area. Optimal habitats are described as deep-water ponds or pools along low-gradient streams with dense stands of overhanging willows and a fringe of cattails between the willow roots and overhanging willow limbs. The California red-legged frog uses a variety of aquatic habitats that meet these requirements, including permanent and ephemeral ponds including stock ponds, perennial and intermittent streams, seasonal wetlands, springs, seeps, marshes, dune ponds, lagoons, and human-made aquatic features (U.S. Fish and Wildlife Service 2007b).

4.A.10.7.4 Upland Cover and Dispersal Habitat Model Descriptions

Upland cover and dispersal habitat for the California red-legged frog is confined to the area south and west of SR 4 from Antioch (Bypass Road to Balfour Road to Brentwood Boulevard) to Byron Highway; then south and west along the county line to Byron Highway; then west of Byron Highway to I-205, north of I-205 to I-580, and west of I-580. Habitat also occurs along the western edge of Suisun Marsh, west of I-680. Modeled upland cover and dispersal habitat is limited to lands within 1 mile of aquatic habitat.

Upland cover and dispersal habitat from the composite vegetation layer includes the following components.

- Grassland–all types
- Valley/foothill riparian–all types
- Vernal pool complex
  - California annual grasslands
  - Ruderal herbaceous grasses and forbs
  - Italian ryegrass (*Lolium multiflorum*)

In 2011, and again in 2012, the species habitat models were updated to include previously unmapped portions of the action area. For most newly mapped areas, vegetation data were not available at the alliance level as in the rest of the action area and so most of the new analysis areas were mapped at the natural community level. In the new analysis areas, the following natural communities were assumed to provide upland cover and dispersal habitat for California red-legged frog.

- Alkali seasonal wetland
- Grassland

4.A.10.7.5 Dispersal Habitat

Modeled upland dispersal habitat also includes agricultural lands within the area described above and within 1 mile of the aquatic habitat, except for agricultural lands where dispersal is bounded on the west by Byron Highway. There is no known, high-value breeding habitat east of that significant boundary.

Upland dispersal habitat from the composite vegetation layer includes the following component.

- Agricultural land–all types

4.A.10.7.6 Assumptions

- **Assumption:** California red-legged frog requires upland nonbreeding habitat within 2 miles of breeding habitat used for cover, aestivation, and migration and other movements.

  Rationale: The California red-legged frog also requires upland nonbreeding habitat used for cover, aestivation, and migration and other movements. Nonbreeding cover habitat may include nearly any areas within 1 to 2 miles (1.6 to 3.2 kilometers) of a breeding site that stays moist and cool through the summer (Fellers and Kleeman 2007). Potential cover habitat includes all aquatic, riparian, and upland areas that provide cover, such as animal burrows, boulders or rocks, organic debris such as downed trees or logs, and industrial debris; agricultural features such as drains, watering troughs, spring boxes, abandoned sheds, or hay stacks may also be used (61 FR 25813). Movement corridors may include annual grasslands, riparian corridors, woodlands, and sometimes active agricultural lands (Fellers and Kleeman 2007).

4.A.11 California Tiger Salamander (*Ambystoma californiense*)

4.A.11.1 Legal Status and Distribution

The Central California distinct population segment of California tiger salamander (which overlaps with the action area) is Federally listed as threatened (50 *Federal Register* [FR] 47212–47248, August 4, 2004). California tiger salamander is also listed as threatened under the California Endangered Species Act (ESA).

Historically, California tiger salamander occurred throughout the grassland and woodland areas of the Sacramento and San Joaquin River Valleys and surrounding foothills, and in the lower elevations of the central Coast Ranges (Barry and Shaffer 1994). The species is found in a relatively dry landscapes where its range is limited by its aestivation and winter breeding habitat requirements, which are generally defined as open grassland landscapes with ephemeral pools and with ground squirrel and pocket gopher burrows (Barry and Shaffer 1994).

Within the coastal range, the species currently occurs from southern San Mateo County south to San Luis Obispo County, with isolated populations in Sonoma and northwestern Santa Barbara Counties (California Department of Fish and Wildlife 2013). In the Central Valley and surrounding Sierra Nevada foothills, the species occurs from northern Yolo County southward to northwestern Kern County and northern Tulare and Kings Counties (California Department of Fish and Wildlife 2013).

4.A.11.2 Life History and Habitat Requirements

California tiger salamander is found in annual grasslands and open woodland communities in lowland and foothill regions of central California where aquatic sites are available for breeding (U.S. Fish and Wildlife Service 2003). The species is typically found at elevations below 1,509 feet (68 FR 13498), although the known elevational range extends up to 3,455 feet (Jennings and Hayes 1994). Ecological characteristics of this area include dry soils, needlegrass grasslands, valley oaks, coast live oaks, and ephemerally flooded claypan vernal pools (U.S. Fish and Wildlife Service 2003).

Adult California tiger salamanders are terrestrial and spend much of the year (6 to 9 months) in the underground burrows of small mammals, such as California ground squirrels (*Spermophilus beecheyi*) and Botta's pocket gopher (*Thomomys bottae*), in grassland and open woodland habitats (Storer 1925; Loredo and van Vuren 1996; Petranka 1998). Active rodent burrow systems are considered an important component of California tiger salamander upland habitat (Loredo et al. 1996; U.S. Fish and Wildlife Service 2013b). Active ground-burrowing rodent populations are probably necessary to sustain California tiger salamander populations because inactive burrow systems begin to deteriorate and collapse over time (Loredo et al. 1996). In a 2-year radiotelemetry project in Monterey County, Trenham (2001) found that salamanders preferentially used open grassland and isolated oaks; salamanders present in continuous woody vegetation were never more than 10 feet from open grassland, potentially because ground squirrels prefer to construct burrows in open habitats (Jameson and Peeters 1988 in Trenham 2001).

Vernal pools and other seasonal rain pools are the primary breeding habitat of California tiger salamanders (Barry and Shaffer 1994; 68 FR 13498). Because the species requires at least 10 weeks of pool inundation in order to complete metamorphosis of larvae (Anderson 1968; East Contra Costa County Habitat Conservancy 2006), California tiger salamanders are usually only found in the largest vernal pools (Laabs et al. 2001). The species is also known to successfully reproduce in ponds (Barry and Shaffer 1994; 69 FR 47212). In the East Bay Regional Park District in Contra Costa and Alameda Counties, California tiger salamanders breed almost exclusively in seasonal and perennial stock ponds (Bobzien and DiDonato 2007). However, the presence of predatory fish and bullfrogs (*Rana catesbeiana*) can affect the habitat suitability of perennial ponds (Holomuzki 1986; Fitzpatrick and Shaffer 2004). Barry and Shaffer (1994) note that perennial stock ponds can be productive breeding sites as long as they are drained annually, which can prevent predatory species from establishing.

Adult California tiger salamanders move from subterranean refuge sites to breeding pools during relatively warm late winter and spring rains (Jennings and Hayes 1994:12). Breeding generally occurs from December through March (Stebbins 2003:154). Development through metamorphosis requires 3–6 months (69 FR 47215). Metamorphosed juveniles leave their ponds in the late spring or early summer and move to terrestrial refuge sites before seasonal ponds dry (Loredo et al. 1996:282).

The distance between occupied upland habitat and breeding sites depends on local topography and vegetation, and the distribution of California ground squirrel or other rodent burrows (WRA Environmental 2005; Cook et al. 2006). While juvenile California tiger salamanders have been observed to disperse up to 1.6 miles from breeding pools to upland areas (Austin and Shaffer 1992) and adults have been observed up to 1.2 miles from breeding ponds, most movements are closer to the breeding pond. Trenham et al. (2001) observed California tiger salamanders moving up to 0.42 mile between breeding ponds in Monterey County. Similarly, Shaffer and Trenham (2005) found that 95% of California tiger salamanders resided within 0.4 mile of their breeding pond at Jepson Prairie in Solano County.

Interconnectivity of breeding sites may be an important factor in long-term conservation of this species in order to sustain the species' metapopulation structure, where local extinction and recolonization by migrants of other subpopulations are probably common (69 FR 47212). Thus, providing movement corridors between potential breeding sites and avoiding isolation of these sites may counterbalance the effects of normal ecological processes (e.g., drought) that may result in local extinctions by allowing for movements to new sites and facilitating recolonization (Semlitsch et al. 1996).

4.A.11.3 Reasons for Decline

Conversion of land to residential, commercial, and agricultural activities is considered the most significant threat to California tiger salamanders, resulting in destruction and fragmentation of upland and/or aquatic breeding habitat and killing of individual California tiger salamanders (Twitty 1941; Shaffer et al. 1993; Jennings and Hayes 1994; Fisher and Shaffer 1996; Loredo et al. 1996; Davidson et al. 2002; California Department of Fish and Game 2010). Roads can fragment breeding habitats and dispersal routes in areas where they traverse occupied habitat.

Features of road construction, such as solid road dividers, can further impede migration, as can other potential barriers such as berms, pipelines, and fences.

Exotic species, such as bullfrog, mosquitofish (*Gambusia affinis*), sunfish species (e.g., largemouth bass [*Micropterus salmoides*] and bluegill [*Lepomis macrochirus*]), catfish (*Ictalurus* spp.), and fathead minnows (*Pimephales promelas*), that live in perennial ponds—such as stock ponds—are considered to have negatively affected California tiger salamander populations by preying on larval salamanders (Anderson 1968; Shaffer et al. 1993; Fisher and Shaffer 1996; Lawler et al. 1999; Laabs et al. 2001; Leyse 2005; U.S. Fish and Wildlife Service 2013b). Hybridization with the barred tiger salamander (*Ambystoma tigrinum mavortium*) is also a threat to this species, although it is unlikely that hybridization or nonnative alleles occur in California tiger salamander populations found in the action area, and hybridization does not appear to be a serious threat in this area (California Department of Water Resources 2013; Riley et al. 2003; Fitzpatrick et al. 2009).

Pesticides, hydrocarbons, and other pollutants are all thought to negatively affect breeding habitat, while rodenticides used in burrowing mammal control (e.g., chlorophacinone, diphacinone, strychnine, aluminum phosphide, carbon monoxide, and methyl bromide) are considered toxic to adult salamanders (Salmon and Schmidt 1984). California ground squirrel and pocket gopher control operations may have the indirect effect of reducing the availability of upland burrows for use by California tiger salamanders (Loredo-Prendeville et al. 1994).

4.A.11.4 Status of the Species in the Action Area/Environmental Baseline

Several occurrences of California tiger salamander are located immediately west of Clifton Court Forebay, in the vicinity of the action area (California Department of Fish and Wildlife 2013). Current occupancy of some of these sites was confirmed by larval surveys conducted between 2009 and 2011 by the California Department of Water Resources. There are numerous additional occurrences of California tiger salamander in vernal pool and pond habitats in the grassland foothills west of the action area and south of Antioch. Vernal pool habitats in Yolo and Solano Counties west of Liberty Island and in the vicinity of Stone Lakes in Sacramento County also provide suitable habitat for the species.

4.A.11.5 Critical Habitat

Final designation of critical habitat for the Central California Population of California tiger salamander was published in the *Federal Register* on August 23, 2005 (70 FR 49380-49458). There is no designated critical habitat for California tiger salamander in the action area. Critical habitat Unit 2, the Jepson Prairie Unit, is located west of the action area.

4.A.11.6 Suitable Habitat Definition

As described above in Section 4.A.11.2, *Life History and Habitat Requirements*, and below in Section 4.A.11.7, *Species Habitat Suitability Model*, suitable habitat for California tiger salamander includes aquatic habitat consisting of vernal pools, other seasonal pools, and ponds that inundate for at least 10 weeks and upland habitat consisting of adjacent annual grassland, including alkali grasslands, with small mammal burrows for refugia. The areas of suitable habitat in the action area are limited to those areas described below. Though the model for upland habitat below is limited to 100-acre patch sizes, actual occupied habitat could be in patches smaller than this and thus suitable upland habitat will be determined on the ground during planning level surveys. The extent of suitable upland habitat around suitable aquatic habitat will be determined based on evaluation of site conditions, which will include connectivity of upland habitat and presence of subterranean refugia, and will extend up to 1.24 miles from aquatic habitat based on the USFWS's *Interim Guidance on Site Assessment and Field Surveys for Determining Presence or a Negative Finding of the California Tiger Salamander* (U.S. Fish and Wildlife Service 2003).

4.A.11.7 Species Habitat Suitability Model

4.A.11.7.1 Terrestrial Cover and Aestivation Habitat Model Description

Modeled terrestrial cover and aestivation habitat is defined as all grassland types with a minimum patch size of 100 acres (40.5 hectares) located west of the Yolo Basin but including the Tule Ranch Unit of the CDFW Yolo Basin Wildlife Area; east of the Sacramento River between Freeport and Hood-Franklin Road; east of I-5 between Twin Cities Road and the Mokelumne River; and in the area south and west of SR 4 from Antioch (Bypass Road to Balfour Road to Brentwood Boulevard) to Byron Highway; then south and west along the county line to Byron Highway; then west of Byron Highway to I-205, north of I-205 to I-580, and west of I-580. These geographically described areas were developed into a habitat constraint GIS layer to limit the qualifying terrestrial habitat extents. Grasslands associated with south Montezuma Hills and Potrero Hills were also included. Grassland strips solely occurring atop levees and not adjacent to grassland areas were excluded. The excluded grassland strips were manually selected and developed into a GIS layer by visually reviewing grassland strips that occurred atop the levees, and comparing them to 2005 aerial photographs (U.S. Department of Agriculture 2005). These identified locations were removed from the habitat model. Patches of grassland that were below the 100-acre minimum patch size but were contiguous with grasslands outside of the action area boundary were included.

Terrestrial covered and aestivation habitat includes the following types from the composite vegetation layer.

- Grassland
  - Ruderal herbaceous grasses and forbs
  - California annual grasslands–herbaceous
  - *Bromus diandrus–Bromus hordeaceus*
  - Italian ryegrass (*Lolium multiflorum*)
  - *Lolium mulitflorum–Convolvulus arvensis*
  - Degraded vernal pool complex–California annual grasslands
  - Degraded vernal pool complex–ruderal herbaceous grasses and forbs

- Degraded vernal pool complex–Italian ryegrass (*Lolium multiflorum*)
- Degraded vernal pool complex–vernal pools
- Annual grasses generic
- Annual grasses/weeds
- *Bromus* spp./*Hordeum*
- *Hordeum*/*Lolium*
- *Lolium* (generic)
- *Lotus corniculatus*
- Medium upland graminoids
- Medium upland herbs
- Perennial grass
- Short upland graminoids
- Upland annual grasslands and forbs formation
- Upland herbs

- Alkali seasonal wetland complex
  - Distichlis spicata–annual grasses

In 2011, and again in 2012, the species habitat models were updated to include previously unmapped portions of the action area. For most areas newly mapped, vegetation data were not available at the alliance level as in the rest of the action area and so most of the new analysis areas were mapped at the natural community level. For California tiger salamander, in the new analysis areas, the following natural communities were assumed to provide terrestrial cover and aestivation habitat.

- Alkali seasonal wetland complex
- Grasslands
  - Upland annual grasslands & forbs formation

4.A.11.7.2 Assumptions

- **Assumption:** California tiger salamander terrestrial cover and aestivation habitat in the action area is geographically constrained to areas described in Section 4.A.11.7.1, *Terrestrial Cover and Aestivation Habitat Model Description*.

  Rationale: Habitat for the California tiger salamander includes vernal pools and seasonal and perennial ponds including artificial stock ponds in a grassland landscape (Barry and Shaffer 1994; 69 FR 47212; Bobzien and DiDonato 2007). Because the mapping of aquatic breeding habitats in the action area is incomplete, this element cannot be effectively used to model the extent of suitable habitat for this species. Thus, grasslands are used to more generally describe the extent of suitable habitat. Minimum patch size is 100 acres, which corresponds with the minimum conservation patch size identified by Trenham (2009). Grasslands located along the narrow eastern edge of Suisun Marsh that were contiguous with the larger grassland/agricultural landscape of the Montezuma Hills were reviewed and removed from the terrestrial cover and aestivation habitat component of the model because most appeared transitional to the tidal marsh wetlands that are not suitable for the California tiger salamander. The model is further constrained geographically by eliminating grasslands that are not within seasonal pool or pond/grassland landscapes, such as the central Delta. While periodic flooding may preclude the California tiger salamander from occurring in the Yolo Bypass, the vernal pool landscape on the CDFW Tule Ranch Unit and other similar areas on the CDFW Yolo Bypass Wildlife Area could potentially support this species in some years. These areas are mapped as alkali seasonal wetland complex (*Distichlis spicata*-annual grasses); however, they have a substantial grassland component. The model overestimates suitable habitat by assuming there are sufficient aquatic breeding habitats within the grassland landscape as defined.

4.A.11.7.3 Aquatic Breeding Habitat Model Description

Modeled aquatic breeding habitat for the California tiger salamander includes vernal pools and seasonal and perennial ponds. Aquatic breeding habitat includes the following land cover types and conditions that are within the grassland landscape as defined above.

- Vernal pool complex
  - *Allenrolfea occidentalis* mapping unit
  - Annual grasses generic
  - Annual grasses/weeds
  - California annual grasslands–herbaceous
  - *Distichlis* (generic)
  - *Distichlis*/annual grasses

- *Distichlis/S. maritimus*
- *Distichlis spicata*
- *Distichlis spicata*–annual grasses
- Italian ryegrass (*Lolium multiflorum*)
- Mix Schoenoplectus (formerly *Scirpus*) mapping unit
- Ruderal herbaceous grasses and forbs
- *Salicornia virginica* (formerly *Sarcocornia*)
- *Salicornia*/annual grasses
- Salt scalds and associated sparse vegetation
- Saltgrass (*Distichlis spicata*)
- Seasonally flooded grasslands
- Suadeda moquinii–(*Lasthenia californica*) mapping unit
- Vernal pools

In 2011, and again in 2012, the species habitat models were updated to include previously unmapped portions of the action area. For most areas newly mapped, vegetation data were not available at the alliance level as in the rest of the action area and so most of the new analysis areas were mapped at the natural community level. In the new analysis areas, the following natural community was assumed to provide terrestrial cover and aestivation habitat for the California tiger salamander.

- Vernal pool complex

4.A.11.7.4 Assumptions

- **Assumption:** California tiger salamander breeding habitat in the action area is geographically constrained to areas described in Section 4.A.11.7.3, *Aquatic Breeding Habitat Model Description*.

  Rationale: Aquatic breeding habitats are mapped to the extent data are available, but not used as a model attribute. The data for vernal pools and other seasonal wetlands and stock ponds are insufficient to effectively model California tiger salamander habitat on the basis of aquatic breeding habitat. Vernal pools and other seasonal rain pools are the primary breeding habitat of California tiger salamanders (Barry and Shaffer 1994; 68 FR 13498). California tiger salamander is also known to successfully reproduce in ponds, including artificial stock ponds (Barry and Shaffer 1994; 69 FR 47212). Stock pond habitats are used almost exclusively at occupied sites on the western edge of the action area and in the hills immediately west of the action area (Bobzien and DiDonato 2007). Mapping of vernal pools and other isolated seasonal wetlands and stock ponds is incomplete. In lieu of this, the vernal pool complex natural community was used to represent aquatic breeding habitat, which comprises a combination of aquatic and upland habitat that is considered suitable for the California tiger salamander. Potential habitat included within the vernal complex natural community not having concave surfaces or land uses that are incompatible with the species' habitat requirements were removed from the vernal pool complex and aquatic breeding habitat components of the model. For example, polygons falling on lands that did not have characteristic vernal pool/swale signatures that would demonstrate seasonal inundation did not qualify for this habitat type. In other instances, some other vernal pool aquatic features were located in areas that had unsuitable land uses. These features were removed by developing a GIS layer that excluded habitat from these locations. This element of the model overestimates the extent of potential breeding habitat.

4.A.12 Valley Elderberry Longhorn Beetle (*Desmocerus californicus dimorphus*)

4.A.12.1 Legal Status and Distribution

Valley elderberry longhorn beetle is listed as threatened under the Federal Endangered Species Act (ESA) (45 *Federal Register* [FR] 52803). On October 2, 2006, the U.S. Fish and Wildlife Service (USFWS), in their 5-year review, recommended this species be removed from the endangered species list (U.S. Fish and Wildlife Service 2006a). On October 2, 2012, USFWS issued a proposed rule to remove the species from the endangered species list (77 FR 60238). However, USFWS withdrew the proposed rule on September 17, 2014, based on their determination that the proposed rule did not fully analyze the best available information (79 FR 55873).

Valley elderberry longhorn beetle is one of three species of *Desmocerus* in North America and one of two subspecies of *D. californicus*. The valley elderberry longhorn beetle subspecies is a narrowly defined, endemic taxon, limited to portions of the Central Valley generally below 3,000 feet in elevation (U.S. Fish and Wildlife Service 1999a).

Historically, valley elderberry longhorn beetle presumably occurred throughout the Central Valley from Tehama County to Fresno County (79 FR 55880). The historic range was recently revised to no longer include Tulare and Shasta Counties (79 FR 55880). Little is known about the historical abundance of valley elderberry longhorn beetle. The extensive destruction of its habitat, however, suggests that the beetle's range has been largely reduced and fragmented (U.S. Fish and Wildlife Service 1984).

The current distribution of valley elderberry longhorn beetle is similar to its historic range, though it is "uncommon or rare, but locally clustered." Currently, valley elderberry longhorn beetle is known from 17 hydrologic units and 36 discrete geographical locations within the Central Valley (79 FR 55872–55873).

4.A.12.2 Life History and Habitat Requirements

Valley elderberry longhorn beetle is endemic to moist valley oak riparian corridors in the lower Sacramento and lower San Joaquin valleys (U.S. Fish and Wildlife Service 1984). Valley elderberry longhorn beetle is closely associated with elderberry (*Sambucus* spp.). These plants are an obligate host plant for larvae and are necessary for the completion of the life cycle (Eng 1984; Barr 1991; Collinge et al. 2001). The two main species of elderberry used by this species are the blue elderberry (*Sambucus nigra* subsp. *caerulea*, formerly *S. mexicana*) and red elderberry (*S. racemosa*). Blue elderberry is a component of riparian habitats throughout the Central Valley. Although this shrub occasionally occurs outside riparian areas, shrubs supporting the greatest beetle densities are located in areas where the shrubs are abundant and interspersed in significant riparian zones (Talley et al. 2006).

Adult valley elderberry longhorn beetles live for a few days to a few weeks between mid-March and mid-May, and are most active from late April to mid-May. The adult beetles feed on the elderberry foliage and possibly its flowers. During this time of activity, the beetles mate, and the female lays eggs on the living elderberry plant host. The eggs are typically placed individually or in small clusters within crevices in the bark or junctions of the stem and trunk or leaf petiole and stem. Eggs hatch within a few days and soft-bodied larvae emerge. The larvae are on the surface of the elderberry from a few minutes to several hours or a day and then bore to the center of the elderberry stems where they create a feeding gallery in the pith at the center of the stem. The larvae develop for 1 to 2 years feeding on pith. The late instar larvae chew through the inner bark, all or most of the way to the surface, then return inside plugging the holes with wood shavings. The larvae move back down the feeding gallery to an enlarged pupal chamber packed with frass. Here the larvae metamorphose into pupae between December and April (Talley et al. 2006).

The length of pupation is thought to be about one month with the emergent adult remaining in the chamber for up to several weeks. Adults complete the hole in the outer bark and emerge during the flowering season of elderberry shrubs. The exit holes are circular to oval and range in size from 4 to 10 millimeters in diameter (Talley et al. 2006).

4.A.12.3 Reasons for Decline

Habitat occupied by valley elderberry longhorn beetle tends to form and exist in riparian corridors and on the level open ground of periodically flooded river and stream terraces and floodplains. This geomorphic setting historically has been desirable for agricultural, urban, or industrial development. As a result, much of this habitat type has been converted, through the construction of dams and levees, to land that could be developed. Although it has been estimated that 90% of California riparian habitat has been lost over the last century and a half (Smith 1980; Barr 1991; Naiman et al. 1993; Naiman and Décamps 1997), these losses are difficult to accurately quantify in terms of valley elderberry longhorn beetle habitat losses (Talley et al. 2006). Therefore, an unknown amount of riparian forest and elderberry savannah habitat has been lost and an unknown number of valley elderberry longhorn beetle populations as well (Collinge et al. 2001).

The greatest historical threat to the valley elderberry longhorn beetle has been the elimination, loss, or modification of its habitat by urban, agricultural, or industrial development and other activities that reduce or eliminate its host plants (Talley et al. 2006). While mitigation and restoration actions do not come close to restoring the enormous amount of habitat lost in the more remote past, they appear to be adequate for current levels of impact (Talley et al. 2006). However Talley et al. (2006) observed that the quality and persistence of mitigation and restoration efforts are uncertain and that there have been declines in the total number of valley elderberry longhorn beetle–occupied sites and in the number of riparian sites. Talley et al. (2006) also noted that the information included in reports is often unusable, making assessments of mitigation and restoration success difficult.

Argentine ant (*Linepithema humile*) has been identified as a potential threat to valley elderberry longhorn beetle (Talley et al. 2006). This ant is an aggressive competitor and predator of native arthropods throughout riparian habitats in California, and has been observed preying on valley elderberry longhorn beetle larvae (Talley et al. 2006). Argentine ants have been inadvertently introduced into valley elderberry longhorn beetle mitigation sites from nursery stock and are able to proliferate there due to irrigation established for mitigation plantings (Argentine ants require moisture) (Talley et al. 2006).

The nonnative invasive European earwig (*Forficula auricularia*) is also considered to be a threat to the valley elderberry longhorn beetle through predation or by supporting higher populations of insect predators (Talley et al. 2006), although there is no distinct information to suggest that earwig predation or presence constitutes a specific threat to the beetle (77 FR 60237).

Nonnative invasive plant species such as black locust (*Robinia pseudoacacia*), giant reed (*Arundo donax*), red sesbania (*Sesbania punicea*), Himalaya blackberry (*Rubus armeniacus*), tree of heaven (*Ailanthus altissima*), Spanish broom (*Spartium junceum*), Russian olive (*Elaeagnus angustifolia*), edible fig (*Ficus carica*), and Chinese tallowtree (*Sapium sebiferum*), may have significant indirect impacts on the valley elderberry longhorn beetle by affecting elderberry shrub vigor and recruitment (Talley et al. 2006). Ripgut brome (*Bromus diandrus*), foxtail barley (*Hordeum murinum*), Italian ryegrass (*Festuca perennis*, formerly *Lolium multiflorum*), and yellow starthistle (*Centaurea solstitialis*) may impair elderberry germination or establishment, or elevate fire risk (Talley et al. 2006).

4.A.12.4 Status of the Species in the Action Area/Environmental Baseline

The current distribution of valley elderberry longhorn beetle in the action area is largely unknown. There are only three reported occurrences of valley elderberry longhorn beetle in the action area, including one along Middle River north of Tracy and two occurrences along small drainages between the Sacramento River and the Sacramento Deep Water Ship Channel in the vicinity of West Sacramento (California Department of Fish and Wildlife 2013). There are additional historical occurrences from along the Sacramento River corridor and Putah Creek in Yolo County (Jones & Stokes 1985, 1986, 1987; U.S. Fish and Wildlife Service 1984; Barr 1991; Collinge et al. 2001). Comprehensive surveys for the species or its host plant, elderberry, have not been conducted and thus the population size and location of the species in the action area is unknown. Distribution is typically based on the occurrence of elderberry shrubs, which are known to occur along riparian corridors throughout the action area, including the Sacramento River, Stanislaus River, San Joaquin River, and along smaller natural and channelized drainages, as well as in upland habitats.

4.A.12.5 Critical Habitat

The USFWS promulgated the final ruling designating critical habitat for valley elderberry longhorn beetle on August 8, 1980 (45 FR 52804). Two critical habitat areas were designated along portions of the American River in Sacramento County (the Sacramento Zone and the American River Parkway Zone). Critical habitat for valley elderberry longhorn beetle is not located within the action area.

4.A.12.6 Suitable Habitat Definition

As described above in Section 4.A.12.2, *Life History and Habitat Requirements*, and below in Section 4.A.12.7, *Species Habitat Suitability Model*, suitable habitat for valley elderberry longhorn beetle are elderberry shrubs throughout the action area. Elderberry shrubs in the action could be found in riparian areas, along levee banks, grasslands, and in agricultural settings where vegetation is not being maintained (e.g., fence rows, fallow fields).

4.A.12.7 Species Habitat Suitability Model

4.A.12.7.1 GIS Model Data Sources

The valley elderberry longhorn beetle model uses vegetation types and associations from the following data sets: composite vegetation layer (Hickson and Keeler-Wolf 2007 [Delta]; Boul and Keeler-Wolf 2008 [Suisun Marsh]; TAIC 2008 [Yolo Basin]; aerial photography (U.S. Department of Agriculture 2005), and land use survey of the Sacramento–San Joaquin River Delta (Delta) area-version 3, land use survey of the Delta and Suisun Marsh area - version 3 (California Department of Water Resources 2007). Using these data sets, the model maps the distribution of suitable valley elderberry longhorn habitat in the action area. Vegetation types were assigned based on the species requirements as described above and the assumptions described below.

4.A.12.7.2 Habitat Model Descriptions

Riparian modeled habitat in the Delta includes the following types from the composite vegetation layer.

- Valley/foothill riparian–all types

Riparian modeled habitat in the Suisun Marsh and Yolo Basin includes the following riparian types from the composite vegetation layer.

- Fraxinus latifolia
- Fremont cottonwood–valley oak–willow (ash–sycamore) riparian forest NFD alliance
- Mixed Fremont cottonwood–willow NFD alliance
- Mixed willow super alliance
- Quercus agrifolia
- Salix lasiolepis/Quercus agrifolia
- Valley oak alliance–riparian

Nonriparian channel and grassland modeled habitat in Suisun Marsh includes the following grassland and vernal pool complex types from the composite vegetation layer within 200 feet of streams.

- Annual grasses, generic
- Annual grasses/weed
- *Bromus* spp./*Hordeum*

- *Hordeum/Lolium*
- *Lolium* (generic)
- *Lotus corniculatus*
- Medium upland graminoids
- Medium upland herbs
- Perennial grass
- Short upland graminoids
- Upland annual grasslands and forbs formation
- Upland herbs
- Vernal pool complex types
    - *Distichlis* (generic)
    - *Distichlis spicata*
    - *Distichlis*/annual grasses
    - *Distichlis*/*Schoenoplectus maritimus* (formerly *Scirpus*)
    - *Salicornia virginica* (formerly *Sarcocornia*)
    - *Salicornia*/annual grasses

Nonriparian channels and grasslands modeled habitat in the Delta includes the following grassland and vernal pool complex types from the composite vegetation layer within 200 feet of streams.

- Grasslands–all types
- Vernal pool complex types
    - California annual grasslands–herbaceous
    - *Distichlis spicata*–Annual grasses
    - Ruderal herbaceous grasses and forbs
    - Italian ryegrass (*Lolium multiflorum*)

In 2011, and again in 2012, the species habitat models were updated to include previously unmapped portions of the action area. For most areas newly mapped, vegetation data were not available at the alliance level as in the rest of the action area and so most of the new analysis areas were mapped at the natural community level. In the new analysis areas, the following natural community was assumed to provide habitat for valley elderberry longhorn beetle.

- Valley/foothill riparian

While the valley elderberry longhorn beetle model remains unchanged, the model's use in the impact analysis has changed. Acres of impacted modeled habitat are now converted to an estimate of impacted shrubs and stems (with and without exit holes). The methods and assumptions for this new portion of the analysis are described in Table 6.B-2 in Appendix 6.B, *Terrestrial Impact Assessment Methods*.

4.A.12.7.3 Assumptions

- **Assumption:** Valley elderberry longhorn beetle habitat in the action area is restricted to areas and vegetative types described in Section 4.A.12.7.2, *Habitat Model Description*.

  Rationale: This model identifies habitat for the valley elderberry longhorn beetle as locations where the elderberry shrub is expected to be found in the action area and designates additional habitat as grasslands within 200 feet of streams. Note that elderberry shrubs are unevenly distributed along riparian corridors and adjacent upland habitats and in some areas may be lacking entirely. Thus, the model overestimates the extent of suitable habitat for valley elderberry longhorn beetle. Elderberry shrubs also occur incidentally along fence rows and in a variety of other disturbed conditions, particularly where birds may congregate and deposit seeds. This model does not include these incidental habitat areas and, thus, in this respect may underestimate the distribution of potential habitat (i.e., elderberry shrubs) for the valley elderberry longhorn beetle in the action area.

4.A.13 Vernal Pool Fairy Shrimp (*Branchinecta lynchi*)

4.A.13.1 Legal Status and Distribution

Vernal pool fairy shrimp is listed as threatened under the Federal Endangered Species Act (ESA) throughout its range (59 *Federal Register* [FR] 48136). In September 2007, the U.S. Fish and Wildlife Service (USFWS) published a 5-year review recommending that the species remain listed as threatened. In addition, on May 25, 2011, USFWS initiated a new 5-year review to determine if the species should remain listed as endangered.

There is little information on the historical range of vernal pool fairy shrimp. The species is currently known to occur in a wide range of vernal pool habitats in the southern and Central Valley areas of California, and in two vernal pool habitats in the Agate Desert area of Jackson County, Oregon (U.S. Fish and Wildlife Service 2005). It has the largest geographical range of listed fairy shrimp in California, but is seldom abundant (Eng et al. 1990). The species is currently found in fragmented habitats across the Central Valley of California from Shasta County to Tulare and Kings Counties, in the central and southern Coast Ranges from Napa County to Los Angeles County, and inland in western Riverside County, California (U.S. Fish and Wildlife Service 2005, 2007a; California Department of Fish and Wildlife 2013).

4.A.13.2 Life History and Habitat Requirements

Vernal pool fairy shrimp is entirely dependent on the aquatic environment provided by the temporary waters of natural vernal pool and playa pool ecosystems as well as the artificial environments of ditches and tire ruts (King et al. 1996; Helm 1998; Eriksen and Belk 1999). The temporary waters fill directly from precipitation as well as from surface runoff and perched groundwater from their watersheds (Williamson et al. 2005; Rains et al. 2006, 2008; O'Geen et al. 2008). The watershed extent that is necessary for maintaining the hydrological functions of the temporary waters depends on a number of complex factors including the hydrologic conductivity of the surface soil horizons, the continuity and extent of hardpans and claypans underlying nonclay soils, the existence of a perched aquifer overlying the pans, slope, effects of vegetation on evapotranspiration rates, compaction of surface soils by grazing animals, and other factors (Marty 2005; Pyke and Marty 2005; Williamson et al. 2005; Rains et al. 2006, 2008; O'Geen et al. 2008).

The temporary waters that are habitat for the vernal pool fairy shrimp are extremely variable and range from clear sandstone pools with little alkalinity to turbid vernal pools on clay soils with moderate alkalinity (King et al. 1996; Eriksen and Belk 1999). Vernal pool fairy shrimp have also occasionally been found in degraded vernal pool habitats and artificially created seasonal pools (Helm 1998). Vernal pool fairy shrimp commonly co-occur with other fairy shrimp and vernal pool tadpole shrimp (*Lepidurus packardi*) (U.S. Fish and Wildlife Service 2005).

Vernal pool fairy shrimp are adapted to the environmental conditions of their ephemeral habitats. One adaptation is the ability of vernal pool fairy shrimp cysts to remain dormant in the soil when their vernal pool habitats are dry. The cysts survive the hot, dry summers and cold, wet winters that follow until vernal pools and swales fill with rainwater and conditions are right for hatching. When the pools refill in the same or subsequent seasons some, but not all, of the cysts may hatch.

The cyst bank in the soil may comprise cysts from several years of breeding (U.S. Fish and Wildlife Service 2005, 2007a). Beyond inundation of the habitat, the specific cues for hatching are unknown, although temperature and conductivity (solute concentration) are believed to play a large role (Helm 1998; Eriksen and Belk 1999).

In a study using large plastic pools to simulate natural vernal pools, Helm found that vernal pool fairy shrimp can reproduce in as early as 18 days following hatching, with the average being 40 days (Helm 1998). Site-specific conditions, primarily water temperature, have been shown to affect time to reach reproductive maturity (Helm 1998).

4.A.13.3 Reasons for Decline

Threats to vernal pool habitat and vernal pool branchiopods in general, as well as specific threats to vernal pool fairy shrimp, are described in the *Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon* (U.S. Fish and Wildlife Service 2005, 2007a).

Habitat loss and fragmentation were identified as the largest threats to the survival and recovery of vernal pool species. Habitat loss generally is a result of agricultural conversion from rangelands to intensive farming, urbanization, aggregate mining, infrastructure projects (such as roads and utility projects), and recreational activities (such as off-highway vehicles and hiking) (U.S. Fish and Wildlife Service 2005). Habitat fragmentation occurs when vernal pool complexes are broken into smaller groups or individual vernal pools and become isolated from each other as a result of activities such as road development and other infrastructure projects (U.S. Fish and Wildlife Service 2005).

Inappropriate grazing practices include complete elimination of grazing in areas where nonnative grasses dominate the uplands surrounding vernal pools, and inappropriate timing or intensity of grazing. Appropriate grazing regimes help control nonnative weed plants such as Italian ryegrass (*Lolium multiflorum*) and waxy mannagrass (*Glyceria declinata Brébiss*), which, if unchecked, can increase thatch buildup, decrease ponding durations, and decrease the aquatic habitat available to the vernal pool fairy shrimp (U.S. Fish and Wildlife Service 2007a).

Human disturbances and changes in land use practices can alter the hydrology of temporary waters and result in a change in the timing, frequency, or duration of inundation in vernal pools, which can create conditions that render existing vernal pools unsuitable for vernal pool species (U.S. Fish and Wildlife Service 2005).

Climate change is expected to have an effect on vernal pool hydrology through changes in the amount and timing of precipitation inputs to vernal pools and the rate of loss through evaporation and evapotranspiration. It is unknown at this time if climate change in California will result in a localized, relatively small cooling and drying trend, or a warmer trend with higher precipitation events. However, it is possible that either scenario would result in negative effects on vernal pool invertebrate species. Cooling and drying trends could adversely affect vernal pool fairy shrimp through decreased inundation periods that do not allow the species sufficient time to complete its life cycle. In contrast, warmer conditions could increase inundation periods, which would not necessarily be a negative effect because increased inundation periods would increase available habitat for vernal pool fairy shrimp. However, increased inundation periods associated with a warming trend could also negatively affect the species by not providing cool enough temperatures for vernal pool fairy shrimp to hatch or reproduce (U.S. Fish and Wildlife Service 2007a).

Specific threats to vernal pool fairy shrimp habitat identified in the 2005 vernal pool recovery plan include the following.

- Within the entire range of the species, more than half of the known populations of vernal pool fairy shrimp are threatened by development or agricultural conversion. Several populations are found on military bases, and although not an immediate threat, military activities can result in alteration of pool characteristics, including introduction of nonnative plant species (U.S. Fish and Wildlife Service 2005, 2007a).

- In the Livermore Vernal Pool Region, the vernal pool fairy shrimp is located primarily on private land, where it is threatened by development, including expansion of the Byron Airport.

- In the Northeastern Sacramento Valley Vernal Pool Region, most of the known occurrences are located on California Department of Transportation (Caltrans) rights-of-way and are thus threatened by various future road improvement projects in this region, particularly the future expansion of SR 99. Additional populations are threatened by commercial and residential development projects.

- Some occurrences on private land in the Northwestern Sacramento Vernal Pool Region may be threatened by agricultural conversion or development.

- In the Southern Sacramento Vernal Pool Region, the vernal pool fairy shrimp is threatened by urban development. Both Sacramento and Placer Counties are currently developing habitat conservation plans to address growth in the region.

- In the San Joaquin Valley Region, the vernal pool fairy shrimp is found primarily on private land where it is threatened by direct habitat loss, including urban development and agricultural conversion.

- In the Solano-Colusa Region, the vernal pool fairy shrimp is threatened by development on the private property where it occurs.

4.A.13.4 Status of the Species in the Action Area/Environmental Baseline

Vernal pool fairy shrimp has been reported from several locations in the action area (U.S. Fish and Wildlife Service 2005, 2007a; California Department of Fish and Wildlife 2013). In general, in the action area, vernal pools that may support the species occur in Jepson Prairie, in the California Department of Fish and Wildlife Tule Ranch Unit of the Yolo Bypass Wildlife Area, in the Stone Lakes Wildlife Refuge, west of Clifton Court Forebay near the town of Byron, and along the eastern and northern boundary of Suisun Marsh. Other potential vernal pool habitat occurs along the eastern boundary of Stone Lakes. Vernal pool fairy shrimp were observed at seven locations in the south Stone Lakes area and in three locations in the Clifton Court Forebay during 2009 surveys conducted by the California Department of Water Resources. A comprehensive survey of vernal pools or habitat for vernal pool fairy shrimp has not been conducted in the action area.

4.A.13.5 Critical Habitat

The final rule designating critical habitat for vernal pool fairy shrimp was published in the *Federal Register* on February 10, 2006 (71 FR 7118–7316).

Designated critical habitat for vernal pool fairy shrimp is located along the northern margin of Suisun Marsh and west of Clifton Court Forebay near Byron. The designated critical habitat for vernal pool fairy shrimp is in Unit 11D (10,707 total acres; an estimated 9,579 acres in the action area). The primary constituent elements (PCEs) of critical habitat for vernal pool fairy shrimp are the habitat components listed below.

- Topographic features characterized by mounds and swales, and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools, providing for dispersal and promoting hydroperiods of adequate length in the pools.

- Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum time period (18 days for vernal pool fairy shrimp) in all but the driest years, thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

- Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding.

- Structure within the vernal pools, consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

4.A.13.6 Suitable Habitat Definition

As described above in Section 4.A.13.2, *Life History and Habitat Requirements*, and below in Section 4.A.13.7, *Species Habitat Suitability Model*, suitable habitat for vernal pool fairy shrimp includes vernal pools, seasonal wetlands, and alkali seasonal wetlands. Vernal pool fairy shrimp can also be found in artificial features such as seasonal ditches and un-vegetated low spots that pool during the winter.

4.A.13.7 Species Habitat Suitability Model

4.A.13.7.1 Habitat Model Description

The habitat model for vernal pool fairy shrimp was modified in 2015 to include the verified wetland delineation model (California Department of Water Resources 2015). The wetland (WaterGroup) type and Cowardin Class type assumed to provide potential vernal pool crustacean model habitat is detailed in Table 4.A.13-1, below. For the purpose of the impact analysis, when a vernal pool crustacean wetland type intersects with the water conveyance facility footprint the entire pool is considered affected for both direct and indirect impacts. Also for the purposes of this analysis, effects within 250 feet of the vernal pool are not assumed to affect the entire pool permanently. See Appendix 6.B, *Terrestrial Impact Analysis Methods*, for more detail.

Table 4.A.13-1. Wetland Types Selected from the Verified Wetland Data as Vernal Pool Crustacean Habitat.

Wetland Type	Cowardin Class
Playa Vegetated Natural	PEM
Playa Vegetated Unnatural	PEM
Vernal Pool	PEM2

4.A.13.7.2 Assumptions

- **Assumption:** The vernal pool fairy shrimp potentially occurs in vernal pool complexes throughout the action area.

  Rationale: This species is dependent on the aquatic environment provided by the temporary waters of natural vernal pool and playa pool ecosystems (King et al. 1996; Helm 1998; Eriksen and Belk 1999). Vernal pool fairy shrimp have been reported from several locations within vernal pool complexes in the action area (Figure 6.11-1).

- **Assumption:** Alkali seasonal wetlands provide high-value habitat for the vernal pool fairy shrimp.

  Rationale: Vernal pools in the western part of the action area tend to be alkali/saline pools of the *Lastenia fremontii-Distichlis spicata* alliance and *Frankenia salina* alliance (Sawyer et al. 2009). The alkali/saline vernal pool complexes often occur in a mosaic with alkali seasonal wetlands. Many of the species that occur in the vernal pool complex in this area also occur in the alkali seasonal wetland complex within this mosaic of natural communities.

- **Assumption:** Mapped degraded vernal pool complex and areas without concave surfaces as indicated by LiDAR data represent low-value habitat for the vernal pool fairy shrimp.

  Rationale: Mapped degraded vernal pool complex in the action area ranges from areas with vernal pool and swale visual signatures that display clear evidence of significant disturbance due to plowing, discing, or leveling to areas with clearly artificial basins such as shallow agricultural ditches, depressions in fallow fields, and areas of compacted soils in pastures. The aquatic features in this habitat generally do not hold water for as long as intact and fully functional vernal pools: in many cases the features become saturated but never pond, or only pond after the largest storm events. Additionally, the aquatic features in the degraded vernal pool complex are at much lower densities than in the intact vernal pool complexes. Because these features are saturated or inundated during the wet season and may have historically been located in or near areas with natural vernal pool complex, they may support individuals or small populations of species that are found in vernal pools and swales. However, they do not possess the full complement of ecosystem and community characteristics of natural vernal pools, swales, and their associated uplands, and they are generally ephemeral features that are eliminated during the course of normal agricultural practices.

Areas with appropriate soil conditions and for which no concave surfaces are apparent on the LiDAR data may include features that occasionally inundate but are too small or shallow to show up on the LiDAR imagery. If present, these features are likely occur at low densities and may be too ephemeral to support the species. However, because these areas do have the potential to support the species at low densities, they were classified as low-value habitat.

4.A.14 Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)

4.A.14.1 Legal Status and Distribution

Vernal pool tadpole shrimp was listed as endangered throughout its range under the Federal Endangered Species Act (ESA) on September 19, 1994 (59 *Federal Register* [FR] 48136). In September, 2007, the U.S. Fish and Wildlife Service (USFWS) published a 5-year review recommending that the species remain listed as endangered. In addition, on May 25, 2011, USFWS initiated a new 5-year review to determine if the species should remain listed as endangered.

Historically, vernal pool tadpole shrimp probably did not occur outside of the Central Valley and Central Coast regions (U.S. Fish and Wildlife Service 2005). Currently, vernal pool tadpole shrimp occurs in the Central Valley of California and in the San Francisco Bay area. The species has a patchy distribution across the Central Valley of California from Shasta County southward to northwestern Tulare County (U.S. Fish and Wildlife Service 2007a). In the Central Coast Vernal Pool Region, the vernal pool tadpole shrimp is found the San Francisco National Wildlife Refuge and on private land in Alameda County near Milpitas (U.S. Fish and Wildlife Service 2007a; California Department of Fish and Wildlife 2013). The largest concentration of vernal pool tadpole shrimp occurrences is found in the Southeastern Sacramento Vernal Pool Region, where the species occurs on a number of public and private lands in Sacramento County (U.S. Fish and Wildlife Service 2005, 2007a).

4.A.14.2 Life History and Habitat Requirements

Vernal pool tadpole shrimp occur in a wide variety of seasonal habitats, including vernal pools, ponded clay flats, alkaline pools, ephemeral stock tanks, and roadside ditches. Habitats where vernal pool tadpole shrimp have been observed range in size from small (less than 25 square feet), clear, vegetated vernal pools to highly turbid alkali scald pools to large (more than 100 acres) winter lakes (Helm 1998:134–138; Rogers 2001:1002–1005). These pools and other ephemeral wetlands must dry out and be inundated again for the vernal pool tadpole shrimp cysts to hatch. This species has not been reported in pools that contain high concentrations of sodium salts, but may occur in pools with high concentrations of calcium salts (Helm 1998:134–138; Rogers 2001:1002–1005).

Vernal pool tadpole shrimp commonly co-occur with other fairy shrimp (U.S. Fish and Wildlife Service 2005).

Like other vernal pool branchiopods, vernal pool tadpole shrimp are adapted to the environmental conditions of their ephemeral habitats. One adaptation is the ability of vernal pool tadpole shrimp eggs, or cysts, to remain dormant in the soil when their vernal pool habitats are dry. The cysts survive the hot, dry summers and cold, wet winters that follow until the vernal pools and swales fill with rainwater and conditions are right for hatching. When the pools refill in the same or subsequent seasons some, but not all, of the cysts may hatch. The cyst bank in the soil may comprise cysts from several years of breeding (U.S. Fish and Wildlife Service 2005, 2007a). Beyond inundation of the habitat, the specific cues for hatching are unknown, although temperature and conductivity (solute concentration) are believed to play a large role (Helm 1998; Eriksen and Belk 1999).

In a study using large plastic pools to simulate natural vernal pools, Helm found that vernal pool tadpole shrimp can reproduce as early as 41 days following hatching with the average being 54 days (Helm 1998). Site-specific conditions, primarily water temperature, have been shown to affect time to reach reproductive maturity (Helm 1998).

Vernal pool tadpole shrimp have relatively high reproductive rates and may be hermaphroditic. Sex ratios can vary, perhaps in response to changes in water temperature (Ahl 1991). Genetic variation among vernal pool tadpole shrimp corresponded with differences between sites in physical and chemical aspects of the pool habitat (depth, surface area, solutes concentration, elevation, and biogeographic region), and species richness was positively correlated with both depth and surface area (King et al. 1996). This result corresponds with the findings of other researchers that vernal pool crustaceans have low rates of gene flow between separated sites (U.S. Fish and Wildlife Service 2005). The low rate of exchange between vernal pool tadpole shrimp populations is probably a result of the spatial isolation of their habitats and their reliance on passive dispersal mechanisms (U.S. Fish and Wildlife Service 2005). However, the studies also found that gene flow between pools within the same vernal pool complex is much higher (U.S. Fish and Wildlife Service 2005). This indicates that vernal pool tadpole shrimp populations, like most vernal pool crustacean populations, are defined by vernal pool complexes and not by individual vernal pools (U.S. Fish and Wildlife Service 2005).

4.A.14.3 Reasons for Decline

Threats to vernal pool habitat and vernal pool branchiopods in general, as well as specific threats to vernal pool tadpole shrimp, are identified in the *Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon* (U.S. Fish and Wildlife Service 2005).

Habitat loss and fragmentation were identified as the largest threats to the survival and recovery of vernal pool species. Habitat loss generally is a result of agricultural conversion from rangelands to intensive farming, urbanization, aggregate mining, infrastructure projects (such as roads and utility projects), and recreational activities (such as off-highway vehicles and hiking) (U.S. Fish and Wildlife Service 2005). Habitat fragmentation occurs when vernal pool complexes are broken into smaller groups or individual vernal pools and become isolated from each other as a result of activities such as road development and other infrastructure projects (U.S. Fish and Wildlife Service 2005).

Inappropriate grazing practices include complete elimination of grazing in areas where nonnative grasses dominate the uplands surrounding vernal pools, and inappropriate timing or intensity of grazing. Appropriate grazing regimes help control nonnative weed plants such as Italian ryegrass (*Lolium multiflorum*) and waxy mannagrass (*Glyceria declinata Brébiss*), which if unchecked can increase thatch buildup and decrease ponding durations and decrease the aquatic habitat available to the vernal pool tadpole shrimp (U.S. Fish and Wildlife Service 2007a).

Human disturbances and changes in land use practices can alter the hydrology of temporary waters and result in a change in the timing, frequency, or duration of inundation in vernal pools, which can create conditions that render existing vernal pools unsuitable for vernal pool species (U.S. Fish and Wildlife Service 2005).

Climate change is expected to have an effect on vernal pool hydrology through changes in the amount and timing of precipitation inputs to vernal pools and the rate of loss through evaporation and evapotranspiration. It is unknown at this time if climate change in California will result in a localized, relatively small cooling and drying trend, or a warmer trend with higher precipitation events. However, it is possible that either scenario would result in negative effects on vernal pool invertebrate species. Cooling and drying trends could adversely affect the vernal pool tadpole shrimp through decreased inundation periods that do not allow the species sufficient time to complete its life cycle. In contrast, warmer conditions could increase inundation periods, which would not necessarily be a negative effect because increased inundation periods would increase available habitat for the vernal pool tadpole shrimp. However, increased inundation periods associated with a warming trend could also negatively affect the species by not providing cool enough temperatures for the vernal pool tadpole shrimp to hatch or reproduce (U.S. Fish and Wildlife Service 2007a).

Specific threats to vernal pool tadpole shrimp habitat identified in the 2005 vernal pool recovery plan included the following.

- The species is threatened by the encroachment of nonnative annual grasses on the San Francisco Bay National Wildlife Refuge in the Central Coast Region, and by urban development where it is known to occur on private land in Alameda County.

- In the Northeastern Sacramento Valley Region, most of the known occurrences of the vernal pool tadpole shrimp are on California Department of Transportation (Caltrans) rights-of-way, where they continue to be threatened by road improvement projects related to general urban growth.

- In the Northwestern Sacramento Valley Vernal Pool Region, the vernal pool tadpole shrimp is threatened by development on the few sites on private land where it is known to occur.

- In the Southeastern Sacramento Vernal Pool Region, extant populations of the vernal pool tadpole shrimp are threatened by continued extensive urban development.

- In the San Joaquin Vernal Pool Region, the species is threatened by development on private land.

- In the Solano-Colusa Region, the species is threatened by urbanization on private lands.

- In the Southern Sierra Foothills Vernal Pool Region, the species is threatened by development of the University of California, Merced campus, which will likely contribute to significant growth in the region. Populations on the Stone Corral Ecological Reserve may be threatened by pesticide drift from adjacent farmlands.

4.A.14.4 Status of the Species in the Action Area/Environmental Baseline

Vernal pool tadpole shrimp has been reported from several locations in the action area (U.S. Fish and Wildlife Service 2005, 2007a; California Department of Fish and Wildlife 2013). In general, within the action area, vernal pools that may support the species occur in Jepson Prairie, in 'California Department of Fish and Wildlife's Tule Ranch Unit of the Yolo Bypass Wildlife Area, in the Stone Lakes, west of Clifton Court Forebay near the town of Byron, and along the eastern and northern boundary of Suisun Marsh. Vernal pool tadpole shrimp was found in six locations in the Stone Lakes area during 2009 surveys conducted by the California Department of Water Resources. A comprehensive survey of vernal pools or habitat for the vernal pool tadpole shrimp has not been conducted in the action area.

4.A.14.5 Critical Habitat

Final designation of critical habitat for vernal pool tadpole shrimp was published in the *Federal Register* on February 10, 2006 (71 FR 7118–7316). Designated critical habitat for vernal pool tadpole shrimp is located along the northern margin of Suisun Marsh, outside the action area. The primary constituent elements (PCEs) of critical habitat for vernal pool tadpole shrimp are the habitat components listed below.

- Topographic features characterized by mounds and swales, and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described in PCE (2), providing for dispersal and promoting hydroperiods of adequate length in the pools.

- Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum time period (41 days for vernal pool tadpole shrimp) in all but the driest years, thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

- Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding.

- Structure within the vernal pools, consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

4.A.14.6 Suitable Habitat Definition

As described above in Section 4.A.14.2, *Life History and Habitat Requirements*, and below in Section 4.A.14.7, *Species Habitat Suitability Model*, suitable habitat for vernal pool tadpole shrimp includes vernal pools, seasonal wetlands, and alkali seasonal wetlands. Vernal pool tadpole shrimp generally occur in pools that inundate for longer periods of time than those of other vernal pool crustaceans; however, for the purposes of this analysis the habitat for vernal pool fairy shrimp and vernal pool tadpole shrimp are treated as equivalent.

4.A.14.7 Species Habitat Suitability Model

4.A.14.7.1 Habitat Model Description

The habitat model for vernal pool fairy shrimp was modified in 2015 to include the verified wetland delineation model (California Department of Water Resources 2015). The wetland (WaterGroup) type and Cowardin Class type assumed to provide potential vernal pool crustacean model habitat is detailed in Table 4.A.14-1. For the purpose of the impact analysis, when a vernal pool crustacean wetland type intersects with the water conveyance facility footprint the entire pool is considered affected for both direct and indirect impacts. Also for the purposes of this analysis, effects within 250 feet of the vernal pool are not assumed to affect the entire pool permanently. See Appendix 6.B, *Terrestrial Impact Analysis Methods*, for more detail.

Table 4.A.14-1. Wetland Types Selected from the Verified Wetland Data as Vernal Pool Crustacean Habitat.

Wetland Type		Cowardin Class
Alkali Wetlands	Playa Vegetated Natural	PEM
		PSS
Seasonal Wetlands	Playa Vegetated Unnatural	PEM
		PSS
Vernal Pool	Vernal Pool	PEM2

4.A.14.7.2 Assumptions

- **Assumption:** The vernal pool tadpole shrimp potentially occurs in vernal pool complexes throughout the action area.

  Rationale: This species is dependent on the aquatic environment provided by the temporary waters of natural vernal pool and playa pool ecosystems (King et al. 1996; Helm 1998; Eriksen and Belk 1999). The vernal pool tadpole shrimp has been reported from several locations within vernal pool complexes in the action area (Figure 6.11-1).

- **Assumption:** Alkali seasonal wetlands provide high-value habitat for the vernal pool tadpole shrimp.

  Rationale: Vernal pools in the western part of the action area tend to be alkali/saline pools of the *Lastenia fremontii-Distichlis spicata* alliance and *Frankenia salina* alliance (Sawyer et al. 2009). The alkali/saline vernal pool complexes in the western part of the action area often occur in a mosaic with alkali seasonal wetlands. Many of the species that occur in the vernal pool complex in this area also occur in the alkali seasonal wetland complex within this mosaic of natural communities.

- **Assumption:** Mapped degraded vernal pool complex and areas without concave surfaces as indicated by LiDAR data represent low-value habitat for the vernal pool tadpole shrimp.

  Rationale: Mapped degraded vernal pool complex in the action area ranges from areas with vernal pool and swale visual signatures that display clear evidence of significant disturbance due to plowing, discing, or leveling to areas with clearly artificial basins such as shallow agricultural ditches, depressions in fallow fields, and areas of compacted soils in pastures. The aquatic features in this habitat generally do not hold water as long as intact and fully functional vernal pools: in many cases the features become saturated but never pond, or only pond after the largest storm events. Additionally, the aquatic features in the degraded vernal pool complex are at much lower densities than the intact vernal pool complexes. Because these features are saturated or inundated during the wet season and may have historically been located in or near areas with natural vernal pool complex, they may support individuals or small populations of species that are found in vernal pools and swales. However, they do not possess the full complement of ecosystem and community characteristics of natural vernal pools, swales, and their associated uplands, and they are generally ephemeral features that are eliminated during the course of normal agricultural practices.

  Areas with appropriate soil conditions and for which no concave surfaces are apparent on the LiDAR data may include features that occasionally inundate but are too small or shallow to show up on the LiDAR imagery. If present, these features are likely to occur at low densities and may be too ephemeral to support the species. However, because these areas do have the potential to support the species at low densities, they were classified as low-value habitat.

4.A.15 Least Bell's Vireo (*Vireo bellii pusillus*)

4.A.15.1 Legal Status and Distribution

Least Bell's vireo was listed as endangered under the Federal Endangered Species Act (ESA) on May 2, 1986 (51 *Federal Register* [FR] 16474–16482). The species is also listed as endangered under the California ESA.

Least Bell's vireo is one of four subspecies of Bell's vireo and is the only subspecies that breeds entirely in California and northern Baja California. Arizona Bell's vireo (*V. bellii arizonae*) is found along the Colorado River and may occur on the California side, but otherwise occurs throughout Arizona, Utah, Nevada, and Sonora, Mexico (Kus 2002a).

Least Bell's vireo, a riparian obligate, had a historical distribution that extended from coastal southern California through the San Joaquin and Sacramento Valleys as far north as Tehama County near Red Bluff (Kus 2002a) (Figure 2A.20-1 in California Department of Water Resources 2013). The Sacramento and San Joaquin Valleys were the center of the historical breeding range supporting 60 to 80% of the population (51 FR 16474). Least Bell's vireo also occurred along western Sierra Nevada foothill streams and in riparian habitats of the Owens Valley, Death Valley, and Mojave Desert (Cooper 1861 and Belding 1878 in Kus 2002a; Grinnell and Miller 1944). Least Bell's vireo was reported in Grinnell and Miller (1944) from elevations ranging from -175 feet in Death Valley to 4,100 feet in Bishop, Inyo County. These and other historical accounts described the subspecies as common to abundant (Kus 2002a), but no reliable population estimates are available prior to the Federal listing of least Bell's vireo in 1986.

Coinciding with widespread loss of riparian vegetation throughout California (Katibah 1983), Grinnell and Miller (1944) began to detect population declines in the Sacramento and San Joaquin Valleys by the 1930s. Surveys conducted in late 1970s (Goldwasser et al. 1980) detected no least Bell's vireos in the Sacramento and San Joaquin Valleys, and the subspecies was considered extirpated from the region. By 1986, USFWS determined that least Bell's vireo had been extirpated from most of its historical range and numbered approximately 300 pairs statewide (51 FR 16474).

The historical range was reduced to six California counties south of Santa Barbara, with the majority of breeding pairs in San Diego County (77%), Riverside County (10%), and Santa Barbara County (9%) (51 FR 16474).

Since Federal listing in 1986, populations have gradually increased, and the subspecies has recolonized portions of its historical range. Increases are attributed primarily to riparian restoration and efforts to control the brood parasite brown-headed cowbird (Kus 1998 and Kus and Whitfield 2005 in Howell et al. 2010). By 1998, the total population was estimated at 2,000 pairs and recolonization was reported along the Santa Clara River in Ventura County, the Mojave River in San Bernardino County, and sites in Monterey and Inyo Counties (Kus and Beck 1998; Kus 2002a; U.S. Fish and Wildlife Service 2006c). A single nest was reported from Santa Clara County near Gilroy in 1997 (Roberson et al. 1997). Still, the distribution remained largely restricted to San Diego County (76%) and Riverside County (16%) (U.S. Fish and Wildlife Service 2006c).

By 2005, the population had reached an estimated 2,968 breeding pairs (U.S. Fish and Wildlife Service 2006c) with increases in most southern California counties and San Diego County (primarily Camp Pendleton Marine Corps Base) supporting roughly half of the current population (U.S. Fish and Wildlife Service 2006c). Two recent nesting events, 2005 and 2006 at the San Joaquin River National Wildlife Refuge, and 2010 and 2011 along Putah Creek in Yolo Bypass, indicate the species is attempting to recolonize the Central Valley.

4.A.15.2 Life History and Habitat Requirements

Least Bell's vireo is an obligate riparian breeder. The Least Bell's Vireo typically breeds in willow riparian forest supporting a dense, shrubby understory of mulefat (Baccharis salicifolius) and other mesic species (Goldwasser, 1981; Gray and Greaves, 1984; Franzreb, 1989). Oak woodland with a willow riparian understory is also used in some areas (Gray and Greaves, 1984), and individuals sometimes enter adjacent chaparral, coastal sage scrub, or desert scrub habitats to forage (Brown 1993). Similar habitats are used during the winter months. Goldwasser (1981) and Salata (1983) believed that structure and composition of vegetation below 3 and 4 m, respectively, were critical. Salata (1983) also reported the importance of a mix of tree size classes, with a mean height of 8 m. Gray and Greaves (1984) recommended protection of ground cover and low shrub layers. Vireos occur in disproportionately high frequencies in the wider sections (greater than 250m) of the riparian relative to site availability (RECON 1989).

Early successional riparian habitat typically supports the dense shrub cover required for nesting and a diverse canopy for foraging. Although least Bell's vireo tends to prefer early successional habitat, breeding site selection does not appear to be limited to riparian stands of a specific age. If willows and other species are not managed, within 5 to 10 years they form dense thickets and become suitable nesting habitat (Goldwasser 1981; Kus 1998). Tall canopy tends to shade out the shrub layer in mature stands, but least Bell's vireo will continue to use such areas if patches of understory exist. In mature habitat, understory vegetation consists of species such as California wild rose (*Rosa californica*), poison oak (*Toxicodendron diversilobum*), California blackberry (*Rubus ursinus*), grape (*Vitis californica*), and perennials that can conceal nests.

Least Bell's vireos use upland habitat, in many cases coastal sage scrub, adjacent to riparian habitat. Vireos along the edges of riparian corridors maintain territories that incorporate both habitat types, and a significant proportion of pairs with territories encompassing upland habitat place at least one nest there (Kus and Miner 1989).The Least Bell's Vireo arrives on its breeding grounds in mid-March (Brown, 1993), with males arriving slightly before females (Nolan, 1960; Barlow, 1962). This vireo shows a high degree of nest site tenacity (Greaves, 1987). Most individuals depart by September (Brown, 1993), although some individuals remain on their breeding grounds into late November (Rosenberg et al., 1991).

Least Bell's vireos winter in Baja California Peninsula. Unlike during the breeding season, they are not limited in winter to willow-dominated riparian areas, but occupy a variety of habitats including mesquite scrub within arroyos, palm groves, and hedgerows bordering agricultural and residential areas. Uplands adjacent to riparian areas provide migratory stopover grounds, foraging habitat, and dispersal corridors for nonbreeding adults and juveniles (Kus and Miner 1989; Riparian Habitat Joint Venture 2004).

Territory size ranges from 0.5 to 7.5 acres (0.2 to 3 hectares), but on average are between 1.5 and 2.5 acres (0.6 and 1 hectare) in southern California (U.S. Fish and Wildlife Service 1998). Spatial differences in riparian habitat structure, patch size, and numerous other factors result in differences in the density of territories within and between drainages. Patch size and crowding did not influence least Bell's vireo reproductive success, at least not through the mechanisms of singing rates and attraction of predators (U.S. Fish and Wildlife Service 1998).

Least Bell's vireos are insectivorous and prey on a wide variety of insects, including bugs, beetles, grasshoppers, moths, and especially caterpillars (Chapin 1925; Bent 1950). They obtain prey primarily by foliage gleaning (picking prey from leaf or bark substrates) and hovering (removing prey from vegetation surfaces while fluttering in the air). Foraging occurs at all levels of the canopy but appears to be concentrated in the lower to middle level strata, particularly when pairs have active nests (Grinnell and Miller 1944; Goldwasser 1981; Gray and Greaves 1981; Salata 1983). U.S. Fish and Wildlife Service (1998) determined that least Bell's vireo foraging time across heights was not simply a function of the availability of vegetation at those heights, but rather represented an actual preference for the 10- to 20-foot (3- to 6-meter) zone. Foraging occurs most frequently in willows (Salata 1983; U.S. Fish and Wildlife Service 1998), but occurs on a wide range of riparian species and even some nonriparian plants that may host relatively large proportions of large prey (U.S. Fish and Wildlife Service 1998).

4.A.15.3 Reasons for Decline

Loss of habitat, combined with increased brood parasite pressure from Brown-headed Cowbirds (Goldwasser, 1978; Beezley and Rieger, 1987), are the major factors leading to the significant declines in populations of the Least Bell's Vireo (Franzreb, 1989; Franzreb et al., 1992; Salata, 1992; U.S. Fish and Wildlife Service, 1992).

Habitat loss and degradation can occur through clearing of vegetation for agriculture, timber harvest, development, or flood control (U.S. Fish and Wildlife Service 1998).Flood control and river channelization eliminates early successional riparian habitat that least Bell's vireo use for breeding. Dams, levees, and other flood control structures hinder riparian re-establishment, creating more old-growth conditions (dense canopy and open understory) that are unfavorable to breeding vireos. Finally, habitat degradation encourages nest predation and parasitism. Agricultural land uses, flood control projects and river and stream flow manipulation not only directly destroy habitat, but may also reduce water tables to levels that inhibit the growth of the dense vegetation least Bell's vireo prefer (Riparian Habitat Joint Venture 2004).
Grazing can also have a significant effect on riparian vegetation (Sedgwick and Knopf 1987). Cattle and other livestock can trample vegetation and eat seedlings, saplings, shrubs, and herbaceous plants. This can lead to a reduction in cover and nesting sites, and affect insect prey populations.

Brood parasitism from brown-headed cowbirds (Molothrus ater) has a major negative impact on least Bell's vireo. Livestock grazing has reduced and degraded the lower riparian vegetation favored by least Bell's vireo (Overmire 1962) and provided foraging areas for brown-headed cowbird. Sharp and Kus (2006) suggest that microhabitat cover around the nest is the most important habitat feature influencing brood parasitism of least Bell's vireo nests. They found non-parasitized nests had fewer trees greater than 3 inches in diameter at breast height within 37 feet of the nest and had less canopy cover within 16 feet than parasitized nests. They also suggest that cover near the nest reduces the chance that a cowbird will observe nesting activity and later parasitize the nest.

Row crops and orchards also provide feeding grounds for brown-headed cowbirds. Young and Hutto (1999) found that distance to agriculture was the strongest predictor of cowbird presence and abundance. Riparian habitat that is fragmented by agriculture is therefore highly susceptible to cowbird brood parasitism. By as early as 1930, nearly every least Bell's vireo nest found in California hosted at least one cowbird egg (U.S. Fish and Wildlife Service 1998). Because a parasitized nest rarely fledges any vireo young, nest parasitism of least Bell's vireo results in drastically reduced nest success (Goldwasser 1978; Goldwasser et al. 1980; Franzreb 1989; Kus 1999, 2002b).

Predation is a major cause of nest failure in areas where brown-headed cowbird nest parasitism is infrequent or has been reduced by cowbird trapping programs. Most predation occurs during the egg stage. Predators likely include western scrub jays (Aphelocoma californica), Cooper's hawks (Accipiter cooperii), gopher snakes (Pituophis melanoleucus) and other snake species, raccoons (Procyon lotor), opossums (Didelphis virginiana), coyotes (Canis latrans), long-tailed weasels (Mustela frenata), dusky-footed woodrats (Neotoma fuscipes), deer mice (Peromyscus maniculatus), rats (Rattus spp.), and domestic cats (Felis domesticus) (Franzreb 1989). Kus et al. (2008) investigated variables that influenced the likelihood of nest predation on least Bell's vireo at three spatial scales. They did not find strong predictors of predation risk at the nest site, surrounding habitat, or landscape scale, with the exception of proximity to golf courses, parks, and wetlands. Nest predation increased with proximity to golf courses, whereas nests near wetland habitats were twice as likely to succeed as those that were farther from wetlands (Kus et al. 2008).

4.A.15.4 Status of the Species in the Action Area/Environmental Baseline

Data on Least Bell's Vireos from the 1940s through the 1960s are lacking, but extensive surveys of the Central Valley in the late 1970s did not detect a single individual (Goldwasser et al. 1980). Least Bell's vireos are rarely observed in the Central Valley; according to eBird, the species has been observed at 7 distinct locations between 2005 and 2013. No individuals have been observed in the Central Valley in the last 3 years. There are no California Natural Diversity Database records of least Bell's vireos breeding in the action area since at least the 1970s. Two singing males were detected in the Yolo Bypass Wildlife Area in mid-April 2010, and again in 2011 (California Department of Fish and Wildlife 2013). No least Bell's vireos were detected in the Yolo Bypass Wildlife Area during surveys in 2012. A singing male was detected in 2013, and surveys were not conducted in 2014 (Whisler pers. comm. 2015). No least Bell's vireos were detected in the Yolo Bypass in 2015 or 2016, and the site appears to have been abandoned.

The next-nearest known nest site since the 1930sis approximately 7 miles south of the action area at the San Joaquin River National Wildlife Refuge in the San Joaquin and Tuolumne River floodplain (Howell et al. 2010). This occurrence includes three nests between 2005 and 2007, all in a recently restored portion of San Joaquin River National Wildlife Refuge lands known as "Hagemann's Fields 6 and 9." The 2005 and 2006 nests were successful. The 2007 nest was not successful in that only a female returned to the area, and though it constructed a nest and laid eggs, the nest failed. The 2005 and 2006 nest were in a 3-year-old arroyo willow with understory plants including mugwort, sunflower, gumplant, and creeping wild rye. The 2007 nest was in a dead arroyo willow (Howell et al. 2010).

Least Bell's vireos have not been detected within or around the project construction sites. Few least Bell's vireos have been detected north of the project area; those birds may migrate through the action area, but may not migrate through, or stop over in, the construction disturbance area.

4.A.15.5 Critical Habitat

Final designation of critical habitat for least Bell's vireo was published in the *Federal Register* on February 2, 1994 (59 FR 14845-4867). There is no designated critical habitat for least Bell's vireo in the action area.

4.A.15.6 Suitable Habitat Definition

Early successional riparian habitat typically supports the dense shrub cover required for nesting and a diverse canopy for foraging. Although least Bell's vireo tends to prefer early successional habitat, breeding site selection does not appear to be limited to riparian stands of a specific age. If willows and other species are not managed, within 5 to 10 years they form dense thickets and become suitable nesting habitat (Goldwasser 1981; Kus 1998). Tall canopy tends to shade out the shrub layer in mature stands, but least Bell's vireo will continue to use such areas if patches of understory exist. In mature habitat, understory vegetation consists of species such as California wild rose (Rosa californica), poison oak (Toxicodendron diversilobum), California blackberry (Rubus ursinus), grape (Vitis californica), and perennials that can conceal nests. Least Bell's vireos use upland habitat, in many cases coastal sage scrub, adjacent to riparian habitat. Vireos along the edges of riparian corridors maintain territories that incorporate both habitat types, and a significant proportion of pairs with territories encompassing upland habitat place at least one nest there (Kus and Miner 1989).Unlike during the breeding season, least Bell's vireos are not limited in winter to willow-dominated riparian areas, but occupy a variety of habitats including mesquite scrub within arroyos, palm groves, and hedgerows bordering agricultural and residential areas. Uplands adjacent to riparian areas provide migratory stopover grounds, foraging habitat, and dispersal corridors for nonbreeding adults and juveniles (Kus and Miner 1989; Riparian Habitat Joint Venture 2004).

4.A.15.7 Species Habitat Suitability Model

4.A.15.7.1 GIS Model Data Sources

The least Bell's vireo model uses vegetation types from the following data sets: composite vegetation layer (Hickson and Keeler-Wolf 2007 [Delta], Boul and Keeler-Wolf 2008 [Suisun Marsh], TAIC 2008 [Yolo Basin]), and aerial photography (U.S. Department of Agriculture 2005). Using these data sets, the model maps the distribution of suitable least Bell's vireo nesting and migratory habitat in the action area. Vegetation types were assigned based on the species requirements as described above and the assumptions described below.

4.A.15.7.2 Habitat Model Description

Modeled nesting and migratory habitat in the Sacramento–San Joaquin River Delta includes all vegetation types within the valley-foothill riparian category.

4.A.15.7.3 Assumptions

- **Assumption:** Least Bell's vireo habitat is restricted to the vegetation types described in Section 4.A.15.7.2, *Habitat Model Description*.

  Rationale: Although it can use adjacent nonriparian scrub habitats for foraging or migration (Kus and Miner 1989; Riparian Habitat Joint Venture 2004), suitable nonriparian habitats are largely absent from the action area, which is primarily agricultural. Therefore, the habitat model is restricted to riparian vegetation.

4.A.16 References Cited

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