Chapter 2 Status of Aquatic and Terrestrial Species and Designated Critical Habitat

2.1 Sacramento River Winter-Run Chinook Salmon ESU

2.1.1 ESA Listing Status

NMFS, under an emergency interim rule, listed the Sacramento River Winter-Run Chinook Salmon Evolutionarily Significant Unit (ESU) as a threatened species under the ESA in August 1989 (54 Federal Register [FR] 32085). In 1994, NMFS reclassified the ESU as endangered due to several factors: the continued decline and increased variability of run sizes including expected weak returns due to small year classes in 1991 and 1993, and continuing threats to the species (59 FR 440). The ESU consists of one population in the mainstem of the Upper Sacramento River in California's Central Valley below Keswick Dam. NMFS reaffirmed the listing of the Sacramento River Winter-Run Chinook Salmon ESU as endangered on June 28, 2005 (70 FR 37160), and expanded the ESU to include Winter-Run Chinook Salmon produced by the Livingston Stone National Fish Hatchery (LSNFH) artificial propagation program in the ESU.

On May 26, 2016, after a third 5-year status review (81 FR 33468), NMFS (2016a) determined that the viability of the ESU had continued to decline on average 15 percent per year, or from 38 percent to 67 percent since 2010. Although the population size and catastrophe rate and effect have remained at the low-risk threshold (<90 percent decline in one generation, or annual run size of less than 500 spawners) since the 2010 status review, the population decline and hatchery influence criteria have both been elevated to a moderate extinction risk (NMFS 2016a). NMFS concluded that the ESU classification as an endangered species is appropriate and should be maintained (NMFS 2016a).

2.1.2 General Life-History and Habitat Requirements

Chinook Salmon in the Central Valley have four distinct races: Fall, Late-Fall, Winter, and Spring. The name of the runs come from the peak migration timing with peak runs for Fall-Run occurring during August to November, late-fall occurring November to February, Winter-Run January to May, and Spring-Run April to August (Vogel and Marine 1991). Fall and late-fall enter as mature and ready to spawn. Winter-Run and Spring-Run return immature and hold in the river until reaching maturity. The adults enter freshwater in an immature state and migrate far upstream where spawning is delayed for weeks or months (Healey 1991). Juveniles migrate out to sea in November through April after several months of rearing in streams (Healey 1991). The adult Winter-Run Chinook Salmon upstream spawning migration in the Sacramento River occurs from December through July, with the majority of the run passing the Red Bluff Diversion Dam from January through May, peaking in mid-March (NMFS 2009; NMFS 2014). Adults prefer to hold in deep cold pools until they are sexually mature and ready to spawn during spring or summer. Winter-Run Chinook Salmon spawn primarily between mid-April and mid-August, with peak spawning generally occurring in June (Vogel and Marine 1991).

Spawning occurs in gravel substrate in relatively fast-moving, moderately shallow riffles or along banks with relatively high water velocities to promote higher oxygen levels and eliminate fines in the substrate. Depending on ambient water temperature, embryos hatch within 40 to 60 days, and alevin (yolk-sac fry) remain in the gravel beds for an additional 4 to 6 weeks. As their yolk-sacs become depleted, fry begin to emerge from the gravel and start exogenous feeding typically in late July to early August and continuing through October (Fisher 1994). Emergence usually occurs in late July, but as early as mid-June through mid-October. Post-emergent fry inhabit calm, shallow waters with fine substrates and depend on fallen trees, undercut banks, and overhanging riparian vegetation for refuge (Healey 1991).

Winter-Run Chinook Salmon fry and juvenile emigration past the Red Bluff Diversion Dam occurs as early as mid-July and extends as late as the end of March during dry water years (Vogel and Marine 1991; NMFS 1997, both as cited in NMFS 2014), although primary migration ends in December (Poytress and Carillo 2010, 2011, 2012). A large pulse of juvenile Winter-Run Chinook Salmon have been observed to emigrate past Knights Landing and into the Delta during and shortly after the first large autumn storm event (del Rosario et al. 2013). They occur in the Delta as early as November through as late as April (SacPAS, see Figure 2.1-1). Ocean entry begins as early as November and continues through May (Fisher 1994; Myers et al. 1998, both as cited in NMFS 2014). Winter-Run Chinook Salmon then, for the most part, spend 3 years in the ocean before returning to the river as spawning adults.

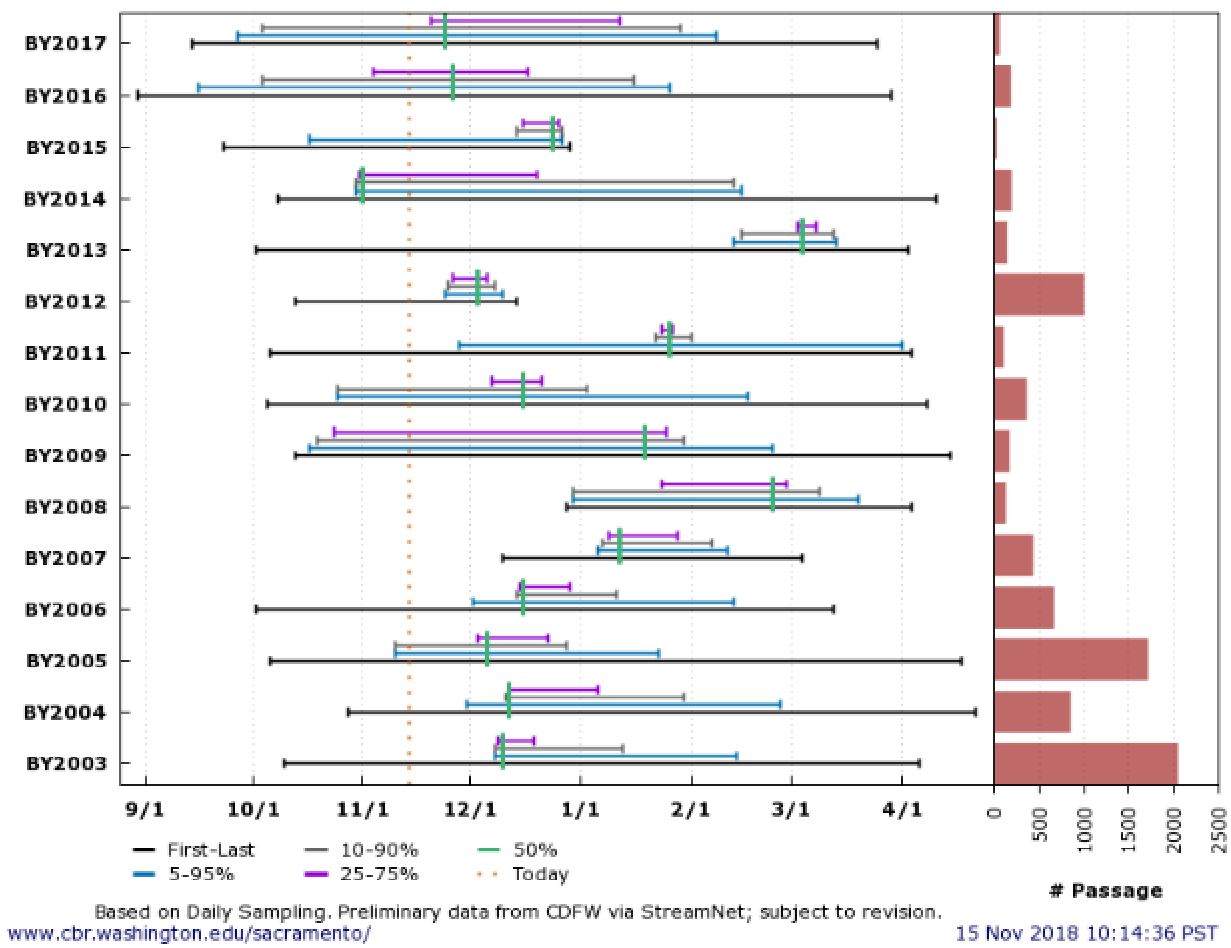

Figure 2.1-1. Migration Timing, Brood Years 2003–2017—Juvenile Winter Chinook, Knight's Landing RST, 7/1–6/30

During juvenile rearing and downstream movement, salmonids prefer stream margin habitats with sufficient depths and velocities to provide suitable cover and foraging opportunities. Ephemeral habitats, such as floodplains and the lower reaches of small streams, are also very important to rearing Chinook Salmon as these areas can be much more productive than the main channel and provide refuge from predatory fishes (Maslin et al. 1997; Sommer et al. 2001a). However, side channels with narrow inverts and nearshore areas with broad flat areas including low-gradient floodplains also can strand and isolate juveniles when high flows subside quickly (NMFS 1997). The greater availability of prey and favorable rearing conditions in floodplains increases juvenile growth rates compared with conditions in the mainstem and this can lead to improved survival rates during both their migration through the Delta and later in the marine environment (Sommer et al. 2001a). However, newer research has not found that the Yolo Bypass, a large floodplain, consistently provides better survival conditions for Chinook Salmon than the mainstem of the Sacramento River (Sommer et al. 2005; Takata et al. 2017).

2.1.3 Historical and Current Distribution and Abundance

Areas where Winter-Run Chinook Salmon historically spawned are now inaccessible due to Keswick and Shasta Dams. Streams in which populations of Winter-Run Chinook Salmon were known to historically exist were fed by cool, constant springs that provided the flows and low temperatures required for spawning, incubation, and rearing during the summer season (Slater 1963). Winter-Run Chinook Salmon spawning occurs in the summer months. Naturally occurring summer flows in river reaches below Keswick Dam would have historically precluded spawning. This suggests that the area below Shasta and Keswick dams was likely utilized for Winter-Run Chinook Salmon juvenile rearing and migration only. The life-history timing of the Winter-Run Chinook Salmon, requiring cold summer flows, indicates that the run historically occurred upstream of Keswick and Shasta dams and included the upper Sacramento River, McCloud River, Pit River, Fall River and Hat Creek and Battle Creek a tributary below Keswick and Shasta Dams (Yoshiyama et al. 1996, 2001; Lindley et al. 2004; NMFS 2014b), where summer flow and water temperature requirements were met (Yoshiyama et al. 2001).

Winter-Run Chinook Salmon are currently found in the mainstem Sacramento River downstream of Keswick dam. This population is maintained through cold water releases from Shasta Reservoir that create spawning and rearing habitat in the reach between Redding and the Red Bluff Diversion Dam. The construction of the Anderson-Cottonwood Irrigation District Diversion Dam in 1916 created a partial passage barrier as did the Red Bluff Diversion Dam in 1962. Since completion of Shasta Dam in 1945, primary spawning and rearing habitats have been confined to the cold water areas between Keswick Dam and the Red Bluff Diversion Dam (NMFS 2014).

Yearly Winter-Run escapement was estimated by counts in traps at the top of fish ladders at the Red Bluff Diversion Dam and more recently has been estimated using carcass counts. Escapements have declined from the 1960s and 1970s. The run size in 1969 was approximately 120,000, while run sizes averaged 600 fish from 1990 to 1997 (Moyle 2002). Escapement subsequently increased after Red Bluff Diversion Dam operations were modified and temperature control shutters were installed on Shasta Dam, but has declined since 2005 (Reclamation 2008; NMFS 2016). Winter-Run Chinook Salmon adult escapement data for the Sacramento River Basin from 1974 to 2016 are included in Figure 2.1-2 below (CDFW 2018). Preliminary data show a decline since 2012.

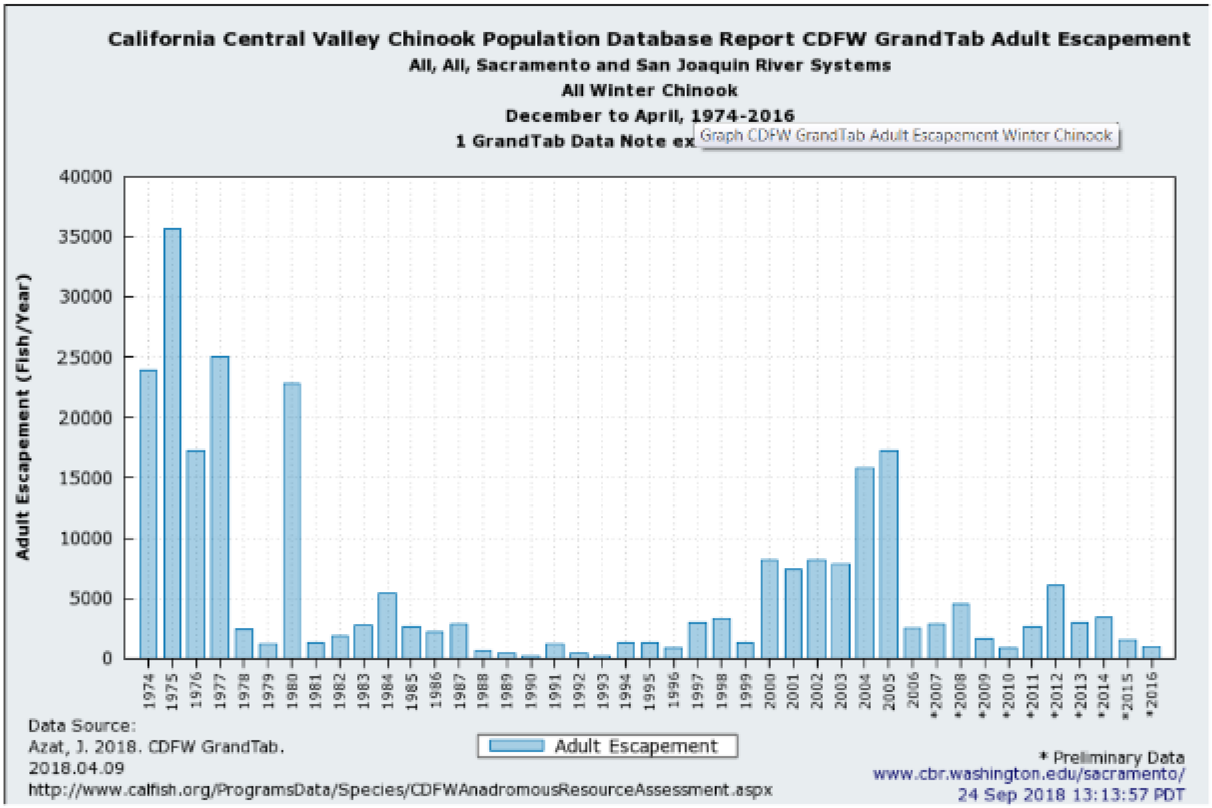

Figure 2.1-2. Winter-Run Chinook Salmon Adult Escapement (1974–2016) (CDFW 2018)

In addition to the Sacramento River, Juvenile Winter-Run Chinook Salmon have also been found to rear in areas including the lower American River, lower Feather River, Battle Creek, Mill Creek, Deer Creek, and the Delta (Phillis et al. 2018). Phillis et al. (2018) found with isotope data that 44 to 65 percent of surviving Winter-Run Chinook Salmon adults reared in non-natal habitats as juveniles. The lower reaches of the Sacramento River, the Delta, and San Francisco Bay serve as migration corridors for the downstream migration of juvenile and upstream migration of adult Winter-Run Chinook Salmon.

Until recent years, salmon passage was not allowed above the Coleman Hatchery barrier weir located on Battle Creek. No Winter-Run Chinook Salmon spawning has been observed in Battle Creek but Winter-Run Chinook Salmon were detected above the weir in 2006 (high flow year). All Winter-Run Chinook Salmon production currently occurs in the Sacramento River or Livingston Stone Fish Hatchery (CDFG 2004).

2.1.4 Limiting Factors, Threats, and Stressors

The major factor that limits the range of Winter-Run Chinook Salmon is the existence of dams, which have created barriers to upstream migration. Factors currently limiting abundance include the altered flow regime, which has led to changed water temperatures, reduced gravel mobilization, reduced riparian recruitment, etc; deteriorated habitat quality; entrainment in water diversions; predation pressure on juveniles; and loss of riparian and floodplain habitat. These factors are discussed below and in the "Past and Present Impacts" section of Chapter 3, Environmental Baseline, Biological Assessment.

Warm water releases from Shasta Dam have been a significant stressor to Winter-Run Chinook Salmon, especially when releases were warmer than usual because of the recent extended drought in California from 2012 through 2015 (NMFS 2016). The optimal water temperature for egg incubation ranges from 46 degrees Fahrenheit (°F) to 56°F (7.8 degrees Celsius (°C) to 13.3°C), and a significant reduction in egg viability occurs in mean daily water temperatures above 57.5°F (14.2°C) (Seymour 1956, Boles 1988, USFWS 1999, EPA 2003, Richter and Kolmes 2005, Geist et al. 2006). New temperature modeling show higher sensitivity to increases in water temperature due to exponential increases in oxygen demand with rise in temperature during the final weeks of egg-embryo maturation before the alevin stage (Martin et al. 2016, Anderson 2018). Despite Reclamation's best efforts to maintain appropriate spawning temperatures, there was increased mortality during the 2012-2015 drought. Warm water releases from Shasta Reservoir in 2014 and 2015 contributed to 5.9 percent and 4.2 percent egg-to-fry survival rates respectively, to the Red Bluff Diversion Dam.

Climate experts predict physical changes to ocean, river and stream environments along the West Coast that include warmer atmospheric temperatures, diminished snow pack resulting in altered stream flow volume and timing, lower late summer flows, a continued rise in stream temperatures, and increased sea-surface temperatures and ocean acidity resulting in altered marine and freshwater food-chain dynamics (Williams et al. 2016). Climate change and associated impacts on water temperature, hydrology, and ocean conditions are generally considered likely to have substantial effects on Chinook Salmon populations in the future (NMFS 2014). Global parameters, such as ocean conditions, have also demonstrated a marked effect on adult escapement (Lindley et al. 2009).

Impacts from hatchery fish (i.e., reduced fitness, weaker genetics, smaller size, less ability to avoid predators) have deleterious impacts on natural in-river populations (Matala et al. 2012). These impacts are associated with hatchery fish spawning naturally (i.e., second generational). During recent years, when the hatchery program was scaled up in size and natural production faltered, hatchery fish made up the vast majority of winter Chinook that spawned both in the river and at LSNFH. The Winter-Run Chinook Salmon conservation program at LSNFH is controlled by the USFWS to reduce such impacts. The average annual hatchery production at LSNFH is approximately 176,348 Winter-Run Chinook Salmon per year compared to the estimated natural production that passes the Red Bluff Diversion Dam, which is approximately 878,000 per year based on the 2012 to 2018 average (Voss et al. 2018), or 4.7 million per year based on the 2002 to 2010 average (Poytress and Carrillo 2011). Therefore, hatchery production can be up to approximately 20 percent of the total in-river juvenile production in any given year.

2.1.4.1 *Habitat Quality*

Construction of Keswick and Shasta Dam for agricultural, municipal, and industrial water supply has eliminated access to historical holding and spawning grounds above Keswick Dam, approximately 200 river miles (Yoshiyama et al. 1996). Rearing habitat quantity and quality has been reduced in the upper Sacramento River as a result of channel modification and levee construction (Lindley et al. 2009). Much of the historical floodplain habitat has been developed or converted, this has decreased shallow water habitat that has high residence time needed for food production (Jefferes et al. 2008; Katz et al. 2018; Ahearn et al. 2006).

More information on stressors of native fish, including physical, hydrologic, and biological alteration are described in the environmental baseline. Additional factors include other water quality parameters (e.g., dissolved oxygen), food quality and quantity, biotic interactions (e.g., predation and competition), altered hydrology in the Sacramento–San Joaquin Delta, loss of tidal marsh, commercial and/or recreational harvest, and predation from introduced species such as striped bass (NMFS 2014).

2.1.5 Water Operations Management

The Sacramento River system includes several major features and facilities that are relevant to temperature management: (1) Shasta Dam and Lake, and the installed Temperature Control Device (TCD); (2) interbasin transfers from the Trinity River Basin, which are conveyed through Whiskeytown Lake, the Clear Creek Tunnel and Carr Powerhouse, and the Spring Creek Tunnel; and (3) Keswick Reservoir, which regulates releases from Shasta Dam and Spring Creek Powerhouses, resulting in a stable flow regime for release from Keswick Dam.

Reclamation currently uses the Shasta TCD to improve temperatures while minimizing power loss. At Shasta Reservoir, Reclamation seeks to build cold water pool for Winter-Run Chinook Salmon spawning and incubation in the summer. Reclamation seeks to build storage through the fall, winter, and spring months. When higher releases from Shasta Dam are required in the fall through spring timeframe, this may reduce the summer cold water pool. Higher releases may be requested to avoid Winter-Run and Fall-Run Chinook Salmon redd dewatering, spring pulses for juvenile outmigration, or increased releases to meet Delta outflow or salinity requirements per D-1641. Usually, flows in the Sacramento River are kept high until the Winter-Run Chinook Salmon fry have emerged from the gravel. However, higher flows sometimes overlap the period in which Fall-Run Chinook Salmon begin to spawn, leading to Fall-Run Chinook Salmon spawning in shallow locations that may be out of water when Reclamation reduces flows for building storage. Once the Fall-Run Chinook Salmon begin spawning, fish agencies frequently want to maintain releases at the same level as where spawning occurred to avoid redd dewatering.

The Sacramento River Temperature Task Group (SRTTG) is a multiagency group, formed pursuant to SWRCB Water Rights Orders 90-5 and 91-1, to assist with increasing and stabilizing Chinook populations in the Sacramento River. Annually, Reclamation develops operation plans for controlling temperatures within the Shasta and Trinity divisions of the CVP. These plans consider impacts on Winter-Run and other races of Chinook Salmon, and associated project operations. Meetings are held initially in the spring to discuss biological, hydrologic, and operational information, objectives, and alternative operations plans for temperature control. Once an operation plan for temperature control is recommended, Reclamation submits the report to the SWRCB, generally on or before June 1 each year. The SRTTG may continue to meet throughout the year or other groups may be formed to discuss temperature management.

Fish agencies generally seek to maintain a 56 degree compliance location as far downstream as possible for as long as possible. Maintaining cold water too far downstream risks prematurely using up the cold water pool and results in warmer-than-desired temperatures at the end of the temperature control season. Fish agencies have further requested that Reclamation operate to experimental water temperature control regimes. The primary examples are the 7 Daily Average Daily Maximum (DADM) () and the 53.5°F Daily Average Temperature (DAT) at the Clear Creek gage on the Sacramento River (CCR). The requested temperature control regimes may deplete cold water faster than the objective of 56°F at Balls Ferry accorning to D-90-5 and pose substantial operational challenges.

Reclamation and DWR coordinate regarding downstream requirements (Delta outflow, Delta salinity, turbidity, etc) under D-1641 via the COA. Reclamation and DWR split requirements between the CVP and SWP. After splitting requirements with the SWP, Reclamation plans how to the meet the CVP share of the requirements via a combination of releases from Folsom Reservoir, releases from Shasta Reservoir, and/or reducing exports. The amount of water from each reservoir depends upon reservoir storage, channel capacity, fishery concerns, projected inflows, and projected end-of-September storage. Reclamation balances releases so that no one reservoir bears the full burden of meeting downstream requirements.

Congress authorized Shasta and Trinity Dams to work in an integrated fashion for “the principal purpose of increasing the supply of water available for irrigation and other beneficial uses” in the Central Valley. 69 Stat. 719. Exports from Trinity Reservoir helped decrease the demand on Shasta Reservoir for water supply and brought colder water directly into the Sacramento River system, preserving a larger cold water pool volume in Shasta Reservoir. Reclamation heavily relies on both reservoirs to meet multiple obligations for listed fish species in both basins. First, there are limitations on transbasin diversions from the Trinity Basin due to requirements in the Trinity River. The 2000 Trinity River Record of Decision (Trinity ROD) strictly limits Reclamation’s transbasin diversions to 55 percent of annual inflow on a 10-year average basis for the restoration and protection of the Trinity fishery, which restricts the amount of water authorized for exportation to the Central Valley. Pursuant to the Trinity ROD, the Trinity Reservoir now also provides flows for the Trinity River Restoration Program to improve conditions for the native fisheries on the Trinity River.

During the extraordinary conditions in 2014 and 2015, under extreme drought, as part of a coordinated response to improve Shasta Reservoir cold water pool management, a number of measures were taken on a temporary basis that included: (1) work with the State Board and water users to lower Wilkins Slough navigational flow requirements; (2) request that the State Board relax D-1641 Delta water quality requirements; (3) delay Sacramento River Settlement Contractor depletions, and transfer a volume of their water in the fall rather than increase depletions throughout the summer; (4) target slightly warmer temperatures during the Winter-Run Chinook Salmon holding period (before spawning occurs); and (5) install temporary improvements on the Shasta Dam TCD curtain (in 2015).

In 2017, Reclamation agreed to a long-term plan to provide fall augmentation flows for the Lower Klamath River. For the previous 15 years, and now under the 2017 long-term plan through 2030, Reclamation has released fall augmentation flows to help support fish health (this practice began following controversy and litigation over a large fish die off event that occurred in 2002 due to low flows).

2.1.6 Recovery and Management Actions

The following sections are actions that have been taken to benefit Winter-Run Chinook Salmon.

2.1.6.1 *Anadromous Fish Restoration Program*

Reclamation annually expends funding for the CVPIA Anadromous Fish Restoration Program (AFRP), CVPIA 3406(b)(1), to undertake reasonable measures to not less than double anadromous fish populations from the 1967-1991 time period and to mitigate other adverse environmental effects. Winter-Run actions are described in the Final Plan for the AFRP (2001).

2.1.6.2 *Anadromous Fish Screen Program*

Section 3406(b)(21) of the CVPIA authorized the Anadromous Fish Screen Program (AFSP) to assist the State of California on unscreened diversions. The AFRP screens or installs “fish protective devices” on diversions. The AFSP has developed guidelines to prioritize screening projects. Factors taken into account include location of the diversion in relation to areas used by anadromous fish for spawning and rearing, size of the diversion (or percent flow diverted in tributaries), season of diversion in relation to anadromous fish use of the stream or reach, and placement of the diversion. All but one of the diversions greater than 100 cubic feet per second (cfs) on the Sacramento River have fish screens.

2.1.6.3 *Anderson Cottonwood Irrigation District*

The Anderson Cottonwood Irrigation District (ACID) operates a diversion dam across the Sacramento River located 3.2 miles downstream from Keswick Dam. The ACID Diversion Dam was improved in 2001 and 2015 with the addition of new fish ladders and fish screens around the diversion. Since upstream passage was improved a substantial shift in Winter-Run Chinook Salmon spawning has occurred. In recent years, more than half of the Winter-Run Chinook Salmon redds have typically been observed above the ACID diversion dam (Killam 2008).

2.1.6.4 *Battle Creek Restoration Program*

The Battle Creek Salmon and Steelhead Restoration Project has a long history that includes research by various organizations and collaboration among many resource agencies and public interest groups. In 1999, a cooperative effort among Reclamation, USFWS, NMFS, CDFW, and the Pacific Gas and Electric Company (PG&E) led to the signing of a Memorandum of Understanding (MOU). The Battle Creek Salmon and Steelhead Restoration Project includes modifications to facilities and adjustments to operations for anadromous fish, including Winter-Run Chinook Salmon. Construction is anticipated to be complete in 2023.

2.1.6.5 *Spawning and Rearing Habitat Restoration*

Reclamation expends annual funding for the CVPIA under Section 3406(b)(13) Spawning and Rearing Habitat Program. The CVPIA (b)(13) program partners with other federal, state, and local agencies, water users, and other stakeholders to develop and implement a continuing program for the purpose of restoring and replenishing, as needed, salmonid spawning gravel lost due to the construction and operation of Central Valley Project dams and other actions that have reduced the availability of spawning gravel and rearing habitat in the Sacramento River.

The upper Sacramento River between Keswick Dam and the Red Bluff Diversion Dam presents several opportunities for improving and restoring salmonid spawning and rearing habitats. Reclamation annually injects spawning gravel into reaches of the Sacramento River where the majority of Winter-Run Chinook Salmon spawn.

2.1.6.6 *Glenn Colusa Irrigation District Hamilton City Fish Screen*

Glenn Colusa Irrigation District (GCID) diverts a maximum of 3,000 cfs from the Sacramento River at the Hamilton City pump station. The peak demand occurs in the spring, often at the same time as the peak outmigration of juvenile salmon. Because GCID diverts up to 25 percent of the Sacramento River flow at Hamilton City, GCID pumping operations were identified as a significant impediment to the downstream juvenile salmon migration. In 2000, GCID and Reclamation completed a 620-foot-long fish screen extension and channel improvements to minimize entrainment of salmonids into GCID’s facility.

2.1.6.7 *Livingston Stone National Fish Hatchery*

The USFWS manages a conservation hatchery program for Winter-Run Chinook Salmon at the LSNFH. This hatchery program supplements the natural population according to strict guidelines developed in conjunction with NMFS. Based on a review of available genetic and other information, this hatchery stock was considered part of the Sacramento River Winter-Run Chinook Salmon ESU in 2005 (70 FR 37160).

2.1.6.8 *Red Bluff Diversion Dam*

The Red Bluff Diversion Dam was decommissioned in 2013 providing unimpaired juvenile and adult fish passage, so that adult Winter-Run Chinook Salmon could migrate through the structure at a broader range of flows reaching spawning habitat upstream of that structure. This project was authorized by CVPIA 3406(b)(10).

2.1.6.9 *Salmon Resiliency Strategy*

The Sacramento Valley Salmon Resiliency Strategy, published in June 2017 by the State of California, is an approach to improving species viability and resiliency by implementing specific habitat restoration actions. Actions include: restoration on Battle Creek, Implement McCloud Reintroduction Pilot Plan, Provide Instream Flows to Support Chinook Salmon and Steelhead in Mill, Deer, Antelope, and Butte Creeks, Restore Fish Passage and Habitat in Upper Sacramento Tributaries, Restoration of Instream Habitats in Upper Sacramento River, Improve Fish Passage by Removing Sunset Pumps Rock Dam on the Feather River, Restore Off-Channel Rearing, Streambank, and Riparian Habitats and Migratory Conditions along Upper/Middle/Lower Reaches of the Sacramento River, Complete Fish Screen Construction on Major Diversions along the Sacramento River, Improve Sutter Bypass and Associated Infrastructure to Facilitate Adult fish Passage and Improved Stream Flow Monitoring, Improve Yolo Bypass Adult Fish Passage, Increase Juvenile Salmonid Access to Yolo Bypass, and Increase Duration and Frequency of Yolo Bypass Floodplain Inundation, Construct Permanent Georgiana Slough Non-Physical Barrier, Restore Tidal Habitat in the Delta, and other actions.

2.1.6.10 *Shasta Temperature Control Device*

Reclamation constructed the Shasta Temperature Control Device (TCD) under the CVPIA 3406(b)(6). Reclamation operates the Shasta TCD to conserve the available cold pool in the reservoir for spawning and egg incubation temperatures for Winter-Run Chinook Salmon without hydropower bypass. Reclamation manages releases to maintain suitable depths over Winter-Run Chinook Salmon redds to avoid dewatering when possible.

2.1.6.11 *Whiskeytown Reservoir Spring Creek and Oak Bottom Temperature Curtains*

Reclamation has replaced both the Spring-Creek and Oak Bottom temperature curtains in Whiskeytown Reservoir to improve temperature flexibility and build cold water pool temperature compliance for Clear Creek and Sacramento River.

2.1.6.12 *Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project*

Most salmonid floodplain rearing habitat in the Sacramento Valley was altered or blocked from use by dams and levees. The Yolo Bypass is the largest remaining floodplain in the Sacramento Valley, but is only accessible when the Sacramento River exceeds the crest of the Fremont Weir during high flow events. The Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project is a joint effort undertaken by DWR and Reclamation. The project largely focuses on infrastructure modifications to increase the number of juvenile salmonids that have access to floodplain habitat in the Yolo Bypass through Fremont Weir; and, to increase the ability of adult salmon and sturgeon to migrate from the Yolo Bypass to the Sacramento River.

2.1.6.13 *California EcoRestore*

California EcoRestore is an initiative to help coordinate and advance at least 30,000 acres of critical habitat restoration in the Sacramento–San Joaquin Delta (Delta) by 2020. The program includes a broad range of habitat restoration projects, including aquatic, sub-tidal, tidal, riparian, flood plain, and upland ecosystem.

Projects completed to date include the following:

- Fremont Weir Fish Passage Modification—This project widened and deepened the existing fish ladder at the Fremont Weir and the upstream and downstream adjoining channels were reconfigured to accommodate migratory fish passage. Existing earthen agricultural road crossings were replaced by permanent crossings that allow for the clear passage of migratory fish.
- Knights Landing Outfall Gates—A positive fish barrier, was constructed (with new concrete wing walls and installation of a metal picket weir) on the downstream side of the existing Knights Landing Outfall Gate in the Colusa Basin Drain. This project serves primarily as a fish passage improvement action that will prevent salmon entry into the Colusa Basin Drain where they become trapped with no access back to the Sacramento River.
- Wallace Weir—The project consisted of constructing a permanent earthen weir that was hardened to withstand winter floods to prevent adult salmon entry into the Colusa Basin Drain. A fish rescue facility was incorporated into the weir so fish that arrive at the Wallace Weir via the Yolo Bypass can be safely and effectively rescued and returned to the Sacramento River to resume their migration to upriver spawning grounds.
- Lindsey Slough—The project restored habitat function and connectivity to 159 acres of freshwater emergent wetlands and 69 acres of alkali wetlands, and recreated and reconnected a 1-mile tidal channel.
- Sherman Island—The project constructed levee setbacks in Mayberry Slough that will augment existing riparian vegetation and restore tidal wetland that will provide habitat for native species including salmonids.

2.1.6.14 *Battle Creek Winter-Run Chinook Salmon Reintroduction Plan*

The Battle Creek Winter-Run Chinook Salmon Reintroduction Plan is a key action in the NMFS Recovery Plan for the Evolutionarily Significant Units of Sacramento River Winter-Run Chinook Salmon. Reintroduction of Winter-Run Chinook Salmon into North Fork Battle Creek is part of a larger strategy in the NMFS Recovery Plan to restore some of the spatial structure of the ESU by reintroducing populations to habitats from which they have been extirpated.

2.1.6.15 *Flyway Farms Tidal Habitat Restoration Project*

This project has restored seasonal wetland and cattle grazing land to sub-tidal, intertidal and seasonal wetlands to benefit native fish species. The project involves restoring and enhancing approximately 300 acres of tidal freshwater wetlands, and an additional 30 acres of seasonal wetlands, at the southern end of the Yolo Bypass. It isdesigned to maximize residency time and foodweb production by capturing and slowly draining water through the excavation of two breaches along the Yolo Bypass Toe Drain and interior channels to connect and enhance existing wetlands on site. The goal is to improve habitat conditions for salmonids by providing rearing habitats for out-migrating juveniles and migratory habitats for adults.

2.1.6.16 *Shasta Dam Fish Passage Evaluation*

The Shasta Dam Fish Passage Evaluation was an effort to determine the feasibility of reintroducing Winter-Run and Spring-Run Chinook Salmon and Steelhead to tributaries above Shasta Dam. The evaluation was part of Reclamation's response to the June 4, 2009, NMFS *Biological Opinion (BO) and Conference Opinion on the Long-Term Operation of the Central Valley Project (CVP) and State Water Project (SWP)* (NMFS 2009).

2.1.6.17 *Harvest Management*

NMFS' current Winter-Run Chinook Salmon harvest management is set based on a 2012 RPA from the NMFS Winter-Run Chinook Salmon ocean harvest fishery consultation. During the consultation, the Pacific Fisheries Management Council expressed concern as initially no fishing was allowed below 500 forecasted Age 3 fish, and the rule did not account for drought. In response to these comments, NMFS proposed a new rule that continues to allow for harvest down to a forecasted population of 0 Age 3 Winter-Run Chinook Salmon (NMFS, March 2018).

Figure 2.1-3 shows the 2012 harvest control rule compared to the 2018 harvest control rule. The x-axis is the forecasted number of Age 3 Winter-Run Chinook Salmon. The y axis is the Impact Rate Cap, a metric of the ocean harvest.

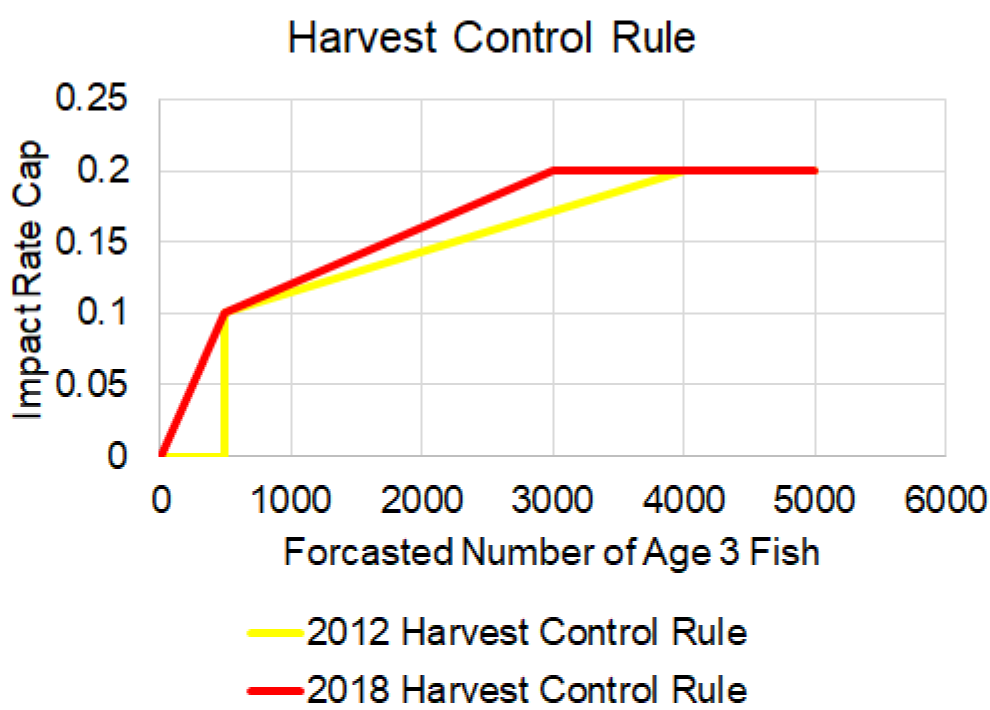

Figure 2.1-3. 2012 Harvest Control Rule

Under the 2018 rules, NMFS has increased harvest pressures on Winter-Run Chinook Salmon.

2.1.7 Monitoring and Research Programs

Monitoring and research programs help provide information on Winter-Run Chinook Salmon migration, survival, and redd distribution. Since 2015, Reclamation has started Enhanced Acoustic Tag Salmonid Monitoring (EATSM). EATSM is part of the Salmon and Sturgeon Assessment of Indicators by Lifestage (SAIL) program, which improves monitoring by addressing vital population statistics rather than reliance upon indexes. In 2018, EATSM conducted studies on hatchery-origin Winter-Run Chinook Salmon movement (Figure 2.1-4), which represents fish arrivals per day at Tower Bridge in downtown Sacramento (DOSS 2018). The two studies shown represent 20.7 percent survival (in red) and 26.9 percent survival (in teal).

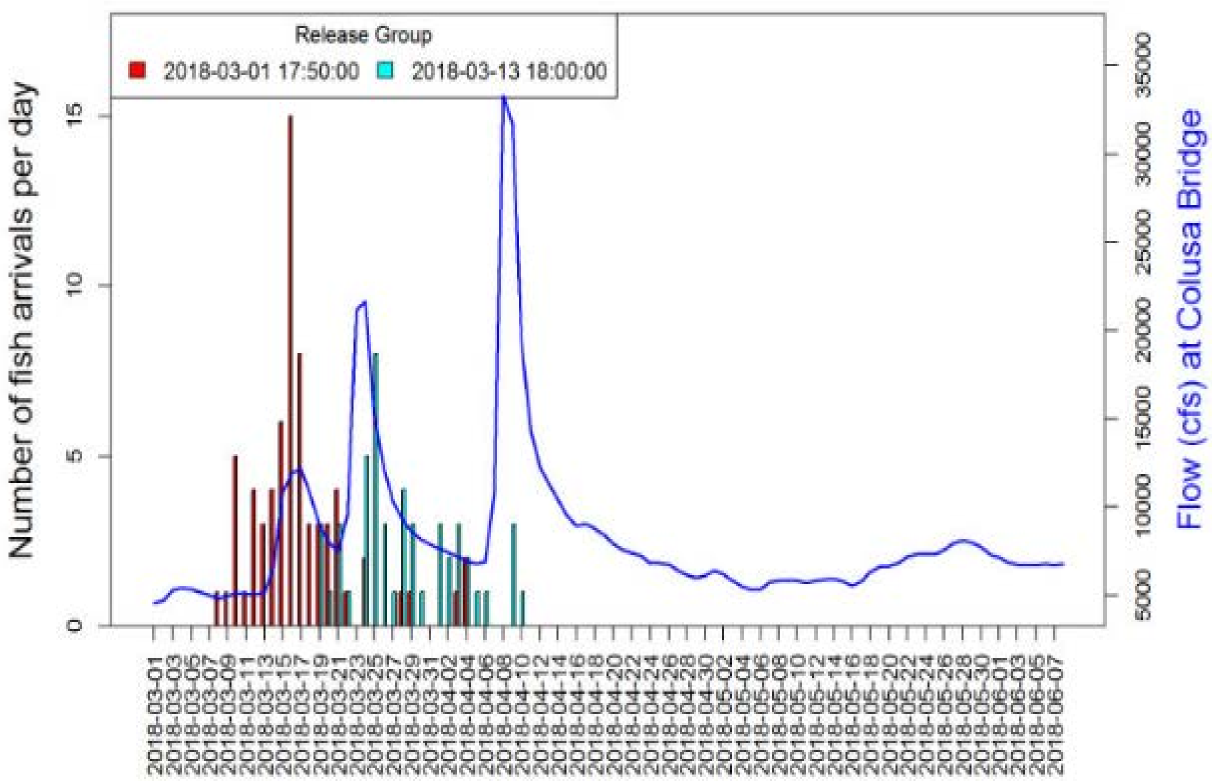

Figure 2.1-4. Tagged Hatchery Winter-Run Chinook Salmon Migration between Redding and Sacramento in 2018

CDFW annually conducts aerial redd surveys and carcass counts. Table 2.1-1 shows the distribution of Winter-Run Chinook Salmon redds from 2001 to 2018, a period after the ACID fish ladders were installed. For the period of 2001–2018, the furthest downstream observed Winter-Run Chinook Salmon redd was upstream of Tehama with over 98 percent of all observed redds occurring in the upper 20 river miles.

Surveys also help CDFW compile annual population estimates of Chinook Salmon. Information is entered into CDFW's GrandTab and is accessible through www.calfish.org. Reclamation funds monitoring, evaluation, and web-based data services through the Central Valley Prediction and

Assessment of Salmon (SacPAS) tool online. This service also provides a publicly accessible reporting system of historical and current information (www.cbr.washington.edu/sacramento/).

Table 2.1-1. Winter-Run Chinook Spawning Distribution, 2001–2018

Reach	Miles below Dam	2001–2018 Distribution	Yearly Average	Percent Distribution
Keswick to ACID Diversion Dam	3	3,482	226	45
ACID Diversion Dam to Highway 44 Bridge	5.5	2,606	154	34
Highway 44 Bridge to Airport Road Bridge	19	1,566	95	20
Airport Rd. Bridge to Balls Ferry Bridge	27	75	4	1
Balls Ferry Bridge to Battle Creek	32	10	1	0
Battle Creek to Jellys Ferry Bridge	36	7	0	0
Jellys Ferry Bridge to Bend Bridge	45	10	1	0
Bend Bridge to Red Bluff Diversion Dam	60	0	0	0
Red Bluff Diversion Dam to Tehama Bridge	74	11	1	0
Total		7,767	482	100

2.2 Winter-Run Chinook Salmon Critical Habitat

NMFS designated critical habitat for the Sacramento River Winter-Run Chinook Salmon ESU on June 16, 1993 (58 FR 33212). Designated critical habitat encompasses 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).

In the Sacramento River, critical habitat is the river water column, river bottom, and adjacent riparian zone and the water column and essential foraging habitat and food resources west of Chipps Island including the San Francisco Bay to the Golden Gate Bridge.

Critical habitat consists of physical and biological habitat features considered essential for the conservation of a species, which are referred to as Physical and Biological Features (PBFs). PBFs outlined in the designation of critical habitat (57 FR 36626) include the following:

1. Unimpeded access from the Pacific Ocean to appropriate spawning areas 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 57.5oF for successful spawning, egg incubation, and fry development;
5. Habitat and prey that is free of contaminants;
6. Riparian habitat that provides for successful juvenile development and survival; and
7. Unimpeded passage of juveniles downstream from the spawning grounds to San Francisco Bay and the Pacific Ocean.

2.3 Chinook Salmon, Central Valley Spring-Run ESU

2.3.1 ESA Listing Status

The Central Valley Spring-Run Chinook Salmon ESU was listed as threatened under the ESA in 1999 because of the reduced range and small size of remaining Spring-Run Chinook Salmon populations (64 FR 50393). On June 28, 2005, NMFS published the final hatchery listing policy (70 FR 37204) and reaffirmed the threatened status of the ESU (70 FR 37160). The ESU consists of naturally spawned Spring-Run Chinook Salmon originating from the Sacramento River and its tributaries, and also from the Feather River Fish Hatchery (FRFH) Spring-Run Chinook Program (NMFS 2016b).

Based on a review of the available information, NMFS (2016b) recommends that the Central Valley Spring-Run Chinook Salmon ESU remain classified as a threatened species. NMFS’ review also indicates that the biological status of the ESU has probably improved since the previous status review in 2010–2011 and that the ESU’s extinction risk may have decreased. However, the ESU is still facing substantial risks (Williams et al. 2016). Spring-Run Chinook Salmon escapement data for the Sacramento River Basin (CDFW 2018b) indicate that Spring-Run Chinook Salmon populations have steadily declined in abundance from 2014 through 2017 since peaking in 2013. As part of the 5-year review, NMFS also re-evaluated the status of the FRFH stock and concluded that it should remain part of the Central Valley Spring-Run Chinook Salmon ESU.

2.3.2 General Life-History and Habitat Requirements

Adult Spring-Run Chinook Salmon enter freshwater as immature fish between mid-February and July and remain in deep cold pools in proximity to spawning areas until they are sexually mature and ready to spawn in late summer and early fall, depending on water temperatures (CDFG 1998; NMFS 2009).

Spawning occurs in gravel substrate in relatively fast-moving, moderately shallow riffles or along banks with relatively high water velocities to promote higher oxygen levels and eliminate fines in the substrate. Fry emerge from the gravel from November to March (Moyle 2002) and can have highly variable emigration timing based on various environmental factors (NMFS 2009). Post-emergent fry inhabit calm, shallow waters with fine substrates and depend on fallen trees, undercut banks, and overhanging riparian vegetation for refuge (Healey 1991).

Some juveniles begin emigrating soon after emergence from the gravel, whereas others over-summer and emigrate as yearlings with the onset of intense fall storms (CDFG 1998). The emigration period for Spring-Run Chinook Salmon can extend from November to early May, with up to 69 percent of the young-of-the-year fish outmigrating through the lower Sacramento River and Delta during this period (CDFG 1998 as cited in NMFS 2009). Peak movement of yearling Spring-Run Chinook Salmon in the Sacramento River at Knights Landing occurs in December and again in March and April for young-of-the-year juveniles (NMFS 2009).

During juvenile rearing and downstream movement, salmon prefer stream margin habitats with sufficient depths and velocities to provide suitable cover and foraging opportunities. As described for Winter-Run Chinook Salmon, off-channel areas and floodplains can provide important rearing habitat. The greater availability of prey and favorable rearing conditions in floodplains increases juvenile growth rates compared with conditions in the mainstem Sacramento River, which can lead to improved survival rates during both their migration through the Delta and later in the marine environment (Sommer et al. 2001a).

2.3.3 Historical and Current Distribution and Abundance

Spring-Run Chinook Salmon populations historically occupied the headwaters of all major river systems in the Central Valley up to any natural barrier (Yoshiyama et al. 1998; Reclamation 2008). The Sacramento River was used as a migratory corridor to spawning areas in upstream tributaries and headwater streams (CDFG 1998). The most complete historical record of Spring-Run Chinook migration timing and spawning is contained in reports to the U.S. Fish Commissioners of Baird Hatchery operations on the McCloud River (Stone 1893, 1895, 1896a, 1896b, 1896c, 1898; Williams 1893, 1894; Lambson 1899, 1900, 1901, 1902, 1904, all as cited in CDFG 1998). Spring-Run Chinook migration in the upper Sacramento River and tributaries extended from mid-March through the end of July with a peak in late May and early June. Baird Hatchery intercepted returning adults and spawned them from mid-August through late September. Peak spawning occurred during the first half of September. The average time between the end of Spring-Run spawning and the onset of Fall-Run spawning at Baird Hatchery was 32 days from 1888 through 1901.

Construction of the Shasta and Keswick Dams in 1945 and 1950, respectively, has blocked passage to areas of historic spawning habitat, limiting potential spawning habitat to areas downstream of the dams. The presence of dams on the Sacramento River has blocked upstream passage of Spring-Run Chinook Salmon to historically available spawning habitat and confined them to a much smaller area of the watershed. Current spawning is restricted to limited areas in mainstem reaches below the lowermost impassable dams and in a few select tributaries with reduced habitat availability. However, Spring-Run spawned and continue to spawn in rivers other than the Sacramento River. The Central Valley drainage as a whole is estimated to have supported annual runs of Spring-Run Chinook Salmon as large as 600,000 fish between the late 1880s and 1940s (CDFG 1998). Following construction of Shasta, Keswick, and Friant dams, annual runs were estimated to be no more than 26,000 fish in the 1950s and 1960s (CDFW GrandTab data; Yoshiyama et al. 1998). Before the construction of Friant Dam (completed in 1942), nearly 50,000 adults were counted in the San Joaquin River (Fry 1961). The San Joaquin populations were essentially extirpated by the 1940s, with only small remnants of the run that persisted through the 1950s in the Merced River (Hallock and Van Woert 1959; Yoshiyama et al. 1998).

The Central Valley Spring-Run Chinook Salmon ESU has displayed broad fluctuations in adult abundance. Estimates of Spring-Run Chinook Salmon in the Sacramento River and its tributaries (not including the lower Yuba and Feather rivers because CDFW's GrandTab does not distinguish between Fall-Run and Spring-Run Chinook Salmon in-river spawners, and not including the FRFH) have ranged from 1,105 in 2017 to 25,890 in 1982.

Since 1995, Spring-Run Chinook Salmon annual run size estimates typically have been dominated by Butte Creek returns. Of the three tributaries producing naturally spawned Spring-Run Chinook Salmon (Mill, Deer, and Butte creeks), Butte Creek has produced an average of two-thirds of the total production over the past 10 years (DWR and Reclamation 2017; CDFW 2018b). During recent years, Spring-Run Chinook Salmon escapement estimates (excluding in-river spawners in the Yuba and Feather rivers) have ranged from 23,696 in 2013 to 1,796 in 2017 throughout the tributaries to the Sacramento River surveyed (CDFW 2018b).

Spring-Run Chinook Salmon population estimates remain low. Spring-Run Chinook escapement was estimated to be 6,453 in 2016 and 1,105 in 2017 (Figure 2.3-1; Azat 2018). In addition, fish monitoring is conducted throughout the year at the Tracy Fish Collection Facility (TFCF) and the John E. Skinner Delta Fish Protective Facility (Skinner Fish Facility) (collectively referred to as the Delta fish facilities). During WY 2017, 26,551 wild juvenile Spring-Run and 963 hatchery Spring-Run were observed at the Delta fish facilities, and 9,487 wild juvenile Spring-Run and 1,010 hatchery Spring-Run were observed during WY 2018. Fish monitoring is also conducted at the Rock Slough Intake by the Contra Costa Water District (CCWD). No Spring-Run have been collected in CCWD's Fish Monitoring Program at the Rock Slough Intake since 2008.

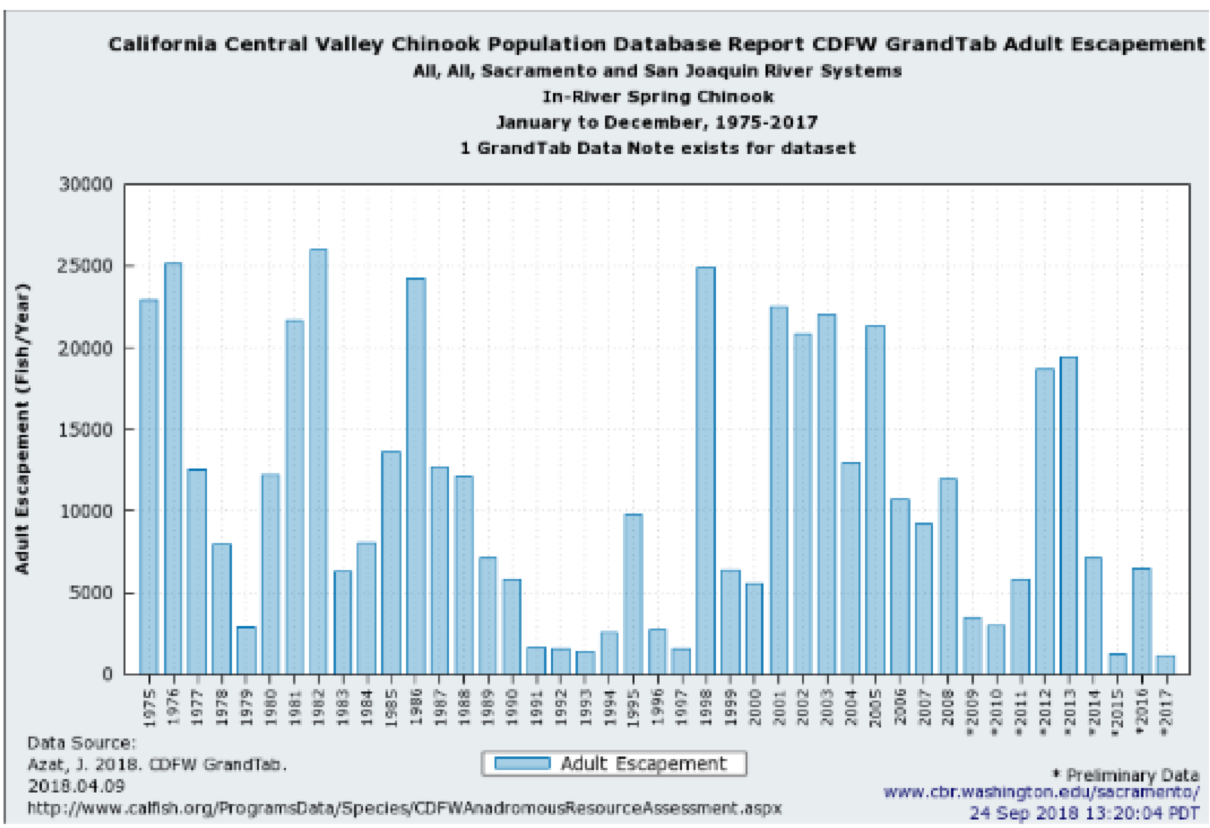

Figure 2.3-1. Estimates of Central Valley Spring-Run Chinook Salmon Escapement, 1975–2017

2.3.4 Limiting Factors, Threats, and Stressors

As discussed in the Winter-Run Chinook Salmon section and in Section 3.1, Past and Present Impacts, , the habitat that remains for Spring-Run Chinook Salmon has been negatively impacted by inadequate flows and increased water temperatures from dam and water diversion operations on streams throughout the Sacramento River Basin including on Deer, Mill, and Antelope Creeks. Losses of suitable spawning gravel, the development of deep channels and levees, pollutants and siltation from urban development, mining, and water diversions are also stressors on this ESU (NMFS 2009; 2014).

The degradation and simplification of aquatic habitat in the Central Valley has greatly reduced the resiliency of Spring-Run Chinook Salmon to respond to additional stressors such as an extended drought and poor ocean conditions. Levee construction and maintenance projects have greatly simplified riverine habitat and have disconnected rivers from the floodplain (NMFS 2016b).

Climate change poses a further threat to the species with increasingly high water temperatures and changes to ocean conditions. Spring-Run Chinook Salmon may be particularly vulnerable as adults over-summer in freshwater streams before spawning in autumn. The Central Valley Spring-Run Chinook Salmon spawn primarily in the tributaries to the Sacramento River, and those tributaries without cold water refugia will be more susceptible to impacts of climate change. Even in tributaries with cool water springs, in years of extended drought and warming water temperatures, unsuitable conditions may occur (NMFS 2016b). Juveniles often rear in their natal stream for one to two summers prior to emigrating, and would be susceptible to warming water temperatures.

2.3.5 Water Operations Management

Spring-Run requirements do not typically control the operation of Shasta, Oroville, Folsom, or New Melones Dams. On Clear Creek, Reclamation has a requirement from its 2002 water right as well as the 2000 Reclamation / USFWS /CDFW agreement to provide 50 cfs flow year-round, increasing to 70 cfs in November and December of critical years and increasing to 100 cfs in November and December of normal years. In addition to these flows, Reclamation makes releases as part of the CVPIA b(2) and (b)(12) program. Reclamation’s operations follow the CVPIA AFRP guidelines (USFWS 2001) which, for Clear Creek, are: “200 cfs October 1 to June 1 from Whiskeytown dam for Spring-Run, Fall-Run, and Late Fall-Run Salmon spawning, egg incubation, emigration, gravel restoration, spring flushing and channel maintenance; and release 150 cfs or less, from July through September to maintain less than 60°F temperatures in stream sections utilized by Spring-Run Chinook Salmon.” Additionally, the less water available for the transbasin diversion, the greater potential impact to Clear Creek temperatures as adequate temperatures in Clear Creek are dependent to a large degree on the volume of water moving through Lewiston and Whiskeytown reservoirs.

2.3.6 Recovery and Management

The NMFS 2014 Recovery Plan for Sacramento River Winter-Run Chinook Salmon, Central Valley Spring-Run Chinook Salmon, and Central Valley Steelhead outlines actions to restore habitat, access, and improve water quality and quantity conditions in the Sacramento River to promote the recovery of listed salmonids.

Under the CVPIA, Reclamation has funded the Service to undertake a number of actions to improve Spring-Run including, but not limited to, the restoration of Butte Creek and passage improvements to facilities on Mill Creek and Deer Creek. Spawning and rearing habitat improvements on the Upper Sacramento River also benefit Spring-Run. For more details concerning Spring-Run Chinook in Clear

Creek, see the 2017 Clear Creek Technical Team Annual Report for the Coordinated Long-Term Operation Biological Opinion.

2.3.6.1 *Clear Creek Restoration Program*

Reclamation annually expends funding for the CVPIA, Section 3406(b)(12) Clear Creek Restoration Program. The goals of the Clear Creek Restoration Program are to (1) provide flows to allow sufficient spawning, incubation, rearing, and outmigration for Salmon and Steelhead; (2) restore the stream channel and associated instream habitat; and (3) determine impacts of restoration actions on anadromous fish and geomorphology. The program manages flows and temperatures through releases from Whiskeytown Dam on a year-round basis to support the different life stages of Salmon and Steelhead in Clear Creek. The amounts of water, considering timing, magnitude, and duration, and water temperature are controlled to meet this goal. The Clear Creek Restoration Program is working on restoration of a 2-mile section of Clear Creek floodplain and stream channel degraded by aggregate and gold mining, dams and diversions, and annually injects gravel to recharge and maintain the system (approximately 8,000 to 10,000 tons of gravel per year). The Clear Creek Restoration Program aims to create and maintain 347,288 square feet of usable spawning habitat in Clear Creek.

2.3.6.2 *Ocean Management*

All of California's Chinook Salmon stocks are impacted to some extent by ocean fisheries (NMFS, 2000). As NMFS (2000) states, "the lack of an annual estimate of ocean harvest rate for the Central Valley fall chinook stocks targeted by ocean fisheries makes assessment of fishery impacts on listed stocks difficult. While the harvest rates on listed ESUs are believed to be less than that occurring on Central Valley fall chinook, the lack of a harvest rate estimate for even the targeted Central Valley stocks requires the Pacific Fisheries Management Council and NMFS to address recovery of weak stocks through "adaptive management" strategies, in which fishing effort is either eliminated or reduced by somewhat judgemental amounts and the effect is then assessed by monitoring spawning escapement in subsequent years." The 2000 BO on ocean harvest's effect (the Pacific Coast Salmon Plan) on Spring-Run Chinook Salmon concluded that continued harvest was not likely to jeopardize the continued existence of Central Valley Spring-Run Chinook Salmon (NMFS 2000).

To address the lack of an annual estimate of ocean harvest rate, one approach would be to estimate age-specific ocean fishing mortality rates by using cohort reconstructions applied to tagged Feather River Hatchery salmon (Satterthwaite et al. 2018). Harvest models that predict how Spring-Run would be affected by fishing regulations could be developed from reference harvest rates (Satterthwaite et al. 2018). Data and monitoring needs to better guide management of Central Valley Spring-Run Chinook including genetic sampling of juvenile emigrants to improve juvenile production data (Satterthwaite et al. 2018). Increased tagging and sampling of Spring-Run is needed to directly estimate ocean fishing mortality rates.

2.3.7 Monitoring and Research Programs

2.3.7.1 *San Joaquin River Restoration Program Experimental Population Management and Monitoring*

The San Joaquin River Restoration Program (SJRRP) has conducted and is in the process of conducting a large number of Spring-Run monitoring and research programs. The SJRRP has released a combination of FRFH and San Joaquin River Conservation and Research Facility (SCARF) Spring-Run Chinook Salmon juveniles to the San Joaquin River since 2014. All juvenile Spring-Run Chinook Salmon released are adipose fin-clipped and coded wire tagged. More information is available here: https://www.westcoast.fisheries.noaa.gov/central_valley/san_joaquin/san_joaquin_reint.html.

Because of previous release/reintroduction efforts, 2017 was the second year that adult Spring-Run Chinook Salmon had the potential to return to the San Joaquin River. However, due to an above average water year that prevented the placement of collection or counting stations, only limited monitoring occurred during the anticipated migration period. No unmarked Spring-Run Chinook (indicating wild origin) were seen in the lower reaches of the river.

UC Davis initiated a 2-year study in 2017 to calculate reach-specific survival and migration conditions for juvenile salmonids in the Lower San Joaquin River and south Delta. In March 2017, 700 individual SJRRP juveniles were tagged with acoustic JSATS tags and released in two evenly sized groups.

The SJRRP has established a parentage based tagging (PBT) program for the San Joaquin River Chinook Salmon populations. PBT involves the annual sampling and genotyping of adult Chinook Salmon returning to the Restoration Area; these data are being used to create a database of genotypes for future parentage assignment of their progeny. Genetic sampling of the San Joaquin River Fall-Run Chinook Salmon population began in 2013. As such, all adult Chinook Salmon returning to the Restoration Area in 2017/2018 were tissue sampled for genetic testing.

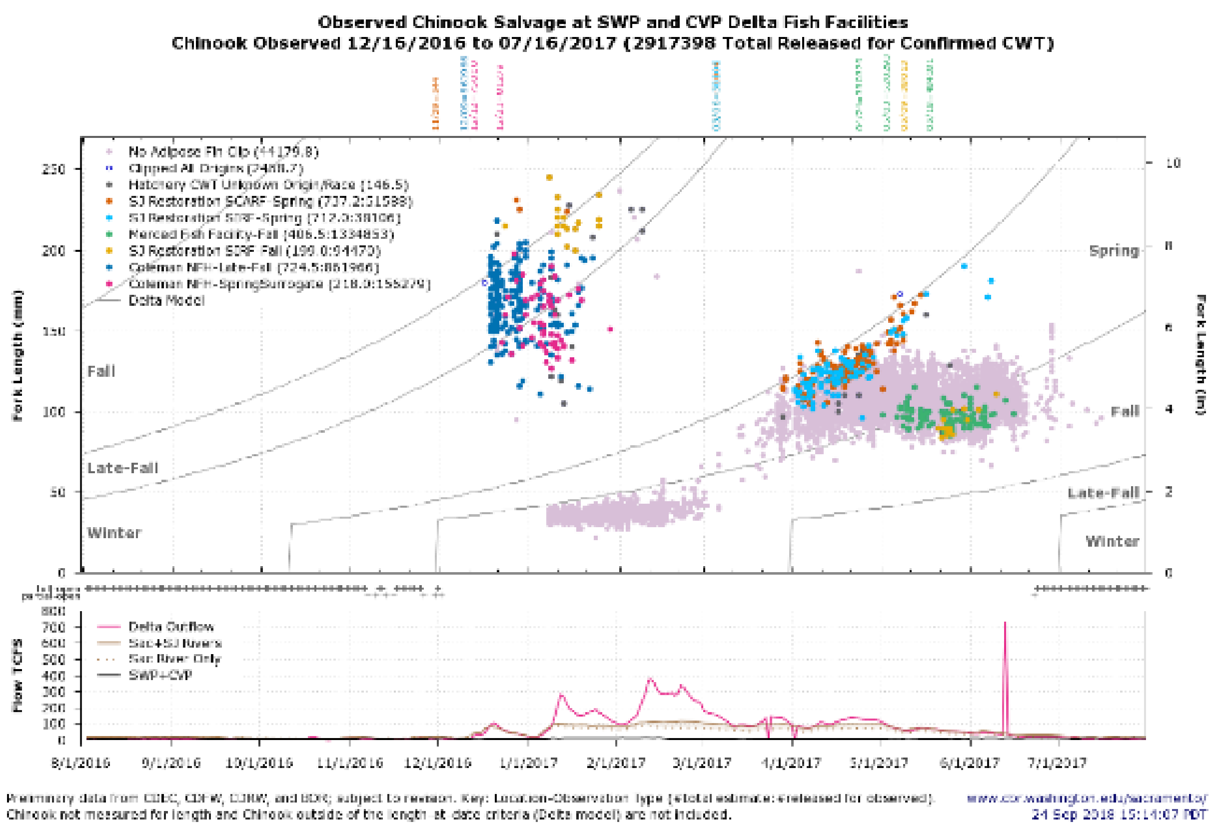

Figure 2.3-2. Observed Chinook Salvage at SWP and CVP Delta Fish Facilities during WY 2017

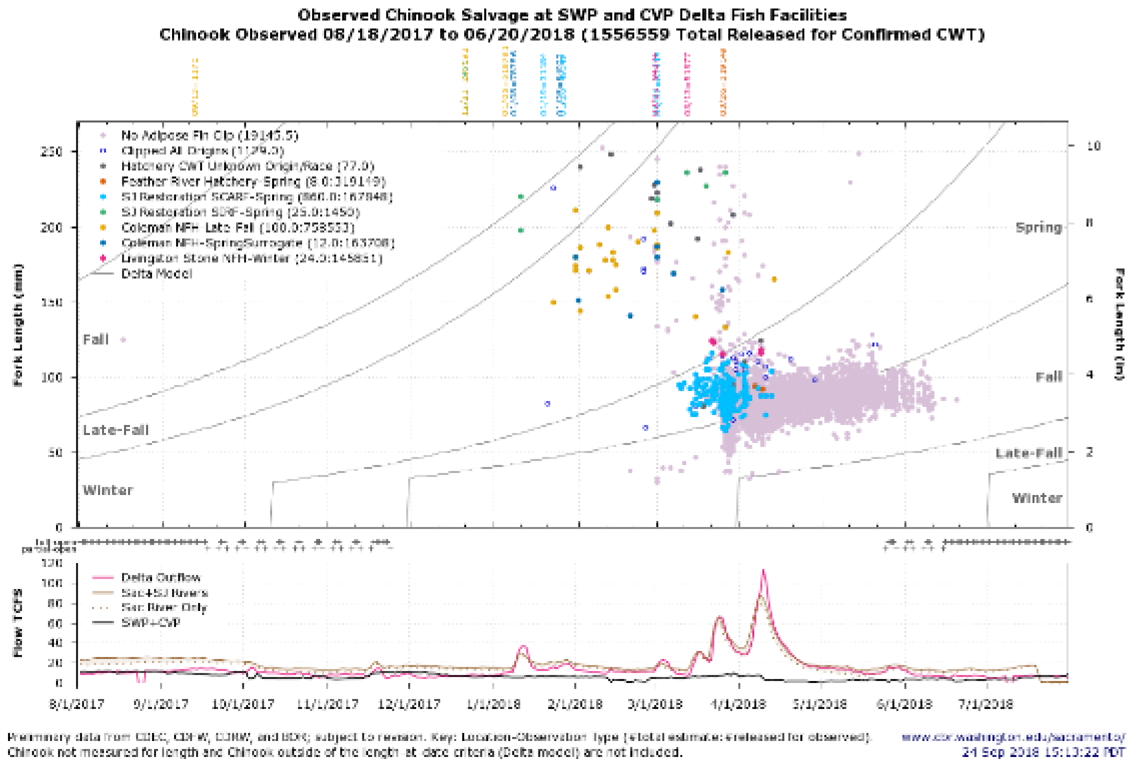

Figure 2.3-3. Observed Chinook Salvage at SWP and CVP Delta Fish Facilities during WY 2018

2.4 Spring-Run Chinook Critical Habitat

Critical habitat for the Central Valley Spring-Run Chinook Salmon was designated on September 2, 2005, and includes the mainstem Sacramento River from Chipps Island (RM 0) to Keswick Dam, and tributary reaches, including the Feather and Yuba rivers; Big Chico, Butte, Deer, Mill, Battle, Antelope, and Clear creeks; and portions of the northern Delta (70 FR 52488).

Physical and Biological Features (PBFs) essential for the conservation of listed Chinook Salmon ESUs are those sites and habitat components that support one or more life stages and include:

1. Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development.
2. Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
3. Freshwater migration corridors free of obstruction 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 with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; natural cover

such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

5. Nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels.
6. Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

2.5 Central Valley Fall-Run and Late Fall-Run Chinook Salmon, Evolutional Significant Unit

2.5.1 ESA Listing Status

The Fall-Run and Late Fall–Run Chinook Salmon includes all spawning populations of Fall-Run and Late Fall–Run Chinook Salmon in the Sacramento and San Joaquin River basins and their tributaries east of Carquinez Strait, California (64 FR 50394). After reviewing the best available scientific and commercial information, NMFS on September 16, 1999, determined that listing Central Valley Fall-Run and Late Fall–Run Chinook Salmon was not warranted. On April 15, 2004, the Central Valley Fall-Run and Late Fall–Run Chinook Salmon ESU was identified by NMFS as a Species of Concern (69 FR 19975).

Freshwater Essential Fish Habitat (EFH) for Pacific Salmon in the California Central Valley includes waters currently or historically accessible to salmon within the Central Valley ecosystem as described in Myers et al. (1998). EFH includes not only the watersheds of the Sacramento and San Joaquin River basins but also the San Joaquin Delta (Delta), Suisun Bay, and the Lower Sacramento River.

2.5.2 General Life-History and Habitat Requirements

Chinook Salmon have evolved a broad array of life history patterns that allow them to take advantage of diverse riverine conditions throughout the year. These life history patterns generally fall into two main generalized freshwater life history types: stream-type and ocean-type (Healey 1991). Ocean-type Chinook Salmon like Fall-Run and Late-Fall–Run enter freshwater in late summer and fall and spawn soon after, and juveniles typically migrate to the ocean as young-of-the-year after several months or rearing.

Adult Fall-Run Chinook Salmon migrate through the Delta and into Central Valley rivers from June through December. Individuals spawn in the Sacramento River and eggs and alevins are in the gravel primarily between September and January, with a peak during October through December. Most individuals (83.4 percent) spawn upstream of Red Bluff Diversion Dam, although, unlike other races of Chinook salmon, a moderate percentage (16.6 percent) spawn below Red Bluff Diversion Dam (Table 2.5-1).

Table 2.5-1. The Temporal Occurrence of Adult and Juvenile Fall-Run Chinook Salmon at Locations in the Central Valley

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult												
Delta[1]												
Sacramento River Basin[2]												
San Joaquin River[2]												
Juvenile												
Sacramento River at Red Bluff[3]												
Delta (beach seine)[4]												
Mossdale (trawl)[4]												
West Sacramento River (trawl)[4]												
Chipps Island (trawl)[4]												
Knights Landing (trap)[5]												

Relative Abundance: = High = Medium = Low

Note: Darker shades indicate months of greatest relative abundance.

Sources:

[1] State Water Project and Federal Water Project fish salvage data 1981–1988.

[2] Yoshiyama et al. 1998; Moyle 2002; Vogel and Marine 1991.

[3] Martin et al. 2001.

[4] U.S. Fish and Wildlife Service 2001b.

[5] Snider and Titus 2000.

Source: DWR and Reclamation 2016, p.11A–103

Table 2.5-2 shows the timing of the upstream presence of adult and juvenile life stages Late Fall-Run Chinook Salmon in the Sacramento River. The months included in this table represent the periods during which the majority (more than approximately 90 percent) of fish in a life stage are present. The life history characteristics of Late Fall–Run Chinook Salmon are not well understood. Late Fall–Run Chinook Salmon spawn in the Sacramento River and eggs and alevins are in the gravel primarily between December and June, with a peak during January through March. Most adults (83.4 percent) spawn upstream of Red Bluff Diversion Dam, and roughly two thirds (67.6 percent) spawn just below Keswick Dam in the reach to the ACID Dam (Table 2.5-2).

Table 2.5-2. The Temporal Occurrence of Adult and Juvenile Late Fall-Run Chinook Salmon at Locations in the Central Valley

Location	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult												
Delta[1]												
Sacramento River Basin[2]												
Juvenile												
Sacramento River at Red Bluff[3]												
West Sacramento River (trawl)[4]												
Delta (beach seine)[4]												
Chipps Island (trawl)[4]												
Knights Landing (trap)[5]												
Relative Abundance:		= High			= Medium				= Low			

Note: Darker shades indicate months of greatest relative abundance.

Sources:

[1] Moyle 2002.

[2] Yoshiyama et al. 1998; Moyle 2002; Vogel and Marine 1991.

[3] Martin et al. 2001.

[4] U.S. Fish and Wildlife Service 2001b.

[5] Snider and Titus 2000.

Source: DWR and Reclamation 2016, p.11A–104

In the Sacramento River, adult Fall-Run Chinook Salmon migrate upstream to spawn primarily during July through December, with a peak during August and September (Table 2.5-1). Adults that reach spawning grounds early in the season during July and August may hold before spawning (D. Swank, pers. comm.). Adult Late Fall-Run Chinook Salmon migrate upstream primarily during November through April (Table 2.5-2.).

Spawning occurs in gravel substrate in relatively fast-moving, moderately shallow riffles or along banks with relatively high water velocities to promote higher oxygen levels and eliminate fines in the substrate. Depending on ambient water temperature, embryos hatch in 40 to 60 days, and alevin (yolk-sac fry) remain in the gravel beds for an additional 4 to 6 weeks. As their yolk-sacs become depleted, fry begin to emerge from the gravel and start exogenous feeding. Fall-Run Chinook Salmon fry (i.e., juveniles shorter than 2 inches long) in the Sacramento River generally emerge from December through March, with peak emergence occurring by the end of January. In general, Fall-Run Chinook Salmon fry abundance in the Delta increases following high winter flows. Most Fall-Run Chinook Salmon fry rear in fresh water from December through June, with emigration occurring from December through June and a peak from January through March (Table 2.5-1). Smolts that arrive in the estuary after rearing upstream migrate quickly through the Delta and Suisun and San Pablo Bays. A very small number (generally less than 5 percent) of Fall-Run juveniles spend over a year in fresh water and emigrate as yearling smolts the following November through April.

Late Fall–Run Chinook Salmon fry generally emerge from March through June. Late Fall–Run fry rear upstream until about July (Table 2.5-2) and in fresh water from April through the following April and emigrate as smolts from November through May.

2.5.3 Historical and Current Distribution and Abundance

Central Valley Fall-Run Chinook Salmon historically spawned in all major tributaries, as well as the mainstem of the Sacramento and San Joaquin Rivers. The historical distribution of Central Valley Late Fall–Run Chinook Salmon is not well understood, but is thought to be less extensive than that of Fall-Run. Late Fall–Run adults most likely spawned in the Upper Sacramento and McCloud Rivers in reaches now blocked by Shasta Dam, as well as in major tributaries with adequate cold water in summer. There is also some evidence they once spawned in the San Joaquin River in the Friant region and in other large San Joaquin tributaries (Yoshiyama et al. 1998).

Currently Fall-Run Chinook spawn below rim dams and barriers to migration in the Sacramento and San Joaquin Rivers and their tributaries. Some smaller streams that lack unpassable barriers have runs that extend into historical Fall-Run habitat. Late Fall–Run currently spawn almost exclusively in the Upper Sacramento River from Keswick Dam to ACID Dam.

Abundance of Central Valley Fall-Run and Late Fall–Run Chinook Salmon escapement before 1952 is not well documented. Production estimates of Fall-Run and Late–Fall Run Chinook Salmon on the San Joaquin River historically approached 300,000 adults and probably averaged approximately 150,000 adults (Reynolds et al.1993.). Calkins et al. (1940) estimated Fall-Run and Late Fall–Run Chinook Salmon abundance at 55,595 adults in the Sacramento River basin from 1931 to 1939. Adult Fall-Run and LateFall–Run Chinook Salmon escapement in the early 1960s, was estimated to be 327,000 in the Sacramento River basin (California Department of Fish and Game 1965). Estimates of Fall-Run and Late Fall–Run Chinook Salmon escapement in the mid-1960s, to the San Joaquin River basin was about 2,400 fish (Reynolds et al. 1993). Sacramento and San Joaquin Rivers in river Fall-Run estimates of escapement from 1975 to 2017 (Figure 2.5-1.). Fall-Run Chinook Salmon of hatchery origin are included in the Pacific Coast Salmon Fishery Management Plan and are included in an EFH type analysis (Pacific Fishery Management Council 2014). Hatchery Fall-Run Chinook Salmon in the Sacramento and San Joaquin Rivers have ranged from over 700,000 in 2005 to just over 20,000 in 2009 (Figure 2.5-2.). Sacramento and San Joaquin River Fall-Run Chinook have been described as primarily a hatchery stock with a smaller natural component. The San Joaquin River Fall-Run Chinook Salmon population also has hatchery and natural components. Huber and Carlson (2015) provide a synthesis of trends in release number, location, size, and timing of Fall-Run Chinook Salmon released from the five Central Valley hatcheries between 1946 and 2012. They found since the mid-1980s the proportion of hatchery Fall-Run Chinook Salmon juveniles released downstream of the Delta has varied from around 20 to 60 percent.

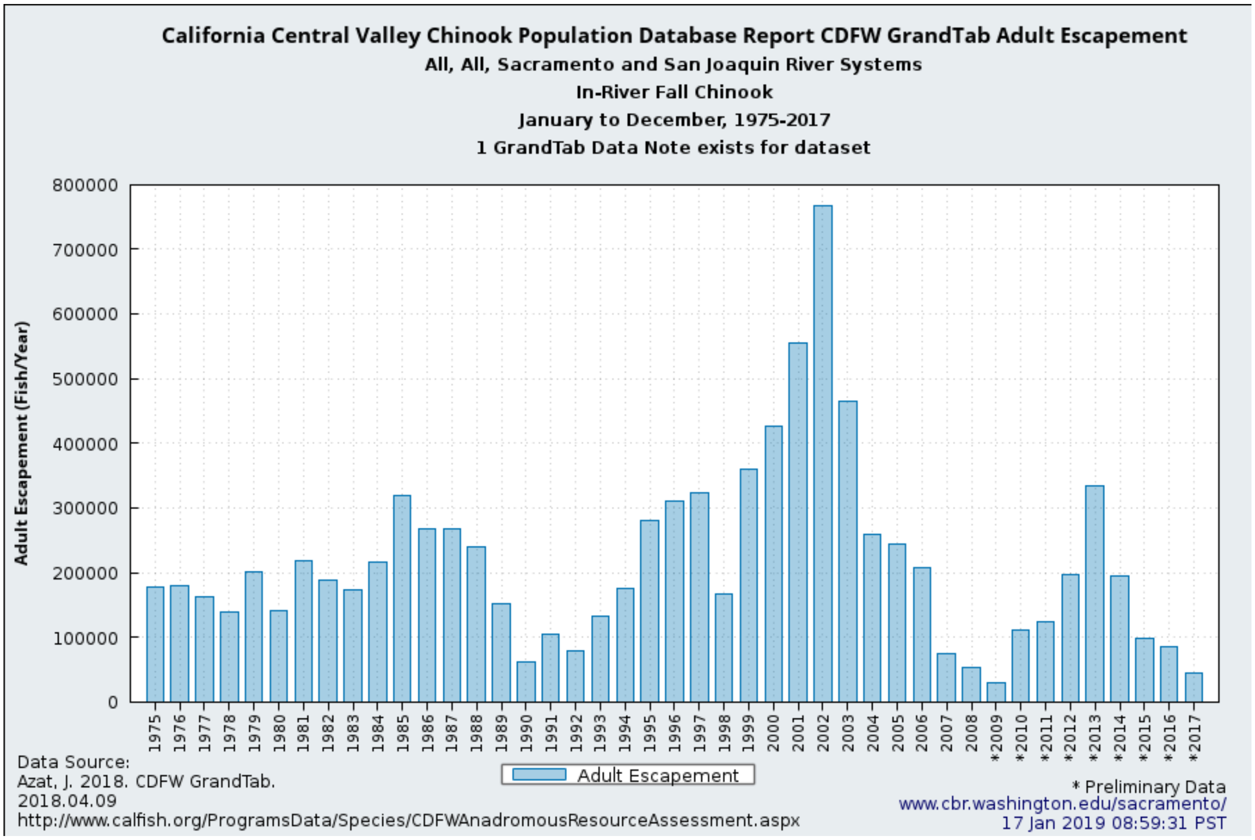

Figure 2.5-1. In-River Escapement Numbers of Fall-Run Chinook, Sacramento and San Joaquin River Systems

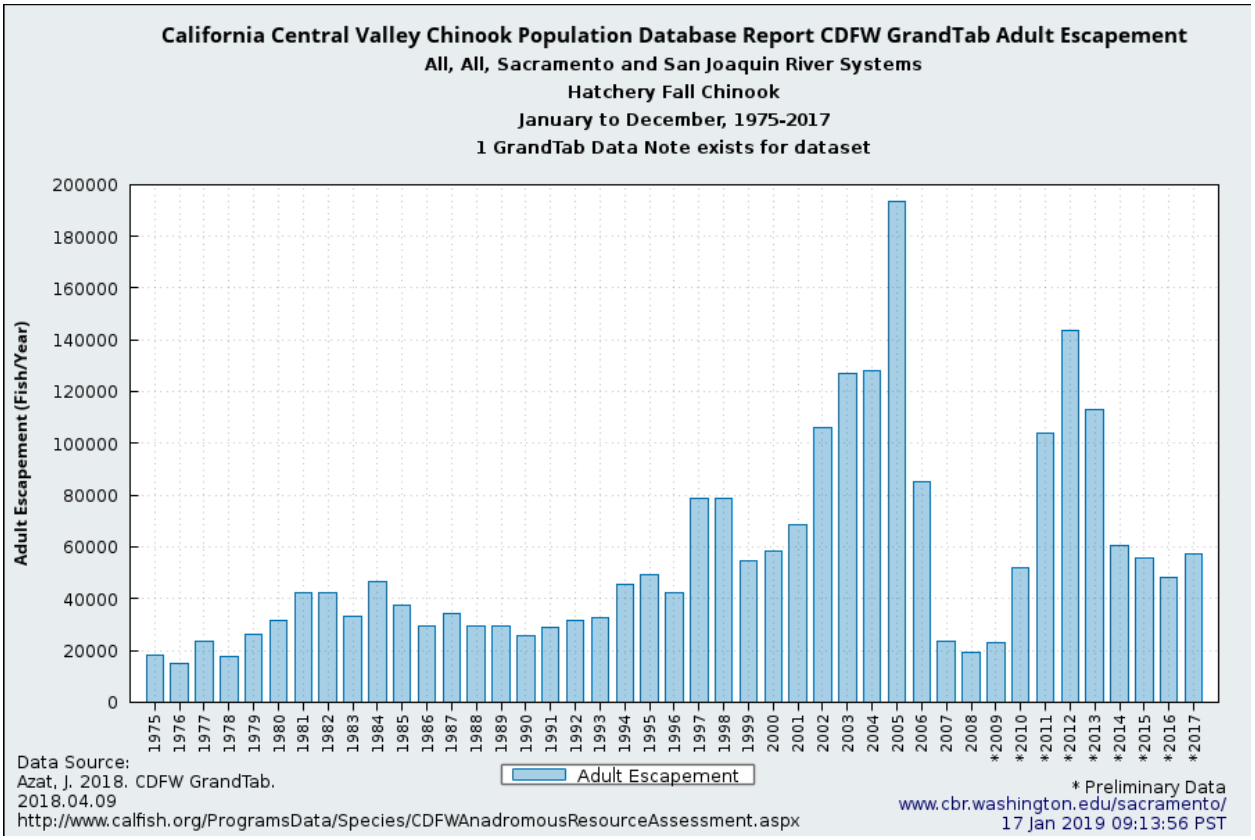

Figure 2.5-2. Hatchery Escapement Numbers of Fall-Run Chinook, Sacramento and San Joaquin River Systems

In the Sacramento and San Joaquin River from 1975 through 2017 adult escapement estimates for Late Fall–Run Chinook Salmon have ranged from several hundred adults to over 40,000 adults (Figure 2.5-3.). Between 1971 and 1997, adult escapement showed a general trend of declining abundance. From 1990 through 2006, escapement increased substantially, but was also highly variable from year to year. Escapement estimates were lower than the previous 4 years in 2008 and 2009, but not on the magnitude that was observed for Fall-Run Chinook Salmon (California Department of Fish and Wildlife 2016). Sacramento River Late Fall–Run Chinook Salmon stock has hatchery and natural components from the Upper Sacramento River basin (Figure 2.5-4).

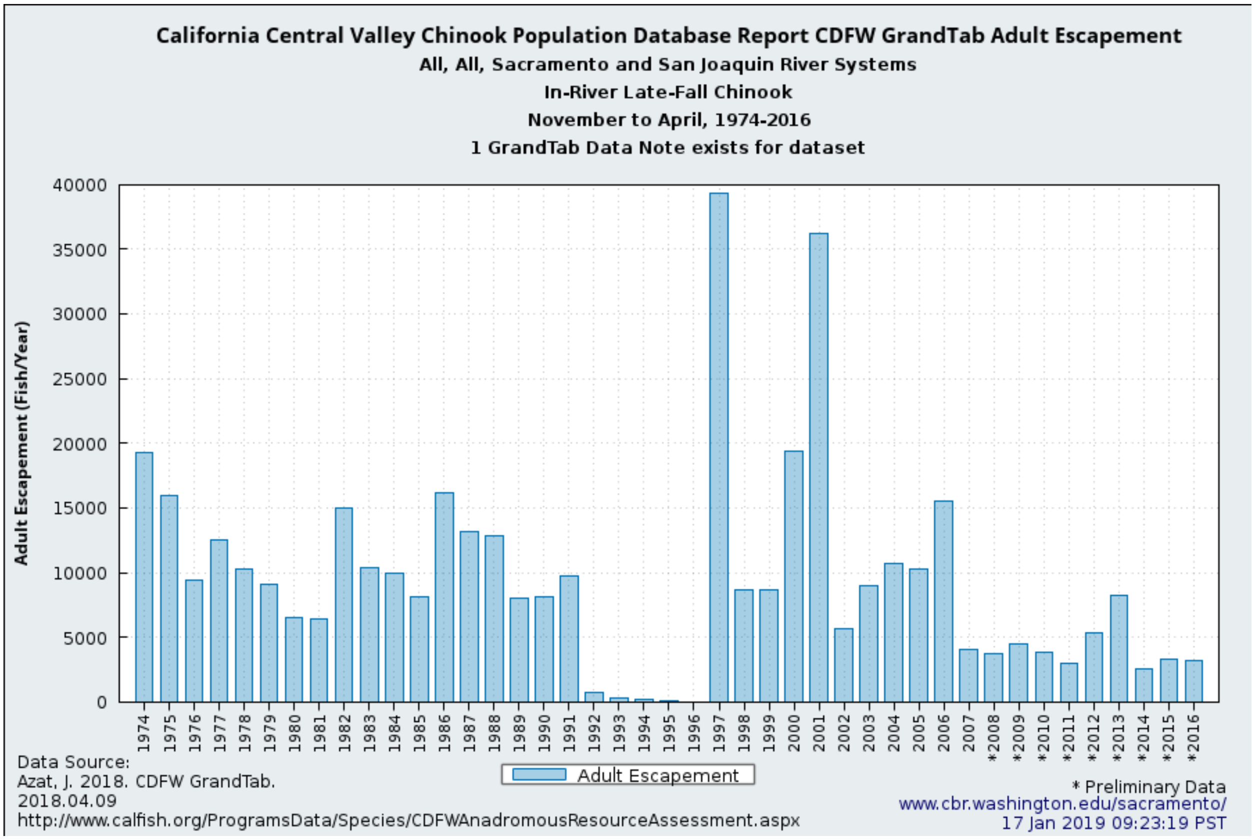

Figure 2.5-3. In-River Escapement Numbers of Late Fall-run Chinook, Sacramento and San Joaquin River Systems 1974–2017

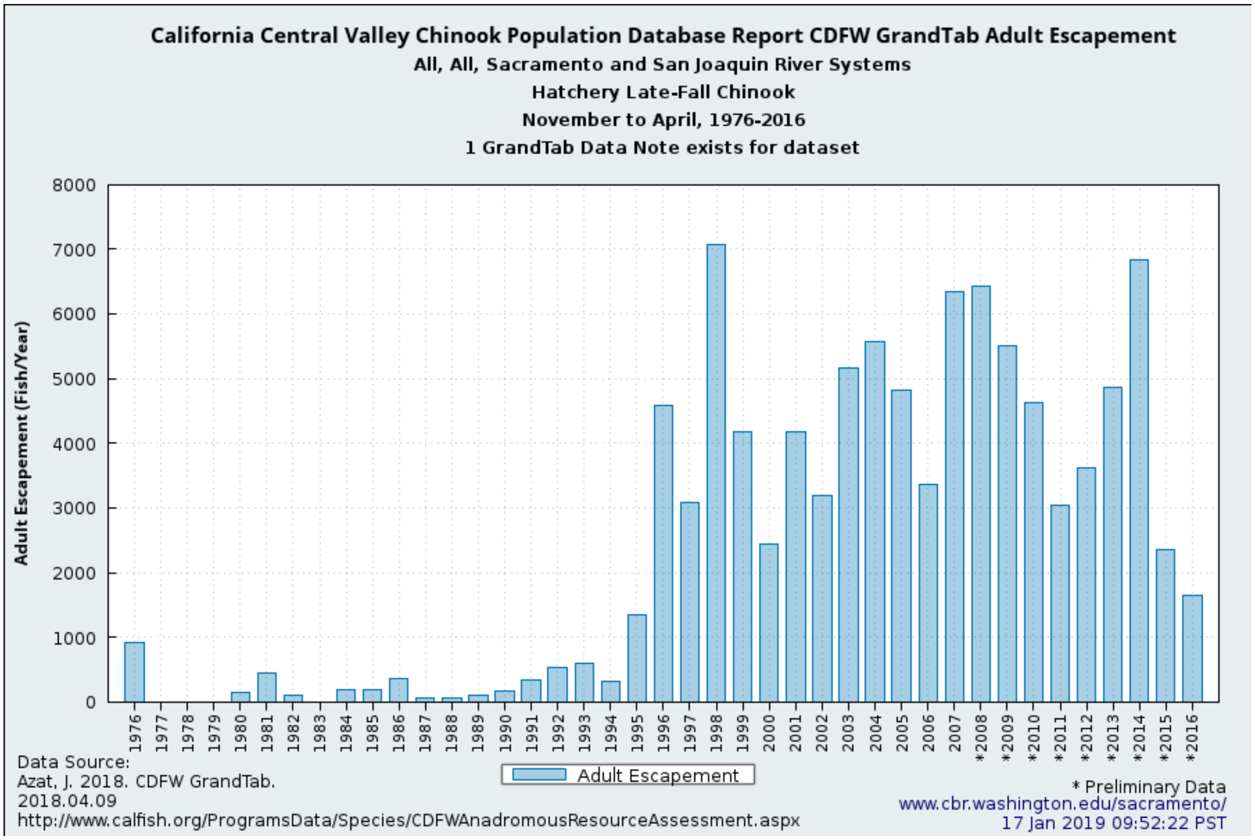

Figure 2.5-4. Hatchery Escapement Numbers of Late Fall-run Chinook, Sacramento and San Joaquin River Systems 1974–2017

2.5.4 Limiting Factors, Threats, and Stressors

The major factors that limit the range and abundance of Chinook Salmon are barriers to upstream migration, altered flow regime, high water temperature, habitat quality, entrainment in water diversions, loss of riparian and floodplain habitat, and ocean conditions.

Access to much or all of their historical spawning habitat was eliminated by high dams with no fish passage structures, although Fall-Run Chinook Salmon were less affected by these barriers than other Chinook races because much of their historical spawning habitat included the lower gradient reaches downstream of these dams (Reynolds et al. 1993; McEwan 2001). Changes in hydrologic patterns like the loss of spring peak flows and extended summer flows resulting from water and power operations have altered water temperatures and other habitat conditions for Fall-Run and Late Fall–Run Chinook (Williams 2006).

Migration and emigration corridors that previously contained high-value habitat types, such as dendritic channel systems, perched stream banks, floodplains and marshes, have been marginalized through channelization and leveed banks lined with riprap (Brandes and Mclain 2001). Natural flow regimes have been modified by upstream reservoirs that capture water during high flow events, thus, dampening the hydrograph and lowering the extent and duration of floodplain inundations and other off-channel, flow-dependent habitat used by emigrating juvenile Chinook salmon (70 FR 52488; Sommer et al. 2001; California Department of Water Resources 2005). Tidal and floodplain habitat areas provide important rearing habitat for foraging juvenile salmonids, including Fall-Run Chinook Salmon. Studies have shown that these salmonids may spend 2 to 3 months rearing in these habitat areas, and losses resulting from land reclamation and levee construction are considered to be major stressors on juvenile salmonids (Williams 2009). Similarly, channel margins provide valuable rearing and connectivity habitat along migration corridors, particularly for smaller juvenile fry, such as Fall-Run Chinook Salmon.

Predation on juvenile salmon by nonnative fish has been identified as an important threat to Fall-Run and Late Fall–Run Chinook Salmon in areas with high densities of nonnative fish that prey on outmigrating juvenile salmon (e.g., Smallmouth and Largemouth Bass, Striped Bass, and Catfish) (Lindley and Mohr 2003). Reduced habitat diversity (e.g., lack of cover) of channelized waterways in the rivers and Delta reduce refuge space for salmon from predators (Raleigh et al. 1984; Missildine et al. 2001; 70 FR 52488).

Climate experts predict physical changes to ocean, river, and stream environments along the West Coast that include warmer atmospheric temperatures, diminished snow pack resulting in altered stream flow volume and timing, lower late summer flows, a continued rise in stream temperatures, and increased sea-surface temperatures and ocean acidity resulting in altered marine and freshwater food-chain dynamics (Williams et al. 2016). Climate change and associated impacts on water temperature, hydrology, and ocean conditions are generally considered likely to have substantial effects on Chinook Salmon populations in the future (NMFS 2014). Global parameters, such as ocean conditions, have also demonstrated a marked effect on adult escapement (Lindley et al. 2009).

Impacts from hatchery fish (i.e., reduced fitness, weaker genetics, smaller size, less ability to avoid predators) have deleterious impacts on natural in-river populations (Hindar et al. 1991; Ryman et al. 1994; Waples 1994; McLean et al. 2005; Ford et al. 2006).

2.5.5 Recovery and Management

The following sections describe recovery and management actions that have been taken to benefit Fall-Run and Late Fall–Run Chinook Salmon.

2.5.5.1 *Anadromous Fish Restoration Program*

Reclamation annually expends funding for the CVPIA AFRP (CVPIA 3406(b)(1)) to undertake reasonable measures to not less than double anadromous fish populations of the 1967 to 1991 period and to mitigate other adverse environmental effects. Fall-Run and Late Fall–Run conservation actions are described in the final plan for the AFRP (2001).

2.5.5.2 *Anadromous Fish Screen Program*

Section 3406(b)(21) of the CVPIA authorized the AFSP to assist the State of California on unscreened diversions. The AFRP screens or installs “fish protective devices” on diversions. The AFSP has developed guidelines to prioritize screening projects. Factors taken into account are location of the diversion in relation to areas used by anadromous fish for spawning and rearing, size of the diversion (or percent flow diverted in tributaries), season of diversion in relation to anadromous fish use of the stream or reach, and placement of the diversion. All but one of the diversions greater than 100 cfs on the Sacramento River have fish screens.

2.5.5.3 *Anderson Cottonwood Irrigation District*

The ACID operates a diversion dam across the Sacramento River located 5 miles downstream from Keswick Dam. The ACID Diversion Dam was improved in 2001 and 2015 with the addition of new fish ladders and fish screens around the diversion. (Killam 2008).

2.5.5.4 *Battle Creek Restoration Program*

The Battle Creek Salmon and Steelhead Restoration Project has a long history that includes research by various organizations and collaboration among many resource agencies and public interest groups. In 1999, a cooperative effort among Reclamation, USFWS, NMFS, CDFW, and PG&E led to the signing of a MOU. The Battle Creek Salmon and Steelhead Restoration Project includes modifications to facilities and adjustments to operations for anadromous fish. Construction is anticipated to be complete in 2021.

2.5.5.5 *Spawning and Rearing Habitat Restoration*

Reclamation expends annual funding for the CVPIA under Section 3406(b)(13) Spawning and Rearing Habitat Program. Federal, state, and local agencies, water users, and other stakeholders have partnered to develop and implement a continuing program for the purpose of restoring and replenishing, as needed, salmonid spawning gravel lost due to the construction and operation of CVP dams and other actions that have reduced the availability of spawning gravel and rearing habitat in the Sacramento River.

The Upper Sacramento River between Keswick Dam and Red Bluff Diversion Dam presents several opportunities for improving and restoring salmonid spawning and rearing habitats.

2.5.5.6 *Glenn Colusa Irrigation District Hamilton City Fish Screen*

Glenn Colusa Irrigation District (GCID) diverts a maximum of 3,000 cfs from the Sacramento River at the Hamilton City pump station. The peak demand occurs in the spring, often at the same time as the peak outmigration of juvenile salmon. Because GCID diverts up to 25 percent of the Sacramento River flow at Hamilton City, GCID pumping operations were identified as a significant impediment to the downstream juvenile salmon migration. In 2000, GCID and Reclamation completed a 620-foot-long fish screen extension and channel improvements to minimize entrainment of salmonids into GCID’s facility.

2.5.5.7 *Red Bluff Diversion Dam*

The Red Bluff Diversion Dam was decommissioned in 2013, providing unimpaired juvenile and adult fish passage so that adult Chinook salmon could migrate through the structure at a broader range of flows and reach spawning habitat upstream of the structure. This project was authorized by CVPIA 3406(b)(10).

2.5.5.8 *Shasta Temperature Control Device*

Reclamation constructed the TCD in under the CVPIA 3406(b)(6). Reclamation operates the Shasta TCD to conserve the available cold pool in the reservoir for spawning and egg incubation temperatures for Chinook salmon without power bypass. Reclamation manages releases to maintain suitable depths over Chinook salmon redds to avoid dewatering.

2.5.5.9 *Whiskeytown Reservoir Spring Creek and Oak Bottom Temperature Curtains*

Reclamation has replaced both the Spring-Creek and Oak Bottom temperature curtains in Whiskeytown Reservoir to improve temperature flexibility and build a cold water pool for Clear Creek and Sacramento River temperature compliance.

2.5.5.10 *Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project*

Most salmonid floodplain rearing habitat in the Sacramento Valley was altered or blocked from use by dams and levees. The Yolo Bypass is the largest remaining floodplain in the Sacramento Valley, but is only accessible when the Sacramento River exceeds the crest of the Fremont Weir during high flow events. The Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project is a joint effort undertaken by DWR and Reclamation. The project largely focuses on infrastructure modifications to increase the number of juvenile salmonids that have access to floodplain habitat in the Yolo Bypass through Fremont Weir and to increase the ability of adult salmon and sturgeon to migrate from the Yolo Bypass to the Sacramento River.

2.5.5.11 *California Ecorestore*

California EcoRestore is an initiative to help coordinate and advance at least 30,000 acres of critical habitat restoration in the Delta by 2020. The program includes a broad range of habitat restoration projects, including aquatic, subtidal, tidal, riparian, flood plain, and upland ecosystems. Projects completed to date include include the following:

- Fremont Weir Fish Passage Modification—Fremont Weir Adult Fish Passage Modification Project widened and deepened the existing fish ladder at the Fremont Weir and the upstream and downstream adjoining channels were reconfigured to accommodate migratory fish passage. Existing earthen agricultural road crossings were replaced by permanent crossings that allow for the clear passage of migratory fish.
- Knights Landing Outfall Gates—A positive fish barrier was constructed (with new concrete wing walls and installation of a metal picket weir) on the downstream side of the existing Knights Landing Outfall Gate in the Colusa Basin Drain. This project serves primarily as a fish passage improvement action that will prevent salmon entry into the Colusa Basin Drain, where they become trapped with no access back to the Sacramento River.
- Wallace Weir—The project consisted of constructing a permanent earthen weir that was be hardened to withstand winter floods to prevent adult salmon entry into the Colusa Basin Drain. A fish rescue facility was incorporated into the weir so fish that arrive at the Wallace Weir via the Yolo Bypass can be safely and effectively rescued and returned to the Sacramento River to resume their migration to upriver spawning grounds.
- Lindsey Slough—The project restored habitat function and connectivity to 159 acres of freshwater emergent wetlands and 69 acres of alkali wetlands, and recreated and reconnected a 1-mile tidal channel.
- Sherman Island—The project constructed levee setbacks in Mayberry Slough that will augment existing riparian vegetation and restore tidal wetland that will provide habitat for native species including salmonids.
- Sacramento River Temperature Task Group—The SRTTG is a multiagency group formed pursuant to SWRCB Water Rights Orders 90-5 and 91-1 to assist with improving and stabilizing

Chinook population in the Sacramento River. Annually, Reclamation develops temperature operation plans for the Shasta and Trinity divisions of the CVP. These plans consider impacts on Winter-Run and other races of Chinook Salmon, and associated project operations. The SRTTG meets initially in the spring to discuss biological, hydrologic, and operational information, objectives, and alternative operations plans for temperature control.

2.5.5.12 *Flyway Farms Tidal Habitat Restoration Project*

Restored seasonal wetland and cattle grazing land to sub-tidal, intertidal and seasonal wetlands to benefit native fish species. Involves restoring and enhancing approximately 300 acres of tidal freshwater wetlands, and an additional 30 acres of seasonal wetlands, at the southern end of the Yolo Bypass. Designed to maximize residency time and food web production by capturing and slowly draining water through the excavation of two breaches along the Yolo Bypass Toe Drain and interior channels to connect and enhance existing wetlands on site. The goal is to improve habitat conditions for salmonids by providing rearing habitats for outmigrating juveniles and migratory habitats for adults.

2.5.5.13 *Shasta Dam Fish Passage Evaluation*

The Shasta Dam Fish Passage Evaluation was an effort to determine the feasibility of reintroducing Winter-run and Spring-Run Chinook Salmon and Steelhead to tributaries above Shasta Dam. The evaluation was part of Reclamation’s response to the June 4, 2009, NMFS BO *Biological Opinion (BO) and Conference Opinion on the Long-Term Operation of the Central Valley Project (CVP) and State Water Project (SWP)* (NMFS 2009).

2.5.6 Water Operations Management

2.5.6.1 *Sacramento River*

The Sacramento River system includes several major features and facilities that are relevant to temperature management: (1) Shasta Dam and Lake and the installed TCD; (2) interbasin transfers from the Trinity River Basin, which are conveyed through Whiskeytown Lake, the Clear Creek Tunnel and Carr Powerhouse, and the Spring Creek Tunnel; and (3) Keswick Reservoir, which regulates releases from Shasta Dam and Spring Creek Powerhouses, resulting in a stable flow regime for release from Keswick Dam.

Reclamation currently strives to meet Sacramento River storage and temperature requirements (NMFS RPA Actions I.2.1 through I.2.4), as well as holding the Sacramento River Temperature Task Group meetings, providing operations plans each year, and using the Shasta TCD to strive to meet temperature targets while minimizing power loss.

Measures taken in 2014 and 2015 as part of a coordinated drought response to improve Shasta Reservoir cold water pool management included: (1) working with the SWRCB and water users to lower Wilkins Slough navigational flow requirements; (2) requesting that the SWRCB relax D-1641 Delta water quality requirements; (3) delaying Sacramento River Settlement Contractor depletions and transfering a volume of their water in the fall rather than increase depletions throughout the summer; (4) targeting slightly warmer temperatures during the Sacramento River Winter-Run Chinook Salmon holding period (before spawning occurs); and (5) installing a Shasta Dam TCD curtain in 2015.

2.5.6.2 *Clear Creek*

Reclamation has a requirement from its 2002 water right as well as the 2000 Reclamation/USFWS/DFW agreement for 50 cfs year-round in all years, increasing to 70 cfs in November and December of critical years and increasing to 100 cfs in November and December of normal years.

Reclamation's operations on Clear Creek follow the CVPIA AFRP guidelines (USFWS 2001) of 200 cfs October 1 to June 1 from Whiskeytown dam for anadromous salmonids and their habitat. A flow of 150 cfs or less, from July through September to maintain 60ºF temperatures in stream sections utilized by Spring-Run Chinook Salmon.

2.5.6.3 *Stanislaus River*

Reclamation operates the Stanislaus River separately from the other Central Valley Project reservoirs. While releases from New Melones Reservoir provide inflow to the Delta, Reclamation does not operate New Melones for Delta salinity, outflow, or export requirements. Reclamation operates New Melones Reservoir to meet instream flow objectives (see 2009 BO, Table 2E flows), dissolved oxygen standards as measured at Ripon, salinity objectives at Vernalis (as set in SWRBC D-1641) and Vernalis flow objectives as set in D-1641 and updated in the 2009 BO.

Prior to the 2009 BO Table 2E flows, instream releases on the Stanislaus River were set pursuant to the 1987 CDFG agreement. This agreement was intended to only be in place for 10 years while a specific set of fishery studies was completed to help inform the decision on what instream flows would be most beneficial. However, while studies have been completed, the agreement has never been updated. Each year Reclamation determines the annual volume of water available to be utilized for fishery releases and transmits that volume to the CDFW. CDFW is then responsible for determining the pattern of water release. Since the initiation of this agreement, the CDFW has routinely put 0 cfs as the required release during the summer months. Reclamation has a separate obligation to meet dissolved oxygen standards at Ripon during the summer months as required by Reclamation's water rights. DFW has always assumed that Reclamation will have to release approximately 300 cfs to meet the dissolved oxygen standards. This allowed CDFW to concentrate their fishery volume in other months. However, this was not the intent of the agreement and had the effect of stressing the reservoir resources, particularly in dry years. Since the BO, this has generally not been a problem because the 2009 NMFS BO Table 2E flows have generally been greater than those requested by CDFW. However, in some years or months, such as December 2018, the requested CDFW releases are higher than the Table 2E releases, reducing storage and affecting other authorized purposes of the reservoir.

2.5.6.4 *American River*

2.5.6.4.1 <u>Flow</u>

Flow releases from Folsom Reservoir are made for both flood control and to meet water quality objectives and demands in the Delta. This can result in rapid increases and decreases of flow during the winter and spring. As a result, dewatering and isolation of Steelhead redds has been documented (Hannon et al. 2003; Water Forum 2005; Hannon and Deason 2008). In addition to flow fluctuations, low flows also can negatively affect Lower American River Steelhead. At low flow levels, the availability of bar complexes and side channel areas characterized by habitat complexity in the form of velocity shelters, hydraulic roughness elements, and other forms of cover is limited.

Reclamation operates Folsom Dam and Reservoir to provide water for irrigation, M&I uses, hydroelectric power, recreation, water quality, flood control, and fish protection. Reclamation, operating under the SWRCB Decision 893 (D-893) adopted in 1958, allows flows at the mouth of the American River to fall as low as 250 cubic feet per second (cfs) from January through mid-September, with a minimum of 500 cfs required between September 15 and December 31. The D-893 decision does not address the requirements of the CVPIA, the 1995 Bay-Delta Plan, or biological opinions issued to protect Central Valley Steelhead. Reclamation and the SWCB and many stakeholders (Water Forum) agreed that D-893 did not provide sufficient protections for Central Valley Steelhead in the Lower American River. Recently, Reclamation has operated the Folsom/Nimbus complex to more modern protective requirements and habitat management plans by providing flows that far exceed those required in D-893.

NMFS provided a reasonable and prudent alternative (RPA) in their 2017 amendment to the 2009 RPA. In this amended RPA, NMFS requires the action of implementing the flow schedule specified in the Water Forum Flow Management Standard. This flow schedule developed by the Water Forum, Reclamation, USFWS, NMFS, and CDFW addresses minimum flows needed for Central Valley Steelhead and Fall-Run Chinook Salmon in the Lower American River. Furthermore, Reclamation shall convene the American River Group (ARG), composed of representatives from Reclamation, NMFS, USFWS, CDFW and the Water Forum, to make recommendations for management within the constraints of the Flow Management Standard. Reclamation shall ensure that flow, water temperature, Steelhead spawning, and Steelhead rearing monitoring is conducted annually to help inform the ARG process and to evaluate take associated with flow fluctuations and warm water temperatures.

2.5.6.4.2 Temperature

Water temperatures in the Lower American River are influenced by operations. In the Lower American River water temperatures are a function of the timing, volume, and temperature of water releases from Folsom and Nimbus Dams. Once water is released, river distance and environmental heat flux influences the water temperature further as it moves through the Lower American River (Bartholow 2000).

In response, NMFS issued an RPA action to maintain suitable oversummering temperatures for juvenile Central Valley Steelhead in the Lower American River. In the action, Reclamation is to prepare a draft Operations Forecast and Temperature Management Plan based on forecasted conditions and submit the draft plan to NMFS for review by May 1 of each year. The information provided in the Operations Forecast will be used in the development of the Temperature Plan. Reclamation will use an iterative approach, varying proposed operations, with the objective to attain the temperature compliance point at Watt Avenue Bridge. Operation of Folsom/Nimbus Dam complex and the water temperature control shutters at Folsom Dam will be used to maintain a daily average water temperature of 65°F or lower at Watt Avenue Bridge from May 15 through October 31.

2.5.6.5 *Feather River*

DWR will operate Oroville Dam consistent with the applicable NMFS and USFWS biological opinions for the Oroville Complex (FERC Project #2100-134). During the summer, DWR typically releases water from Lake Oroville to meet instream flow requirements and to supplement non-project Delta inflows needed to meet D-1641 requirements. Releases also include water for local deliveries and south-of-Delta export at Banks Pumping Plant.

DWR balances the cumulative storage between Lake Oroville and San Luis Reservoirs so as to meet its flood control requirements, Sacramento–San Joaquin Delta requirements, and deliver water supplies to its contracted water agencies consistent with all environmental constraints. Lake Oroville may be operated to convey water through the Delta to San Luis Reservoir via Banks under different schedules depending on Delta conditions, reservoir storage volumes, and storage targets. Decisions as to when to move water from Lake Oroville to San Luis Reservoir are based on many real-time factors.

2.5.6.6 *San Joaquin River*

Reclamation operates the Friant Division for flood control, irrigation, M&I, and fish and wildlife purposes. Facilities include Friant Dam, Millerton Reservoir, and the Friant-Kern and Madera Canals. Friant Dam provides flood control on the San Joaquin River, provides downstream releases to meet senior water rights requirements above Gravelly Ford, provides Restoration Flow releases under Title X of Public Law 111-11, and provides conservation storage as well as diversion into Madera and Friant-Kern Canals for water supply. Water is delivered to about a million acres of agricultural land in Fresno, Kern, Madera, and Tulare Counties in the San Joaquin Valley via the Friant-Kern Canal south into Tulare Lake Basin and via the Madera Canal north to Madera and Chowchilla Irrigation Districts. A minimum of 5 cfs is required to pass the last holding contract diversion located about 40 miles downstream of Friant Dam near Gravelly Ford.

The SJRRP implements the San Joaquin River Restoration Settlement Act in Title X of Public Law 111-11. USFWS and NMFS issued programmatic biological opinions in 2012 that included project-level consultation for SJRRP flow releases. Programmatic ESA coverage is provided in both the USFWS and NMFS biological opinions for flow releases, recapture of those flows in the lower San Joaquin River and the Delta, and all physical restoration and water management actions listed in the Settlement.

The Stipulation of Settlement of NRDC vs. Rogers, is based on two goals: the Restoration Goal and the Water Management Goal. To achieve the Restoration Goal, the Settlement calls for, among other things, releases of water from Friant Dam to the confluence of the Merced River (referred to as Restoration Flows) according to the hydrographs in Settlement Exhibit B. To achieve the Water Management Goal, the Settlement calls for the development and implementation of a plan for recirculation, recapture, reuse, exchange, or transfer of Restoration Flows for the purpose of reducing or avoiding impacts on water deliveries to all of the Friant Contractors caused by Restoration Flows. Recapture of Restoration Flows must occur downstream of the Merced River confluence. Recapture can occur at Banta-Carbona, Patterson, or West Stanislaus Irrigation District facilities, or at Jones or Banks Pumping Plants. Recapture of Restoration Flows in the Sacramento–San Joaquin Delta under this proposed action would average 33 TAF and range from about 17 TAF in a critical-high year to about 44 TAF in a normal-wet year. If Voluntary Agreements are approved, up to 50 percent of the February through June volume could be dedicated to Delta Outflow, up to an annual maximum of 50 TAF.

2.5.6.7 *Delta*

The main CVP and SWP facilities in the Delta provide for the export of water to the San Joaquin Valley and Southern California. The major CVP features are the Delta Cross Channel (DCC), Contra Costa Canal, Jones Pumping Plant, TFCF, Delta-Mendota Canal/California Aqueduct Intertie (Intertie), and Delta-Mendota Canal (DMC). The DCC is a controlled diversion channel between the Sacramento River and Snodgrass Slough. The CCWD diversion facilities use CVP water resources, and other water rights, to serve CCWD customers directly and to operate CCWD’s Los Vaqueros Project. The Jones Pumping Plant diverts water from the Delta to the head of the DMC. The main SWP Delta features are Suisun Marsh facilities, Banks Pumping Plant, Clifton Court Forebay (CCF), Skinner Fish Facility, and Barker Slough Pumping Plant. DWR also currently installs agricultural barriers between April and July to improve diversions for Delta water users.

2.5.7 Monitoring and Research Programs

Monitoring and research programs help provide information on Fall-Run and Late Fall–Run Chinook Salmon migration, survival, redd distribution. Since 2015, Reclamation has started Enhanced Acoustic Tag Salmonid Monitoring (EATSM). EATSM is part of the Salmon and Sturgeon Assessment of Indicators by Lifestage (SAIL) program, which improves monitoring by addressing vital population statistics rather than reliance upon old indexes. CDFW annually conducts aerial redd surveys and has gone through the process of digitizing recorded Fall-Run and Late Fall–Run Chinook Salmon redds in the upper Sacramento from 1990 to 2014.

Surveys also help CDFW compile annual population estimates of Chinook salmon. Information is entered into CDFW's GrandTab and is easily accessible through www.calfish.org. Reclamation funds monitoring, evaluation, and web-based data services through the Central Valley Prediction and Assessment of Salmon (SacPAS) tool online. This service also provides a publicly accessible reporting system of historical and current information (www.cbr.washington.edu/sacramento/).

2.5.7.1 *Monitoring Programs for Chinook Salmon*

Table 2.5-3. Summary of Chinook Salmon (SRWC) Monitoring Surveys, Protocols, and Precisions

Life Stage	Location	Agency	Protocol	Level of Precision	Protocol Citation
Upper River					
Adults	Upper river	CDFW	Carcass mark–recapture; McCormick Jolly Seber	Abundance 90% CI 10%	Bergman et al. 2012
Juveniles	RBDD	USFWS	Mark–recapture; gear efficiencies	Abundance 90% CI 35%	Poytress et al. 2014
Middle River					
Juveniles	GCID	GCID	RST	Counts reported	
	Tisdale	CDFW	RST	Counts reported	
	Knights Landing	CDFW	RST	Counts reported	
	Yolo Bypass	DWR	RST; Beach seines	Counts reported	
Tidal Estuary					
Juveniles	Sacramento	USFWS	Kodiak trawl	Counts reported	Honey et al. 2004
	Delta	USFWS	Beach seines	Counts reported	Honey et al. 2004
	Chipps Island	USFWS	Mid-water trawl	Abundance CV 20–-40%	Pyper et al. 2013
	Fish Protective Facility	USBR/ CDWR	Salvage; Loss estimate	Expanded counts per water volume	
Ocean					
Adults	Ocean fishery	NMFS	CWT recoveries; Cohort reconstruction	Not estimated for impact rates	O'Farrell et al. 2012
Multiple regions					
Hatchery juveniles	Multiple regions	NMFS	Reach-specific survival and movement rates; JSAT Acoustic Telemetry	Errors vary by reach	Michel et al. 2015
Adult migration	Flood bypasses	CDFW	Strandings and rescues	Counts reported	Purdy et al. 2015

2.5.7.2 *Fall-Run and Late Fall-Run Essential Fish Habitat*

EFH is defined as those waters and substrates necessary to fish for spawning, breeding, feeding, or growth to maturity. In 1999, the Pacific Fishery Management Council (PFMC) identified EFH for Central Valley Chinook Salmon stocks to include the Sacramento and San Joaquin Rivers and their tributaries as EFH. EFH includes not only the watersheds of the Sacramento and San Joaquin River basins but also the San Joaquin Delta (Delta) hydrologic unit, Suisun Bay hydrologic unit, and the Lower Sacramento hydrologic unit. Freshwater EFH for Chinook Salmon consists of four major habitat functions:

1. Spawning and incubation
2. Juvenile rearing

3. Juvenile migration corridors
4. Adult migration corridors and adult holding habitat

Projected impacts associated with the proposed action are expected to eliminate, diminish, and/or disrupt these EFH habitat functions for Fall-Run and Late Fall–Run Chinook Salmon at many sites within the action area.

2.6 Steelhead, California Central Valley DPS

2.6.1 ESA Listing Status

The California Central Valley Steelhead DPS was originally listed as threatened under the ESA on March 19, 1998 (63FR 13347), and the listing was reaffirmed on January 5, 2006 (71 FR 834) and updated April 14, 2014 (79 FR 20802). The DPS includes all naturally spawned populations of Steelhead in the Sacramento and San Joaquin Rivers and their tributaries, excluding steelhead from San Francisco and San Pablo Bays and their tributaries. The DPS includes all naturally spawned Steelhead populations below natural and man-made impassable barriers in the Sacramento and San Joaquin Rivers and their tributaries (63 FR 13347). Steelhead in two artificial propagation programs, the Coleman National Fish Hatchery (CNFH) and FRFH Steelhead hatchery programs, are considered to be part of the DPS. NMFS determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the DPS (71 FR 834).

In May 2016, NMFS completed a 5-year status review of the Central Valley Steelhead DPS. Based upon a review of available information, NMFS (2016c) recommended that the Central Valley Steelhead DPS remain classified as a threatened species. However, NMFS also indicated that the biological status of the DPS has declined since the previous status review in 2011. NMFS indicated that natural production of Steelhead continues to decline and is now at very low levels (NMFS 2016c). Their continued low numbers in most hatcheries, domination by hatchery fish, and relatively sparse monitoring makes the continued existence of naturally reproduced Steelhead a concern. Due to this declining trend, NMFS suggests that the DPS is likely to become endangered within the foreseeable future throughout all or a significant portion of its range (NMFS 2016c).

Based on new genetic evidence described by Pearse and Garza (2015), NMFS recommended that Steelhead originating from the Mokelumne River Hatchery be added to the Central Valley Steelhead DPS (just as FRFH fish are considered to be a native Central Valley stock and are listed as part of the DPS). NMFS also recommended that the status of the DPS should be monitored and Hatchery and Genetic Management Plans should mandate that all Central Valley Steelhead hatcheries collect a full set of biological data, including scale samples, length, weight, sex, origin, and state of maturity, from a subset of all returning fish (NMFS 2016c) Hatcheries also should be required to conduct studies of smolt survival using modern tagging methods such as PIT tags and/or acoustic tags.

2.6.2 General Life-History and Habitat Requirements

Steelhead have a complex suite of life history traits, including the capability to be anadromous or to be resident (i.e., rainbow trout). Spawning and rearing habitat for Steelhead is usually characterized as perennial streams with clear, cool to cold, fast flowing water with a high dissolved oxygen content and abundant gravels and riffles. The preferred flow velocity is in the range of one to three feet per second

(Raleigh et al. 1986). Steelhead use various mixtures of sand-gravel and gravel-cobble substrate for spawning, but optimal spawning substrate reportedly ranges from 0.25 to 4.0 inches in diameter (Reiser and Bjornn 1979). Optimal water temperatures for Steelhead adult immigration are reported to range from 46°F to 52°F (NMFS 2002; SWRCB 2003). Optimal conditions for Steelhead spawning and embryo incubation reportedly occur at water temperatures 52°F (NMFS 2002; SWRCB 2003). Water temperatures between 45°F and 65°F have been reported as preferred for fry and juvenile Steelhead rearing (NMFS 2002). Upper lethal temperatures for adult Pacific salmonids are in the range of 75°F to 77°F for continuous long-term exposure (Brett et al. 1982). NMFS (2002) reported 65°F as the upper water temperature limit preferred for the growth and development of Sacramento and American river juvenile Steelhead. Steelhead successfully undergo the smolt transformation at water temperatures between 43.7°F to 52.3°F (Myrick and Cech 2001).

Adult Steelhead immigration into Central Valley streams typically begins in August, continues into March or April (McEwan 2001; NMFS 2014), and generally peaks during January and February (Moyle 2002), but adult Steelhead immigration can potentially occur during all months of the year (NMFS 2009). Steelhead spawning generally occurs from December through April, with peaks from January through March, in small streams and tributaries (NMFS 2009).

Eggs usually hatch within 4 weeks, depending on stream temperature, and the yolk sac fry remain in the gravel after hatching for another 4 to 6 weeks (CDFG 1996). After fry emerge, they inhabit shallow areas along the stream margin and prefer areas with cobble substrates, then use a greater variety of habitats as they grow and develop (CDFG 1996). Habitat use is affected by the presence of predators and juvenile Steelhead survival increases when cover, such as wood debris and large cobble, is available (Mitro and Zale 2002). The preferred range of water depths for spawning Steelhead has been observed most frequently between 0.3 and 4.9 feet (Moyle 2002). The reported preferred water velocity for Steelhead spawning is 1.5 to 2.0 feet per second (USFWS 1995).

Juvenile Central Valley Steelhead typically migrate to the ocean after spending 1 to 3 years in freshwater (CDFG 1996). Steelhead fry and fingerlings rear and migrate downstream in the Sacramento River during most months of the year, but the peak period of emigration is January to June (Hallock et al. 1961; McEwan 2001). Based on CDFW sampling at Knights Landing, juvenile Steelhead emigration occurs primarily from January through April, with peaks during January and February (NMFS analysis of 1998-2011 CDFW data.). Because of their varied freshwater residence times Steelhead fry and fingerlings can be rearing and migrating in the Sacramento River year-round (McEwan 2001).

2.6.3 Historical and Current Distribution and Abundance

Historically, Central Valley steelhead were distributed from the upper Sacramento and Pit river systems (upper Sacramento, McCloud, Pit and Fall rivers) south to the Kings River (and possibly Kern river systems in wet years) and in both east- and west- side tributaries of the Sacramento River and east-side tributaries of the San Joaquin River (McEwan 2001). Presently, Central Valley Steelhead are found in the Sacramento River downstream of Keswick Dam and in the major tributary rivers and creeks in the Sacramento River watershed. Zimmerman et al. (2009) found Steelhead present in three tributaries to the San Joaquin River (Stanislaus, Tuolumne, and Merced rivers) as well as in the Calaveras Rivers, and a hatchery supported Steelhead population occurs in the the Mokelumne River. The populations in the Feather and American Rivers are supported primarily by the Feather and Nimbus hatcheries. Other major Steelhead populations in the Sacramento River watershed are found in Battle, Mill, Deer, Clear and Butte Creeks. Steelhead also occur in many tributaries to the Sacramento River including Stony and Thomes Creeks (McEwan 2001), as well as intermittent streams in the Redding area.

In the 1950s, Central Valley Steelhead populations numbered approximately 40,000 fish, while during the mid-1960s, the Steelhead population was estimated at 27,000 (DFG 1965, as cited in McEwan and Jackson 1996). McEwan and Jackson (1996) estimated the annual run size for Central Valley Steelhead to be less than 10,000 fish by the early 1990s. Since 2015, Steelhead population estimates continue to demonstrate significant variation as reflected by the hatchery returns for Feather River Hatchery (Figure 2.6-1), Nimbus Hatchery (Figure 2.6-2), Mokelumne River Hatchery (Figure 2.6-3), and CNFH (Figure 2.6-4). Steelhead returns have been lower than average (n = 1,480) on the American River during recent years with a return of 756 in 2016, 1,032 in 2017, and 513 in 2018. Furthermore, Steelhead redd counts on the American River have been lower than average (n = 122) with 53 redds counted in 2015, 10 in 2017, and 63 counted in 2018 (Figure 2.6-5).

Monitoring efforts throughout the Central Valley inform Central Valley Steelhead abundance and distribution. During WY 2018, Steelhead catches in the Sacramento River drainage totaled: 5 at Tisdale weir; 3 at Knights Landing; none in the Sacramento beach seines; 4 in the Sacramento trawls; 12 in the Chipps Island trawls; 9,298 at Red Bluff Diversion Dam (Figure 2.6-6). In the San Joaquin River drainage, steelhead catches totaled: 11 adults (6 with adipose fin clips) at the Stanislaus weir; no juveniles at the Caswell rotary screw trap (RST); no adults at the Tuolumne weir (juvenile catch at the Tuolumne RST was not reported); and 8 smolts in the Mossdale trawls (USFWS 2018a; Stanislaus Operations Group 2018; Turlock Irrigation District and Modesto Irrigation District 2018). During WY 2017, Steelhead catches in the Sacramento River drainage totaled: 3 at Tisdale weir; 10 at Knights Landing; none in the Sacramento beach seines; 13 in the Sacramento traw16ls; 16 in the Chipps Island trawls; and 22,961 at Red Bluff Diversion Dam (Figure 2.6-7). In the San Joaquin River drainage, steelhead catches totaled: 26 adults (14 with adipose fin clips) at the Stanislaus weir; one adult at the Tuolumne weir; none at the Tuolumne RST (juvenile catch at the Stanislaus RST was not reported); and none in the Mossdale trawls (USFWS 2018a; Stanislaus Operations Group 2018; Turlock Irrigation District and Modesto Irrigation District 2017). During WY 2017, 65 wild juvenile and 43 hatchery Steelhead were observed at the Delta fish facilities, and 1,118 wild juvenile and 732 hatchery Steelhead were observed during WY 2018 (Figures 2.6-8 and 2.6-9). Steelhead have not been observed in CCWD's Fish Monitoring Program at the Rock Slough Intake since 2012.

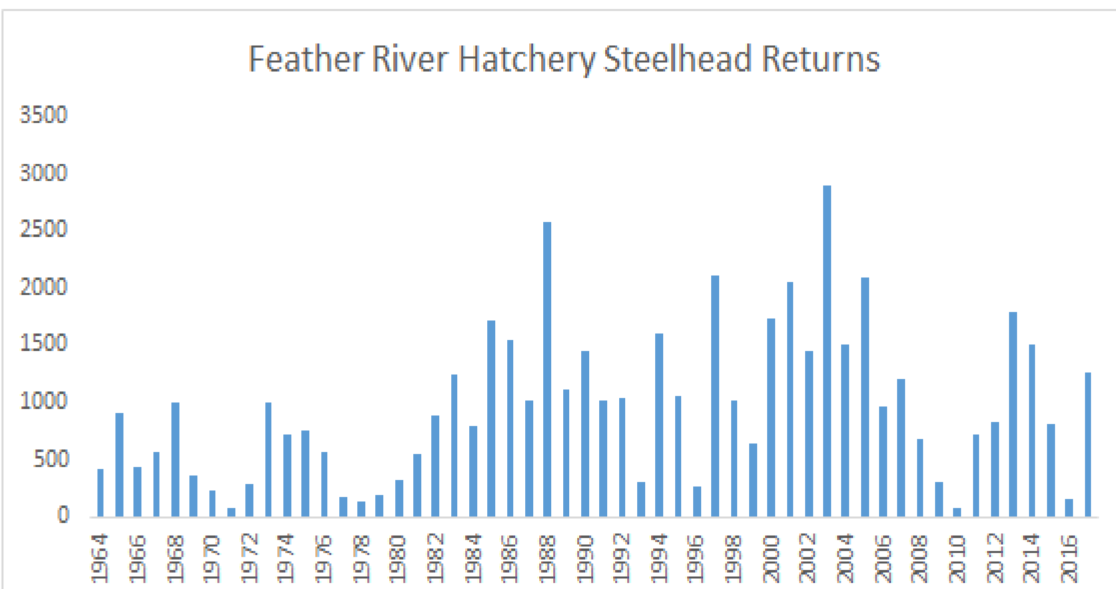

Figure 2.6-1. Steelhead Returns to the Feather River Hatchery, 1956–2017

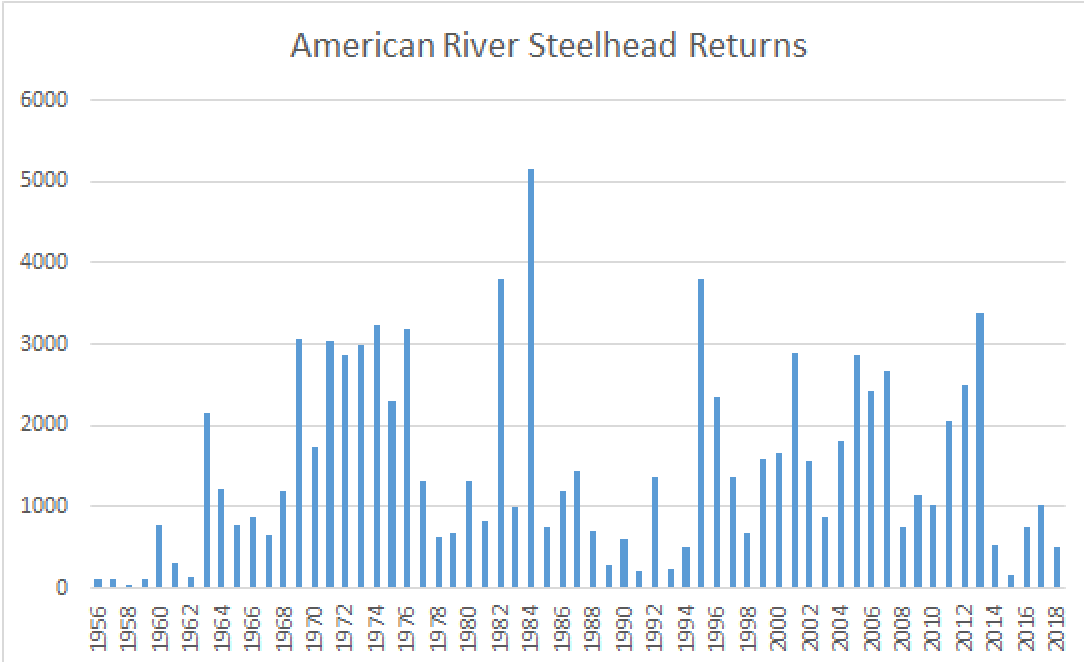

Figure 2.6-2. Steelhead Returns to the Nimbus Hatchery, 1956–2018

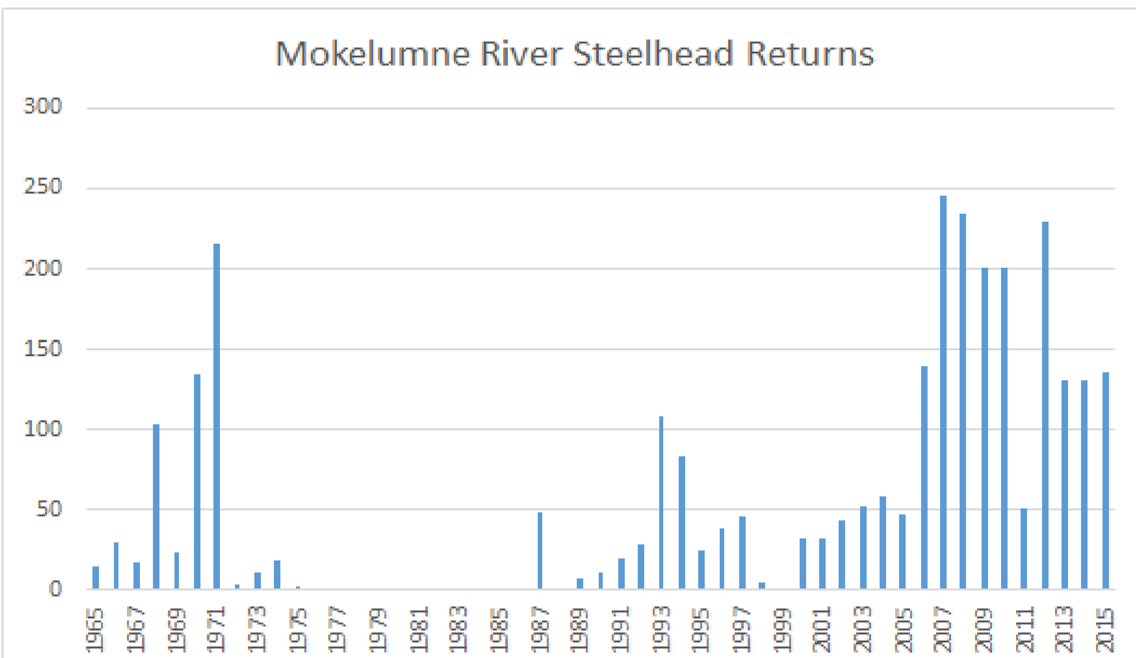

Figure 2.6-3. Steelhead Returns to the Mokelumne River, 1965–2015

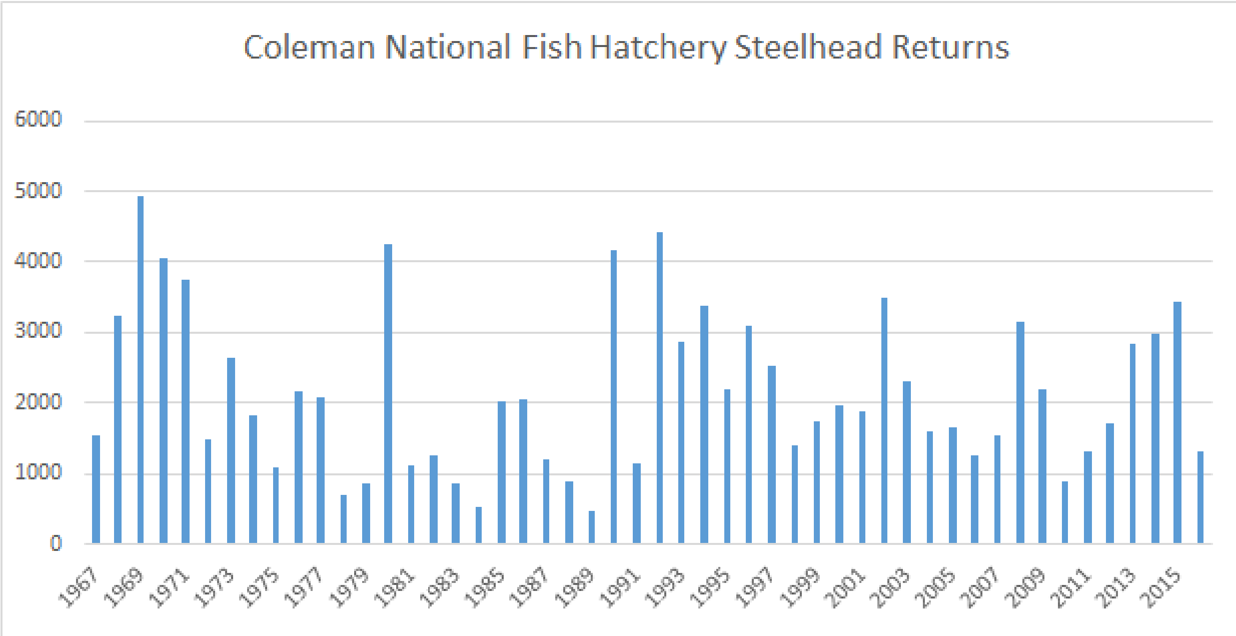

Figure 2.6-4. Steelhead Returns to Coleman National Fish Hatchery, 1967–2016

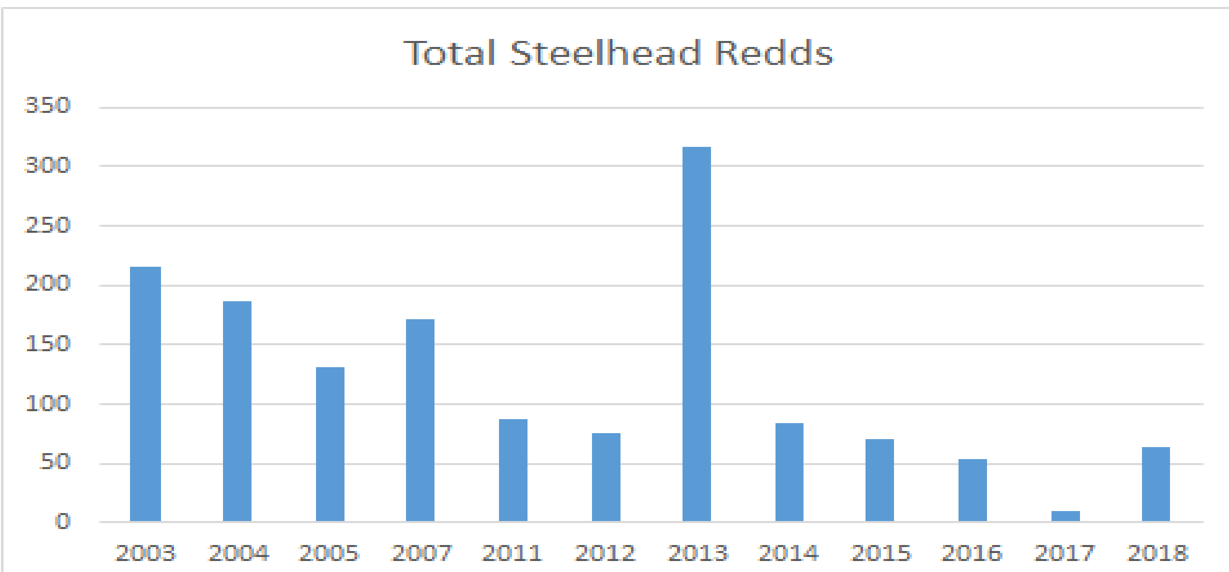

Figure 2.6-5. Total Steelhead Redds on the American River, 2003–2005, 2007, 2011–2018

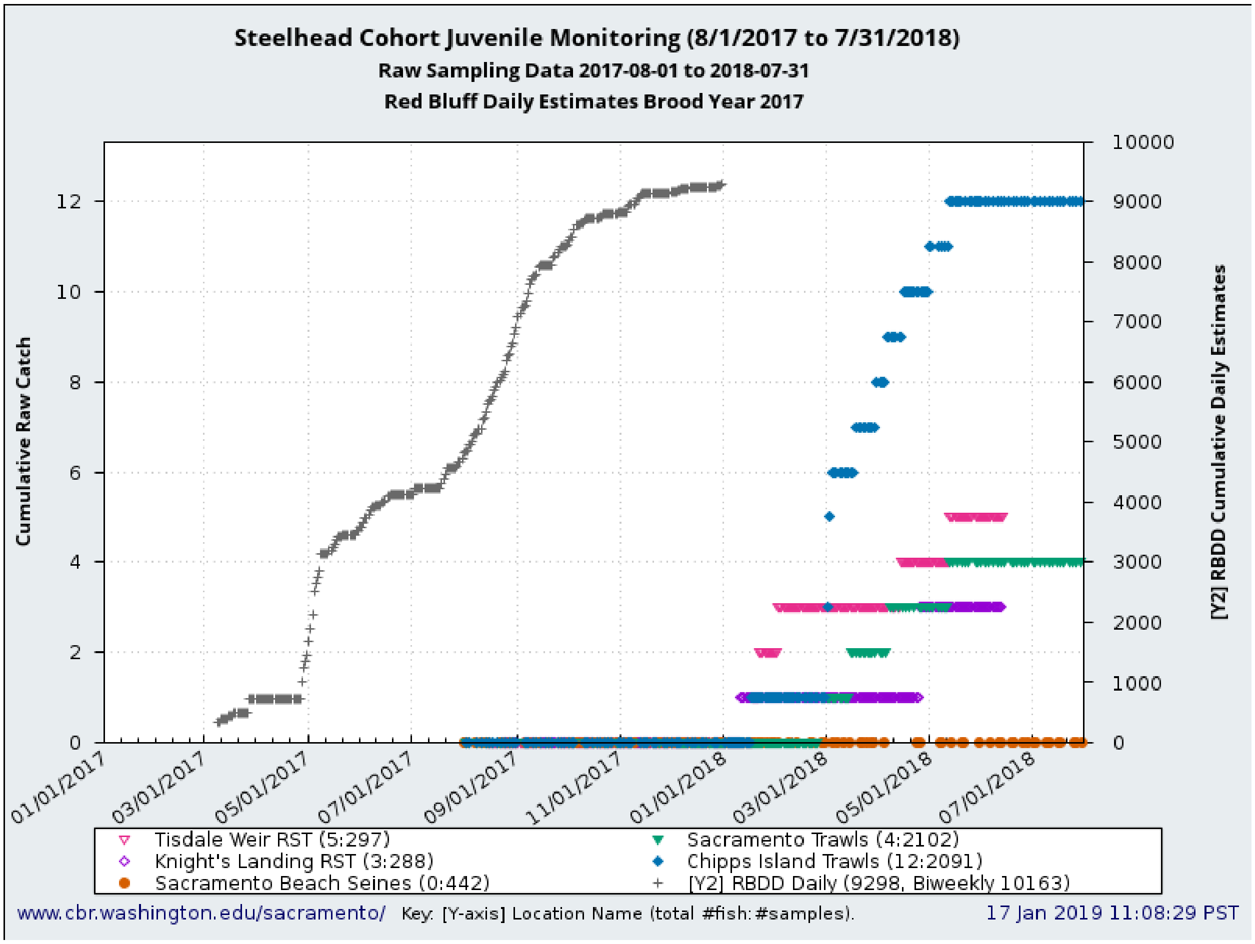

Figure 2.6-6. Juvenile Central Valley Steelhead Monitoring at Tisdale, Knights Landing, Sacramento Beach Seines, Sacramento Trawl, Chipps Island Trawl, and Red Bluff Diversion Dam for WY 2018

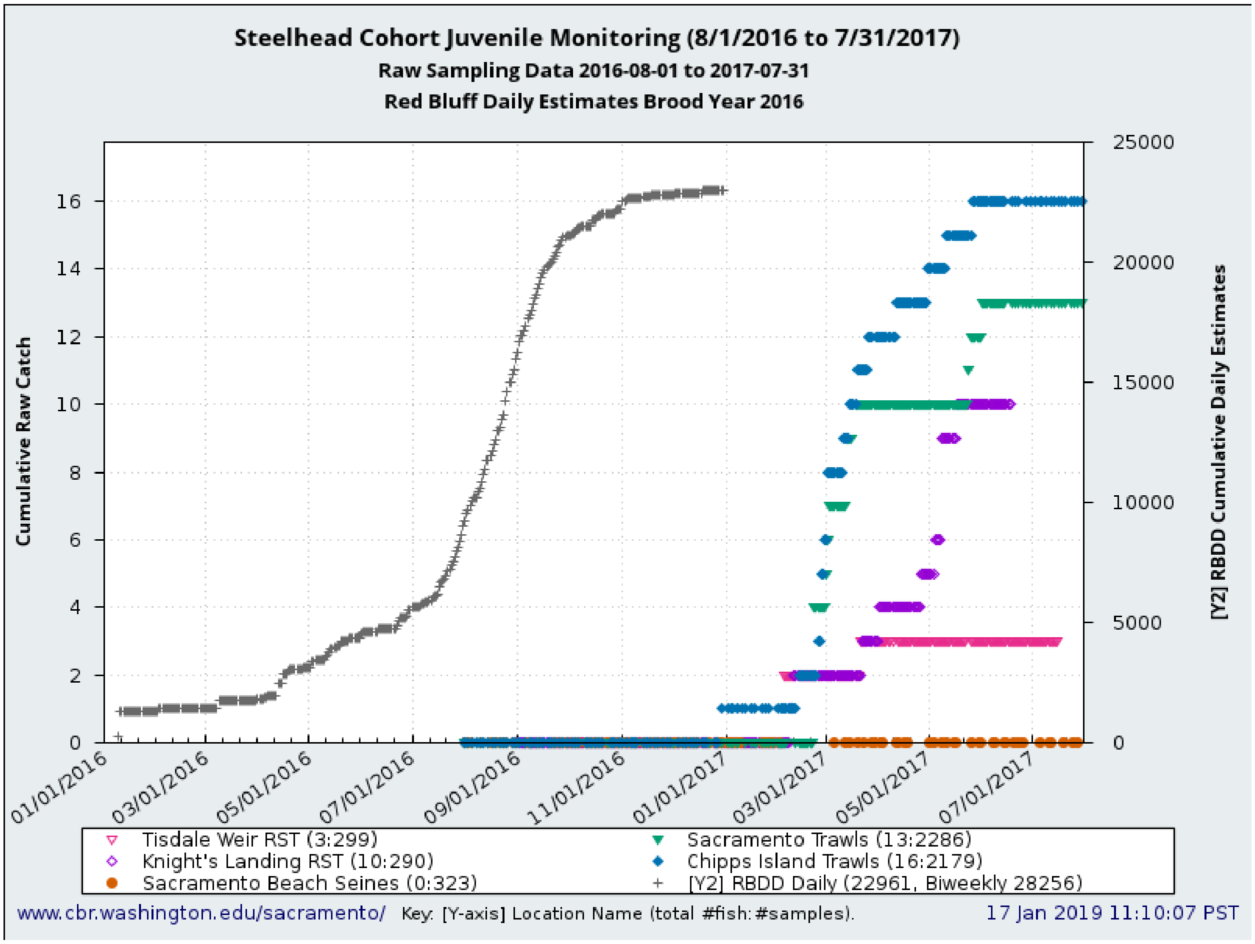

Figure 2.6-7. Juvenile Central Valley Steelhead Monitoring at Tisdale, Knights Landing, Sacramento Beach Seines, Sacramento Trawl, Chipps Island Trawl, and Red Bluff Diversion Dam for WY 2017

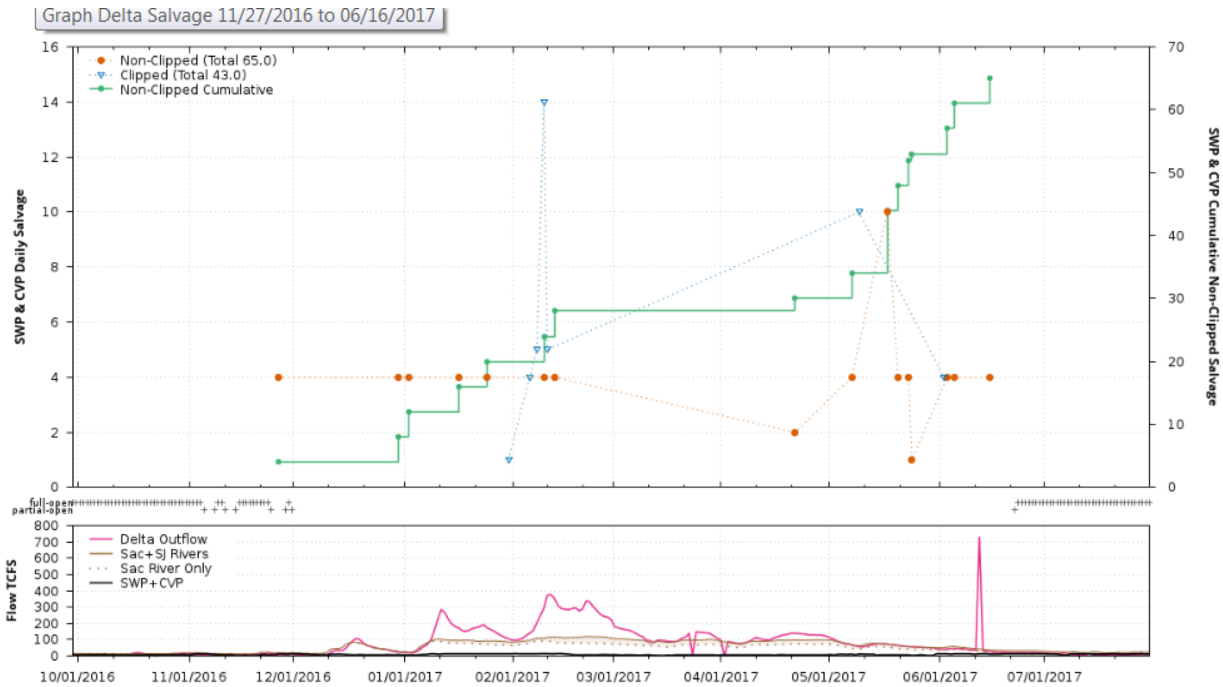

Figure 2.6-8. Central Valley Steelhead Salvage at the Delta Fish Facilities during WY 2017

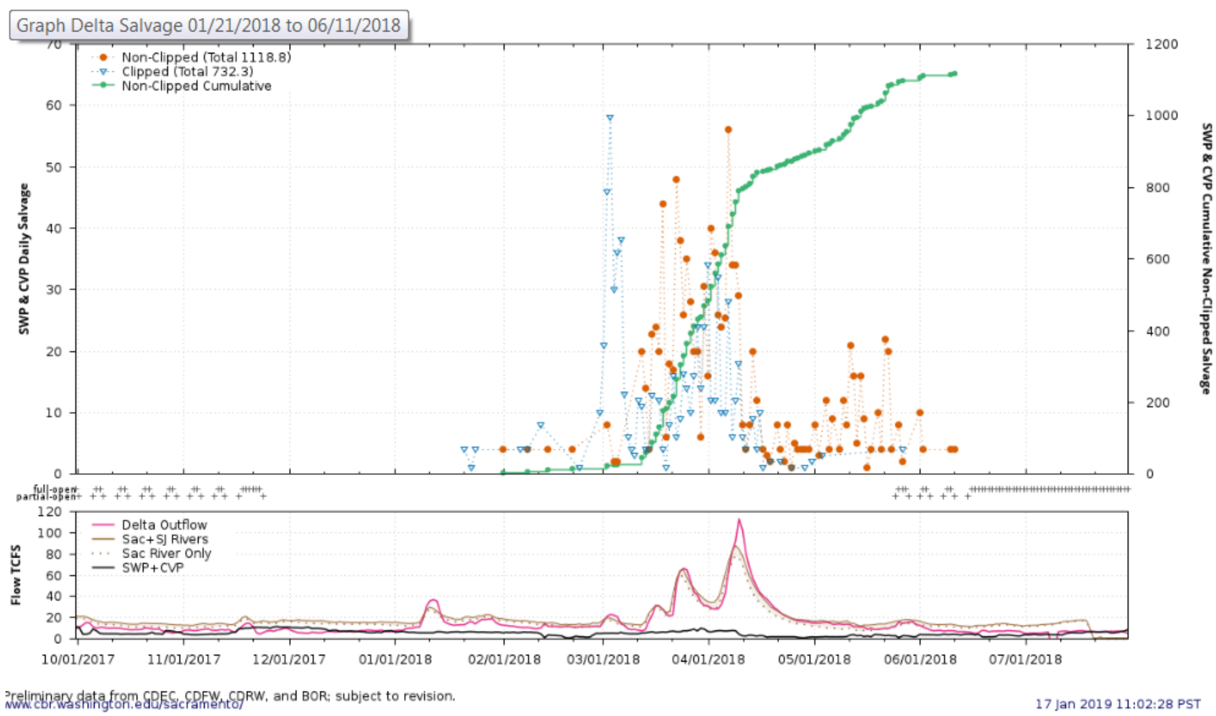

Figure 2.6-9. Central Valley Steelhead Salvage at the Delta fish facilities during WY 2018

2.6.4 Limiting Factors, Threats, and Stressors

As with the other salmonid species described above and further discussed in Section 3.1, high water temperatures in remaining rearing areas, effects from hatcheries and the rearing of out of basin stocks, limited quantity and quality of rearing habitat, ocean conditions, and predation and entrainment into diversions at the CVP and SWP pumping facilities all affect the species. Degradation of the remaining accessible habitat through reducing flow variability, blocking coarse sediment recruitment, operation of outdated fish screens, ladders and diversion dams, simplified habitat due to levee construction and maintenance and disconnection of off-channel habitat, water delivery and hydroelectric operation on both the Sacramento and Feather Rivers affect natural flow regimes.

Future increasing temperatures and altered precipitation patterns due to climate change will also pose stressors on Central Valley Steelhead. These factors are the same for Steelhead as those described previously for Chinook Salmon.

Figure 2.6-10 below shows water year type average flows on the American River along with timing of the fish species in the American River.

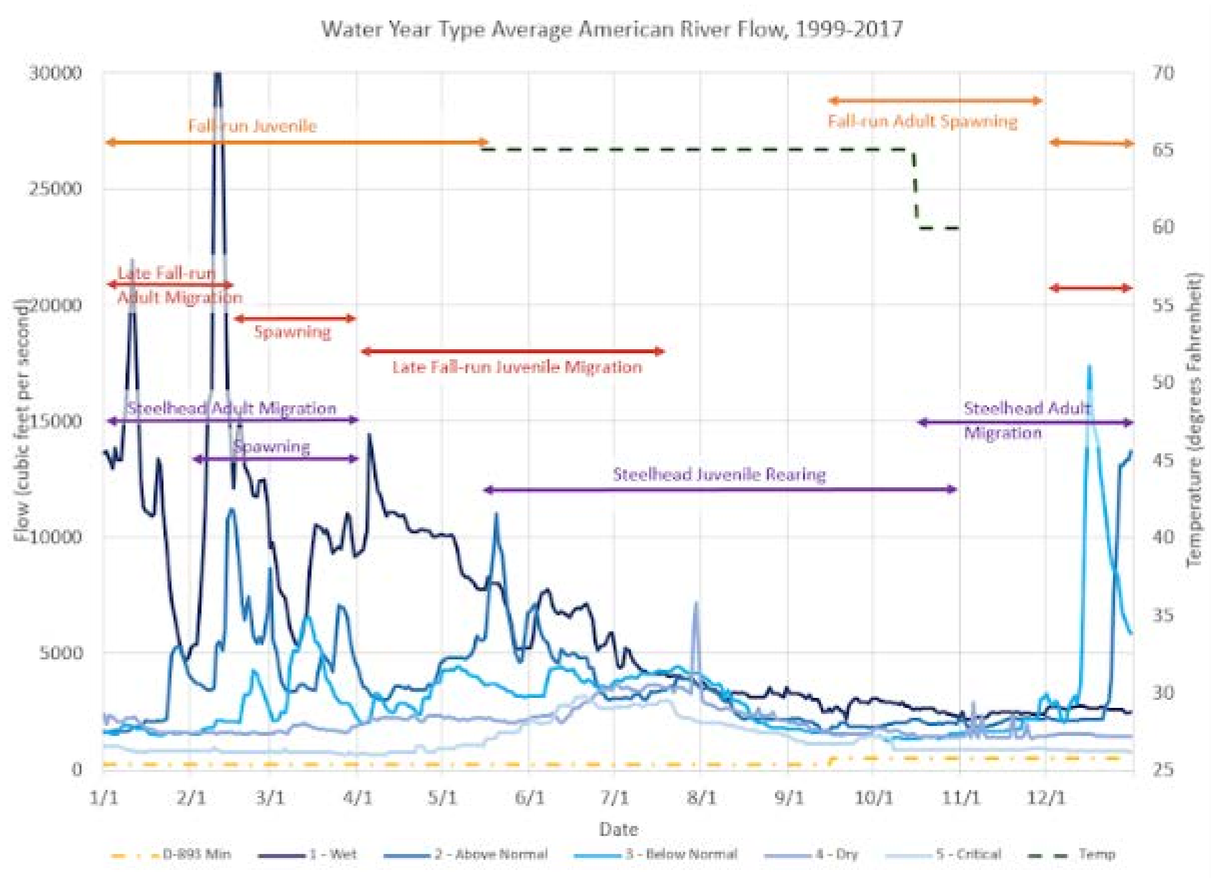

Figure 2.6-10. Water Year Type Average American River Flow, 1999–2017

2.6.5 Recovery and Management

As discussed above in Section 5.1.6, the NMFS 2014 Recovery Plan for Sacramento River Winter-Run Chinook Salmon, Central Valley Spring-Run Chinook Salmon, and Central Valley Steelhead outlines actions to restore habitat, access, and improve water quality and quantity conditions in the Sacramento River to promote the recovery of listed salmonids. Many of the Recovery and Management Action for Winter-Run and Spring-Run Chinook Salmon also benefit Steelhead.

2.6.6 Water Operations Management

2.6.6.1 *Upper Sacramento River*

Water Operations Management for the upper Sacramento River for Central Valley Steelhead would be the same as those in Winter-Run Chinook species account.

2.6.6.2 *American River*

Reclamation operates Folsom Dam and Reservoir to provide water for irrigation, municipal and industrial uses, hydroelectric power, recreation, water quality, flood control, and fish protection. Reclamation operation under SWRCB Decision 893 (D-893) adopted in 1958 allows flows at the mouth of the American River to fall as low as 250 cfs from January through mid-September, with a minimum of 500 cfs required between September 15 and December 31. The D-893 decision does not address the requirements of the CVPIA, the 1995 Bay Delta Plan, or biological opinions issued to protect Central Valley Steelhead.

Reclamation’s 2008 Biological Assessment proposed implementing the Water Forum Flow Management Standard. This flow schedule, developed by the Water Forum, Reclamation, USFWS, NMFS, and CDFW, addresses minimum flows for Central Valley Steelhead and Fall-Run Chinook in the Lower American River.

Reclamation convenes the American River Group (ARG), comprised of representatives from Reclamation, NMFS, USFWS, CDFW and the Water Forum, to make recommendations for management within the constraints of the Flow Management Standard. Reclamation ensures that flow, water temperature, Steelhead spawning, and Steelhead rearing monitoring is conducted annually in order to help inform the ARG process and to evaluate take associated with flow fluctuations and warm water temperatures.

Flow releases from Folsom Reservoir are made for both flood control and to meet water quality objectives and demands in the Delta. This can result in rapid increases and decreases of flow during the winter and spring. Dewatering and isolation of Steelhead redds has been documented (Hannon *et al*. 2003, Water Forum 2005, Hannon and Deason 2008) as a result. In addition to flow fluctuations, low flows also can negatively affect lower American River Steelhead. At low flow levels, the availability of bar complexes and side channel areas characterized by habitat complexity in the form of velocity shelters, hydraulic roughness elements, and other forms of cover is limited.

Water temperatures in the lower American River are influenced by the timing, volume, and temperature of water releases from Folsom and Nimbus dams. Once released, river distance and environmental heat flux influences the water temperature further as it moves through the Lower American River (Bartholow 2000). The NMFS RPA Action II.2 requires suitable over summering temperatures for juvenile CV Steelhead in the Lower American River. In the RPA, Reclamation is to prepare a draft Operations

Forecast and Temperature Management Plan based on forecasted conditions and submit the draft Plan to NMFS for review by May 1 of each year. The information provided in the Operations Forecast will be used in the development of the Temperature Plan. Reclamation uses an iterative approach, varying proposed operations, with the objective to attain the temperature compliance point at Watt Avenue Bridge. Operation of Folsom/Nimbus Dam complex and the water temperature control shutters at Folsom Dam are used to maintain a daily average water temperature of 65°F or lower at Watt Avenue Bridge from May 15 through October 31.

2.6.6.3 *Clear Creek*

Water Operations Management for Clear Creek for Central Valley Steelhead would be the same as those in the Fall-run Chinook species account.

2.6.6.4 *Stanislaus River*

Water Operations Management for the Stanislaus River for Central Valley Steelhead would be the same as those in the Fall-run Chinook species account.

2.6.7 Monitoring and Research Programs

The Central Valley Steelhead Monitoring Program (CVSMP), a pilot study, began implementing monitoring projects on the Sacramento River and select tributaries to help identify Central Valley Steelhead populations. The CVSMP projects include (1) Mainstem Sacramento River Mark-Recapture; (2) Upper Sacramento River Tributary Escapement Monitoring; (3) Sacramento River Tributary Mark-Recapture Monitoring; and (4) Hatchery Broodstock and Angler Harvest Sampling. These projects began July 2015 under contract with Pacific States Marine Fisheries Commission (PSMFC). The objective of the CVSMP pilot study was to evaluate the efficacy and success of these monitoring projects in order to expand these techniques throughout the Sacramento and San Joaquin watersheds.

Reclamation performed a 6-year Steelhead telemetry study on the Stanislaus River and currently working to continue an acoustic tagging study on the San Joaquin River to determine entrainment of SJR origin Steelhead into Tracy and Jones Pumping Plants.

The Stanislaus River Research and Monitoring Program is the most comprehensive and longest running Salmon and Steelhead monitoring programs in California’s San Joaquin Basin, although data is not publicly available. Initiated by FISHBIO personnel in 1993 for the Oakdale and South San Joaquin Irrigation Districts and Tri-Dam Project, the program’s suite of ongoing monitoring activities tracks the abundance, distribution, migration characteristics, and habitat use of salmon and Steelhead trout.

2.7 Steelhead, Central Valley Critical Habitat

Critical habitat for the Central Valley Steelhead DPS was designated in 2005 and includes all river reaches accessible to Steelhead in the Sacramento and San Joaquin rivers and their tributaries, the Delta, and the Yolo Bypass (70 FR 52488). A 2016 status review found that the DPS continues to be at a high risk of extinction (NMFS 2016c). In the Sacramento and San Joaquin Rivers and tributaries, critical habitat includes the river water column, river bottom, and adjacent riparian zone and including the water column and essential foraging habitat and food resources west of Chipps Island including San Francisco Bay to the Golden Gate Bridge.

Critical habitat consists of PBFs considered essential for the conservation of a species. PBFs outlined in the designation of critical habitat (70 FR 52488) are:

1. Unimpeded access from the Pacific Ocean to appropriate spawning areas in the Sacramento and San Joaquin River and their tributaries
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 for successful spawning, egg incubation, and fry development
5. Habitat and prey free of contaminants
6. Riparian habitat that provides for successful juvenile development and survival
7. Unimpeded passage of juveniles downstream from the spawning grounds to San Francisco Bay and the Pacific Ocean

2.8 Steelhead, Central California Coast DPS

2.8.1 ESA Listing

The Central California Coast (CCC) Steelhead DPS was listed as threatened under the ESA on January 5, 2006 (71 FR 834). The CCC Steelhead DPS includes all naturally spawned Steelhead populations below natural and human-made impassable barriers in California streams from the Russian River (inclusive) to Aptos Creek (inclusive), and the drainages of San Francisco, San Pablo, and Suisun bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers. Tributary streams to Suisun Marsh include Suisun Creek, Green Valley Creek, and an unnamed tributary to Cordelia Slough, excluding the Sacramento-San Joaquin River Basin, as well as two artificial propagation programs (NMFS 2009).

2.8.2 General Life-History and Habitat Requirements

Steelhead return to their natal streams to spawn typically as 2- to 4-year-old adults. Adults generally migrate upstream from November through March to spawn, but may extend into April (NMFS 2011) (Table 2.8-1). Spawning occurs between January and April. Time of incubation and hatching varies with region, habitat, water temperature, and spawning season. CCC Steelhead incubation occurs between January and May. Alevins emerge from their redds following yolk sac absorption and are ready to feed as fry or juveniles. Following emergence, fry live in small schools in shallow water along streambanks. The diet of juvenile Steelhead includes emergent aquatic insects, aquatic insect larvae, snails, amphipods, opossum shrimp, and small fish (Moyle 2002). Steelhead usually do not eat when migrating upstream and often lose body weight. As Steelhead grow, they establish individual feeding territories; juveniles typically rear for 1 to2 years (and up to 4 years) in streams before emigration as smolts (NMFS 1996). Steelhead may remain in the ocean from 1 to 4 years, growing rapidly as they feed in the highly productive currents along the continental shelf (Barnhart 1986).

Table 2.8-1. Timeline of Steelhead Life Stages

Life Stage	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Spawning	•	•	•	•								
Migration	•	•	•	•							•	•
Incubation	•	•	•	•	•							
Rearing	•	•	•	•	•	•	•	•	•	•	•	•

Source: NMFS 2011

2.8.3 Historical and Current Distribution and Abundance

Historically, approximately 70 populations of Steelhead existed in the CCC Steelhead DPS (Spence et al. 2008; Spence et al. 2012): 37 independent or potentially independent and 33 dependent. While historical and present data on abundance are limited, CCC Steelhead populations are substantially reduced from historical levels. CDFG (1965) estimated a total of 94,000 adult Steelhead spawned in the rivers and streams of this DPS during the mid-1960s, including 50,000 fish in the Russian River—the largest population within the DPS. Near the end of the 20th Century, the Steelhead population in the Russian River was believed to have declined substantially and local CDFG biologists estimated the wild run population in the Russian River Watershed was between 1,700-7,000 fish (McEwan and Jackson 1996). Abundance estimates for smaller coastal streams in the DPS indicate low but stable levels with individual run size estimates for several streams (Lagunitas, Waddell, Scott, San Vicente, Soquel, and Aptos creeks) of approximately 500 fish or less (62 FR 43937). Some loss of genetic diversity has been documented and attributed to previous out-of-basin transfers of hatchery stock as well as local hatchery production (Bjorkstedt et al. 2005, Good et al. 2005). In particular, for streams that are tributary to San Francisco Bay, reduced population sizes and habitat fragmentation caused by intense urbanization and water resource development have also led to a loss of genetic diversity in these populations.

CCC Steelhead have experienced significant declines in abundance and long-term population trends suggest a negative growth rate. This indicates the DPS may not be viable in the long term. Independent populations that historically provided enough Steelhead immigrants to support nearby dependent populations may no longer be able to do so, placing these dependent populations at increased risk of extirpation. However, because CCC Steelhead remain present in most streams throughout the DPS, roughly approximating the known historical range, CCC Steelhead may possess a resilience that is likely to slow their decline relative to other salmonid DPSs or ESUs in worse condition. Their iteroparous life history and variation in time spent in streams and the ocean have helped the Steelhead populations respond to different pressures on their population (Busby et al. 1996).

The 2005 status review concluded the CCC Steelhead DPS remains “likely to become endangered in the foreseeable future” (Good et al. 2005). In its most recent 5-year review of the DPS, NMFS determined that the DPS should remain listed as threatened (NMFS 2016; Williams et al. 2016).

2.8.4 Limiting Factors, Threats and Stressors

Limiting factors affecting Central Coast Steelhead include degradation of habitat through water quality, water quantity, wetland loss, timber harvest, agriculture including marijuana-related diversion dams, urbanization, and impaired passage; illegal harvest; predation; invasive species; drought and climate change; and the existing small population size.

In addition, the 2012-2015 drought has revealed that during low storage levels, Coyote Valley Dam is known to release highly turbid water for extended periods well after turbidity levels in reservoir inflows and unregulated tributaries have diminished (NMFS 2008a). Turbid flows result in degraded salmonid spawning and rearing habitat (Everest 1969), and may impair food availability for juvenile salmonids by reducing habitat diversity for benthic invertebrates and eliminating certain guilds of invertebrates from the food chain. Similarly, extended periods of warm, turbid, and reduced flow releases have been noted at dams in the San Francisco Bay Area during periods of low storage (Leicester and Smith 2014).

Freshwater poaching or unintentional take of CCC Steelhead may occur. Where current abundance is below the “high risk” threshold (as described in Spence et al. 2008), losing adult fish to poaching could significantly impact population productivity and genetic diversity.

Many populations of CCC Steelhead have declined in abundance to levels that are well below low-risk abundance targets, and several are, if not already extirpated, likely below the high-risk depensation thresholds specified by Spence et al. (2008). Recently the largest donor population in the Russian River has declined, increasing the risk. As natural populations get smaller, stochastic processes may cause alterations in genetics, breeding structure, and population dynamics. Even though recent data suggests some CCC Steelhead populations are doing better than others, all populations remain at severely depressed levels, suggesting stochastic processes continue to remain a high threat to the species (NMFS 2016).

2.8.5 Water Operations Management

Operations of the Suisun Marsh Salinity Control Gates and other infrastructure in Suisun Marsh could affect CCC Steelhead by blocking or allowing access to Suisun Creek, where a population of California Central Coast Steelhead exists.

2.8.6 Recovery and Management

Recovery actions for CCC Steelhead include over 7000 actions ranging from increasing the quality and extent of estuarine habitat to requesting that the SWRCB review and/or modify water use based on the needs of Steelhead and authorized diverters (NMFS Recovery Plan, October 2016).

2.8.7 Monitoring and Research Programs

Reclamation does not currently conduct research or monitoring on CCC Steelhead.

2.9 Steelhead, Central Valley California Coast Critical Habitat

CCC Steelhead critical habitat was designated September 2, 2005. Five watersheds with CCC Steelhead are in the San Francisco-San Pablo Suisun Bay estuarine complex which provides rearing and migratory habitat for this ESU.

2.10 North American Green Sturgeon, Southern DPS

2.10.1 ESA Listing Status

NMFS listed the southern DPS of North American Green Sturgeon as threatened in 2006 (71 FR 17757). In 2015, NMFS issued an updated status review in which the threatened status was confirmed (NMFS 2015). Green Sturgeon are known to spawn in the Sacramento and Klamath Rivers in California, and the Rogue River in Oregon (Moyle et al. 1992; Adams et al. 2002). Genetic analyses indicates that the Sacramento, Klamath, and Rogue Rivers support two distinct reproducing populations identified as southern and northern DPS Green Sturgeon (Israel et al. 2004). The threatened southern DPS is limited to a single reproducing population in the Sacramento River (71 FR 17757). The Northern DPS includes sturgeon from the Klamath and Rogue Rivers and is considered by NMFS a Species of Concern.

2.10.2 General Life-History and Habitat Requirements

Green Sturgeon are anadromous, with larval and juvenile life stages residing in natal rivers and subadult and adult life stages residing in estuarine and coastal marine waters before returning to freshwater to spawn. Green Sturgeon are long lived, reaching maturity at about age 15 and typically spawning every 3 to4 years (NMFS 2015). Adult Green Sturgeon enter San Francisco Bay in late winter through early spring and migrate to spawning areas in the Sacramento River primarily from late February through April. Spawning primarily occurs April through late July although late summer and early fall spawning may also occur based on the presence of larvae in the fall (Heublein et al. 2017). Elevated water flow appears to be an important cue in triggering migration and subsequent spawning of adult Green Sturgeon (Benson et al. 2007; Erickson and Webb 2007; Heublein et al. 2009). Spawning of Southern DPS Green Sturgeon primarily occurs in the mainstem Sacramento River although a spawning event was documented in 2011 in the lower Feather River at the Thermalito Afterbay Outlet (Seesholtz et al. 2012).

Green Sturgeon spawn in deep pools in large, turbulent, freshwater river mainstems (Moyle et al. 2002). Green Sturgeon eggs are generally broadcast eggs over large gravel and cobble substrates where they adhere or settle into crevices (Van Eenennaam et al. 2001; Poytress et al. 2011). Substrates in spawning pools range from small to medium-sized sand to boulders and bedrock (Klimley et al. 2015a; Poytress et al. 2015; Wyman et al. 2018). Water temperature is an important factor for Green Sturgeon spawning and egg viability. Temperatures in the upper Sacramento River during the spawning period have ranged from 10.1°C to 17.6°C (Poytress et al. 2012). Wyman et al. (2017) studied the physical variables selected by adult Green Sturgeon during their spawning period using a two-dimensional model to integrate fish locations, physical habitat characteristics, discharge, bathymetry, and simulated velocity and depth. Results indicated that Green Sturgeon prefer spawning habitats with velocities between 1.0 and 1.1 meters per second, depths of 8 to 9 meters, and gravel and sand substrate (Wyman et al. 2017). After spawning, adults spend variable lengths of time in the river and estuaries before returning to the ocean (Heublein et al. 2017b). Outmigration may occur in late spring or summer (possibly in response to elevated flows) but most adults appear to remain in spawning and holding areas through the summer and leave in the fall (Benson et al. 2007; Heublein et al. 2009).

Development and survival of Green Sturgeon embryos is temperature dependent. Laboratory studies of Northern DPS Green Sturgeon indicate that eggs hatch after 144 to192 hours when incubated at a temperature of 15.7 ± 0.2°C (Deng et al. 2002). Based on exposure of eggs to water temperatures ranging from 11°C to 26°C, Van Eenennaam et al. (2005) found that optimal water temperatures for development generally range from 14°C to 17°C. Water temperatures above 17°C resulted in increased rates of deformities and mortality risk, with total mortality occurring at temperatures of 23°C and above (Van Eenennaam et al. 2005).

After hatching, Green Sturgeon larvae possess limited swimming ability and generally seek refuge in low-velocity habitat, suggesting that complex habitat such as large cobble substrate is critical for this life stage (Kynard et al. 2005). Larvae transition from endogenous to exogenous feeding at approximately 15 days after hatching and initiate downstream migration at approximately 18 days (Gisbert et al. 2001, Poytress et al. 2011). Laboratory studies of nDPS Green Sturgeon indicate that optimal water temperatures for growth of larvae generally range from 17°C to 20°C when food is not limiting. Metamorphosis of Green Sturgeon larvae to juveniles occurs at approximately 45 days post-hatch at lengths of 62 to94 mm (Deng et al. 2002).

Little is known about rearing, migratory behavior, and general emigration patterns of juvenile Southern DPS Green Sturgeon. Based on captures of juveniles in the Sacramento River near Red Bluff, it is likely that juveniles rear near spawning habitat for a few months or more before migrating to the Delta (Heublein et al. 2017a). The lack of juveniles less than 200 mm FL in Delta capture records further supports extended upriver rearing of juveniles before entering the estuary (CDFG 2002), as well as the lack of catch in the 20-mm survey reported in Dege and Brown (2004). Growth of juvenile nDPS Green Sturgeon is rapid as they move downstream, reaching up to 300 mm TL in the first year and more than 600 mm TL in years 2 and 3 (Nakamoto et al. 1995). Laboratory studies of nDPS Green Sturgeon indicate that optimal bioenergetic performance (growth, metabolic rate, temperature preference, and swimming performance) occurs at 15°C to 19°C (Mayfield and Cech 2004). Estuarine residence appears to be variable with some entering the ocean in their first year and others remaining in the Delta for 2 to3 years (Heublein et al. 2017a).

After Green Sturgeon enter the ocean, they appear to make northerly migrations within nearshore waters along the west coast and congregate in non-natal coastal bays and estuaries during the late summer and early fall (Lindley et al. 2008; 2011; Huff et al, 2012).

Feeding data recorded for adult Green Sturgeon indicate that they consume benthic invertebrates such as shrimp, mollusks, amphipods, and small fish (Moyle et al. 1992).

2.10.3 Historical and Current Distribution and Abundance

North American Green Sturgeon are long-lived, wide ranging, and the most marine-oriented species of the sturgeon family. Green Sturgeon spend the majority of their lives in coastal waters between northern Baja California and the Aleutian Islands, Alaska (Moyle et al. 1992). They are known to spawn in the Sacramento and Klamath Rivers in California, and the Rogue River in Oregon (Moyle et al. 1992; Adams et al. 2002). The actual historical and current spawning distribution is unclear because Green Sturgeon make non-spawning movements into coastal lagoons and bays, and because their original spawning distribution may have been reduced because of migration barriers, flow regulation, and other anthropogenic effects (Mora et al. 2009). Based on surveys of sites where adult Green Sturgeon aggregate in the upper Sacramento River in 2010-2015, the total number of adults in the Southern DPS was estimated to be 2,106 ± 1,246-2,966 (Mora 2018).

Based on data from acoustic tags, adult Green Sturgeon currently migrate upstream as far as the mouth of Cow Creek near Bend Bridge on the Sacramento River (NMFS 2009). Spawning occurs from Hamilton City (rkm 332.5) to Cow Creek (rkm 451) based on adult distribution (Heublein et al. 2009; Klimley et al. 2015a; Mora et al. 2018). Egg mat sampling confirmed spawning between Hamilton City and Inks Creek (rkm 426) (Poytress et al. 2015). Green Sturgeon spawning also has been documented in the Feather

(Seesholtz et al. 2015) and Yuba Rivers (Beccio 2018). Records of Green Sturgeon in the San Joaquin River and its tributaries are rare and limited to information from angler report cards. However, Anderson et al. (2018) recently confirmed an adult Green Sturgeon holding in a deep pool near Knights Ferry in the Stanislaus River.

Based on records of spawning distribution and captures of larval Green Sturgeon, the distribution of larvae is estimated to extend at least 100 km downstream from spawning habitats on the Sacramento and Feather rivers in high flow years (NMFS 2018). Captures of larvae in traps at the Red Bluff Diversion Dam during 2003-2012 (27.3 mm average median TL) occurred primarily from May through August, with peak counts typically occurring in June and July, while captures of juveniles occurred sporadically from August through November (Poytress et al. 2014). Current information indicates that juvenile Green Sturgeon rear for up to 3 years in the Sacramento River, Delta, and San Francisco Bay before entering the ocean, but there is little information on residence times, movements, and emigration patterns of juveniles following metamorphosis in the Sacramento River. The lack of juveniles less than 200 mm FL in Delta capture records suggests extended upriver rearing of juveniles before entering the estuary (CDFG 2002).

2.10.4 Limiting Factors, Threats, and Stressors

The principal factor in the decline of the Southern DPS of Green Sturgeon is the reduction in historical spawning habitat (NMFS 2015). The population also is threatened by elevated water temperatures in spawning areas, entrainment and stranding in water and flood diversions, indirect effects of invasive species, potential poaching, and exposure to contaminants (NMFS 2015).

Fish passage barriers such as dams, weirs, and other flood control structures block or impede Green Sturgeon migration. Adams et al. (2007) hypothesized that significant amounts of historically-utilized Green Sturgeon spawning habitat may be blocked by Shasta Dam and Oroville Dam on the Feather River. According to habitat and observance monitoring and statistics by Mora et al. (2009), Shasta Dam and reservoir blocks access to reaches of the Pit, McCloud and Little Shasta rivers that contain apparently suitable habitat for Green Sturgeon. Similarly, Oroville Dam and reservoir block some areas of suitable habitat on the middle fork of the Feather River, and Daguerre Point Dam blocks some habitat on the Yuba River (Mora et al. 2009). Other potential migration barriers include the Sacramento Deep Water Channel Locks, Fremont Weir, Sutter Bypass, the DCC in the Sacramento River, and Shanghai Bench and Sunset Pumps on the Feather River.

Quality of the remaining spawning habitat is also of concern in terms of water flow and temperature in the Sacramento, Yuba, and Feather Rivers. Comparative analyses of historic and contemporary hydrologic and thermal regimes indicate that habitats in all of these rivers are different than they were before dam construction (NMFS 2015).

Flood bypass systems along the Sacramento River pose a challenge to Southern DPS Green Sturgeon during spawning migrations. Green Sturgeon are particularly affected at the Yolo and Sutter Bypasses and by Tisdale and Fremont Weirs (Thomas et al. 2013). Reclamation and DWR are working on the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project which will improve Sturgeon passage at the Yolo Bypass and Fremont Weir. Green Sturgeon mature late, live long, and do not reproduce every year, which makes the population susceptible to loss of even a small number of reproductive females. In long-lived species with delayed reproductive maturity, including Sturgeon, population growth rate is most sensitive to adult survival (Heppell 2007). Therefore, stranding of even a few reproductive individuals at flood control structures could have a large impact on the population.

Population impacts also arise from bycatch in fisheries, poaching, and small population size. Climate change could result in elevated water temperatures that would be detrimental to the reproductive success of the Green Sturgeon population if water temperatures in remaining spawning areas became elevated above that suitable for egg incubation and hatching success.

2.10.5 Water Operations Management

Reclamation does not currently manage for Green Sturgeon. However, many operational changes made for Chinook Salmon or Steelhead also benefit Green Sturgeon. Removal of the Red Bluff Diversion Dam in 2013 adds additional spawning habitat for Green Sturgeon in the upper Sacramento River.

2.10.6 Recovery and Management

Heublein et al. (2017) developed a conceptual model to support management and recovery of Sturgeon species in the San Francisco Estuary watershed. Additionally, NMFS issued a final recovery plan for the southern DPS of North American Green Sturgeon in 2018. Reduction of potential spawning habitat, severe threats to the single remaining spawning population coupled with the inability to alleviate these threats using current conservation measures, and the continued observance of declining numbers of juveniles collected in the past two decades threaten the species (NMFS 2015).

Fishing regulations and conservation measures represent a reduction in risk to Green Sturgeon. Recent implementation of Sturgeon fishing restrictions in Oregon and Washington and protective efforts put in place on the Klamath, Trinity, and Eel Rivers in the 1970s, 1980s, and 1990s may offer protection to the Southern DPS.

The retention of Green Sturgeon is prohibited along the west coast of North America. California also revised its regulations to provide additional protection for Green Sturgeon. Effective March 1, 2010, Sturgeon fishing was prohibited year-round in the mainstem Sacramento River from Highway 162 to Keswick Dam to protect spawning adults (NMFS 2015).

One of the most important conservation actions that has occurred in the last 10 years is the permanent removal of the gates of the Red Bluff Diversion Dam, where originally, the dam was closed year-round (NMFS 2015) and prevented Sturgeon passage. Further conservation efforts include floodplain and river restoration; riparian habitat protection; fish screening and passage projects; environmental water acquisitions; and contaminant studies conducted under the CVPIA, the Anadromous Fish Restoration Program, and the California Bay-Delta Program for the conservation of the southern DPS Green Sturgeon and other anadromous fishes.

Rescue of stranded individuals trapped behind weir structures can have a positive impact on this slow-reproducing species, but does not offer a viable long-term solution to maintaining the population. Thomas et al. (2013) present a modeling analysis indicating that rescue of the animals is important for population viability, but also note that fish passage improvement (rather than continued rescue) is a more appropriate long-term goal for mitigating this threat.

2.10.7 Monitoring and Research Programs

Research needs for Green Sturgeon in the Sacramento-San Joaquin Watershed are described in Klimley et al. (2015) and Heublein et al. (2017). Additionally, priority monitoring programs are described in the NMFS (Acipenser medirostris) (2018).

During July 2018, CDFW and University of California Davis began a 3-year monitoring study to investigate rearing and migratory behavior of Green Sturgeon in the lower Sacramento River (CDFW 2018). Results indicate that Green Sturgeon recruitment may be low during critically dry years.

Reclamation funds the USFWS to establish a Green Sturgeon monitoring program in the upper Sacramento River. Large numbers of larvae (n=4,881; greater than the long-average) and juveniles (n=26) were observed at the Red Bluff fish monitoring RSTs between May 28 and November 17, 2017. Eighty-five juveniles ranging in size from 72 to 322 mm mean = 176 mm) were collected in benthic trawl during 2017. Juvenile Salmon Acoustic Telemetry System (JSATS) acoustic tags were implanted into 45 of these fish. All juveniles were sampled between downtown Red Bluff and below Woodson Bridge over an approximately 60 river kilometer reach. A strong correlation was observed between movement and flow events (i.e., discharge or turbidity). Approximately 83 percent of Sturgeon movement detected during 2017 occurred from November 15 to November 22, 2017 while flows were increasing (Poytress personal communication 2017).

Sampling tools that are less invasive are especially useful for monitoring ESA-listed species. Environmental DNA (eDNA) is a quick, inexpensive method that could be used to efficiently monitor distribution of fish. Green Sturgeon have been identified using eDNA techniques in the Sacramento River (Bergman et al. 2016).

Research on physiological processes are important for informing temperature ranges for survival and targeting future restoration sites. Poletto et al. (2018) studied the effects of temperature and food availability on the growth of larval Green Sturgeon. The study indicated that larval sturgeon that reared and the greatest temperature tests exhibited optimal condition when fed optimally; however, when food was restricted larval sturgeon condition was the poorest at the greatest temperature tested (Poletto et al. 2018). Sardella and Kultz (2014) assessed Green Sturgeon ability to tolerate salinity fluctuations. They found that Sturgeon can acclimate to changes in salinity; however, these salinity fluctuations result in cellular stress (Sardella and Kultz 2014).

2.11 Green Sturgeon Critical Habitat

On October 9, 2009, NMFS designated critical habitat for the southern DPS of North American Green Sturgeon. In the Central Valley, critical habitat for Green Sturgeon includes the Sacramento River downstream of Keswick Dam, the Feather River downstream of Fish Barrier Dam, the Yuba River downstream of Daguerre Point Dam, a portion of the lower American River, the Sutter and Yolo bypasses, the Sacramento–San Joaquin Delta, and the San Francisco Estuary (74 FR 52300). Critical habitat also includes marine waters (out to the 60-fathom depth bathymetry line, relative to Mean Low Low Water) and several coastal bays and estuaries extending from Monterey Bay, California northward to the Strait of Juan de Fuca, Washington (74 FR 52300).

NMFS has outlined specific PBFs essential for the conservation of the Southern DPS in freshwater riverine systems, estuarine areas, and coastal marine areas (74 FR 52300):

Freshwater riverine systems:

1. Food resources—Abundant prey items for larval, juvenile, subadult, and adult life stages.
2. Substrate type or size - 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).

3. 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.
4. Water quality—Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.
5. Migratory corridor—A migratory pathway necessary for the safe and timely passage of Southern DPS fish 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).
6. Depth—Deep (≥5 meters) 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.
7. Sediment quality - Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages.

Estuarine habitats:

1. Food resources—Abundant prey items within estuarine habitats and substrates for juvenile, subadult, and adult life stages.
2. 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.
3. Water quality—Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages.
4. Migratory corridor—A migratory pathway necessary for the safe and timely passage of Southern DPS fish within estuarine habitats and between estuarine and riverine or marine habitats.
5. Depth—A diversity of depths necessary for shelter, foraging, and migration of juvenile, subadult, and adult life stages.
6. Sediment quality—Sediment quality (i.e., chemical characteristics) necessary for normal behavior, growth, and viability of all life stages.

Nearshore coastal marine areas:

1. Migratory corridor—A migratory pathway necessary for the safe and timely passage of Southern DPS fish within marine and between estuarine and marine habitats
2. 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.
3. Food resources—Abundant prey items for subadults and adults, which may include benthic invertebrates and fishes

2.12 Killer Whale, Southern Resident DPS

2.12.1 ESA Listing Status

The Southern Resident DPS of Killer Whales was listed as endangered under the Endangered Species Act on November 18, 2005 (70 FR 69903). Their range in the Northeastern Pacific Ocean overlaps with other whale populations classified as transient, resident, and offshore populations. The Southern Resident population consists of three pods designated J, K and L, each containing 22, 18 and 34 members, respectively (Center for Whale Research 2018). These pods generally spend late spring, summer and fall in inland waterways of Washington State and British Columbia. They are also known to travel as far south as central California and as far north as the Queen Charlotte Islands. Winter and early spring movements are largely unknown for this DPS.

On August 2, 2012, National Marine Fisheries Service (NMFS) received a petition to delist the Southern Resident Killer Whale DPS under ESA submitted by the Pacific Legal Foundation on behalf of the Center for Environmental Science Accuracy and Reliability, Empresas Del Bosque, and Coburn Ranch (Pacific Legal Foundation 2012). The petitioners claimed that there is no scientific basis for the designation of the unnamed North Pacific Resident subspecies of which the Southern Resident Killer Whales are a purported DPS. Therefore, because NMFS is without authority to list a DPS of a subspecies, the listing of the Southern Resident Killer Whale DPS is illegal and NMFS should delist the DPS.

On November 27, 2012, NMFS indicated that the petition to delist the DPS was warranted and they would initiate a status review to determine whether delisting is warranted and to examine the application of the DPS policy (77 FR 70733).

On August 5, 2013, NMFS determined that the delisting of Southern Resident Killer Whale DPS was not warranted because, after a legal and scientific review, they found that there was no new information leading to a different conclusion from that reached in the 2005 rulemaking, and the weight of evidence continues to support their conclusion that the North Pacific Resident Killer Whales represent a taxonomic subspecies (78 FR 47277).

On April 25, 2014, NMFS accepted a petition from the Center for Biological Diversity to review the critical habitat designation for the Southern Resident Killer Whale DPS. The petition requested that NMFS revise the critical habitat designation to include inhabited Pacific Ocean marine waters along the West Coast of the United States that constitute essential foraging and wintering areas (79 FR 22933).

On February 23, 2015, National Oceanic and Atmospheric Administration (NOAA) Fisheries announced a 12-month finding on a petition to revise the Critical Habitat Designation for the Southern Resident Killer Whale DPS was warranted (80 FR 9682).

2.12.2 General Life-History and Habitat Requirements

Wild female Southern Resident Killer Whales give birth to their first surviving calf between the ages of 12 and 16 years (mean = about 14.9 years) (Olesiuk et al. 1990; Matkin et al. 2003). Females produce an average of 5.4 surviving calves during a reproductive life span lasting about 25 years (Olesiuk et al. 1990). Males become sexually mature at body lengths ranging from 5.2-6.4 meters, which corresponds to between the ages of 10 to 17.5 years (mean = about 15 years) (Christensen 1984; Perrin and Reilly 1984; Olesiuk et al. 1990), and are presumed to remain sexually active throughout their adult lives (Olesiuk et al. 1990).

Most mating of Southern Resident Killer Whales in the North Pacific is believed to occur from May to October (Olesiuk et al. 1990); however, conceptions apparently happen year-round because births of calves are reported in all months. The mean interval between viable calves is 4 years (Bain 1990). Mothers and offspring maintain highly stable social bonds throughout their lives and this natal relationship is the basis for the matrilineal social structure in the Southern Resident population (Bigg et al. 1990; Baird and Whitehead 2000).

As the oceans' apex predator, Killer Whales feed on a great diversity of prey. More than 120 species of fishes, invertebrates, sea turtles, sea birds and marine mammals have been recorded in the species' diet (Ford and Ellis 2006). Most published information on Southern Resident Killer Whale prey originates from studies (Ford et al. 1998; Ford and Ellis 2005) in British Columbia, including southeastern Vancouver Island. These studies focused primarily on Northern Residents and included a relatively small number of observations for Southern Residents. Of the 487 records of apparent fish predation events from 1974 to2004, only 68 (14 percent) observations came from Southern Residents. The study recorded surface observations from predation events and also analyzed the stomach contents from stranded Killer Whales. Southern Resident Killer Whales are known to consume 22 species of fish and one species of squid (Ford et al. 1998, 2000; Saulitis et al. 2000; Ford and Ellis 2005). In recent years, additional data have been collected on Southern Resident Killer Whales in parts of Puget Sound (Hanson et al. 2010). In addition to collections of scales from observed predation events, fecal samples have also been collected for analysis.

Ford and Ellis (2005) found that salmon represent over 96 percent of the prey consumed during the spring, summer, and fall. Chinook Salmon were selected over other species, comprising over 70 percent of the identified salmonids taken. This preference occurred despite the much lower abundance of Chinook in the study area in comparison to other salmonids, and is probably related to the species' large size, high fat and energy content, and year-round occurrence in the area. Other salmonids eaten in smaller amounts include Chum (22 percent of the diet), pink (3 percent), Coho (2 percent), Sockeye (less than 1 percent), and Steelhead (less than 1 percent) (Ford and Ellis 2005). This work suggested an overall preference of these whales for Chinook during the summer and fall, but also revealed extensive feeding on Chum Salmon in the fall.

Ford et al. (2016) confirmed the importance of Chinook Salmon to Southern Residents in the summer months using DNA sequencing from whale feces. Ford et al. found that more than 90 percent of the whales' inferred diet consisted of salmonids; almost 80 percent was Chinook Salmon. Bellinger et al. (2015) estimated that Central Valley Chinook Salmon made up about 22 percent of the Chinook Salmon sampled off the Oregon coast and about 50 percent of those sampled off the California coast (south to Big Sur). While this apex predator certainly eats a variety of other species as well, Central Valley Chinook Salmon (all runs) can be estimated to make up approximately 40 percent of the Killer Whale diet when Killer Whales are off the California coast, and 18 percent of the Killer Whale diet when the Killer Whales are off the Oregon coast.

The Southern Resident population of Killer Whales is thought to move with the seasonal abundance of salmonids returning to natal rivers to spawn from early summer through fall. There are correlations between the occurrence of Southern Residents and commercial and sport salmon fishery catches in U.S. waters off southeastern Vancouver Island and in Puget Sound (Heimlich-Boran 1986). This population of Killer Whales is commonly found off southeastern Vancouver Island and in Puget Sound, Washington, from late spring to late fall (Ford 2006; Osborne 1999). The winter distribution of Southern Resident Killer Whales is poorly known. Several of the Southern Resident pods have been observed off the mouth of the Columbia River and in Monterey Bay, California, associated with local concentrations of Chinook Salmon (Wiles 2004; Balcomb 2006).

2.12.3 Historical and Current Distribution and Abundance

Southern Resident Killer Whales are found throughout the coastal waters off Washington, Oregon, and Vancouver Island, and are known to travel as far south as central California and as far north as the Queen Charlotte Islands, British Columbia (Figure 2.12-1). Southern Resident Killer Whales spend considerable time from late spring to early autumn in inland waterways of Washington and British Columbia, such as the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound (Bigg 1982; Ford et al. 2000; Krahn et al. 2002; table 4-10). Typically, J, K, and L Pods are increasingly present in May or June and spend considerable time in the core area of Georgia Basin and Puget Sound until at least September. During this time, the pods (particularly K and L) make frequent trips from inland waters to the outer coasts of Washington and southern Vancouver Island, which typically last a few days (Ford et al. 2000).

Southern Residents were formerly thought to range southward along the coast to about Grays Harbor (Bigg et al. 1990) or the mouth of the Columbia River (Ford et al. 2000). However, recent sightings of members of K and L Pods in Oregon and in California as far south as Monterey Bay, have considerably extended the southern limit of their known range (NMFS 2008b, 2014b, 2016). The historical abundance of Southern Residents was estimated based on genetic data to have ranged from 140 to 200 individuals (Krahn et al. 2002; NMFS 2008). The population was depleted by live captures for aquarium programs during the 1960s and 1970s (Balcomb et al. 1982; Olesiuk et al. 1990). Following a steep decline of 20 percent between 1996 and 2001 (from 97 whales to 78) (Krahn et al. 2002, 2004), the population was listed as endangered in the United States and Canada. The population rebounded to 98 whales by 2005 and was 82 whales as of September 2013 (Center for Whale Research 2013). As of July 1, 2018, the total population of Southern Resident Killer Whales was 74, with J Pod having 22 individuals, K Pod having 18, and L Pod having 34, representing the lowest population in 34 years (Center for Whale Research 2018). Because the population is small and the probability of quasi-extinction is sufficiently likely, NMFS (2008) has determined that representation from all three pods is necessary to meet biological criteria for Southern Resident Killer Whale downlisting and recovery.

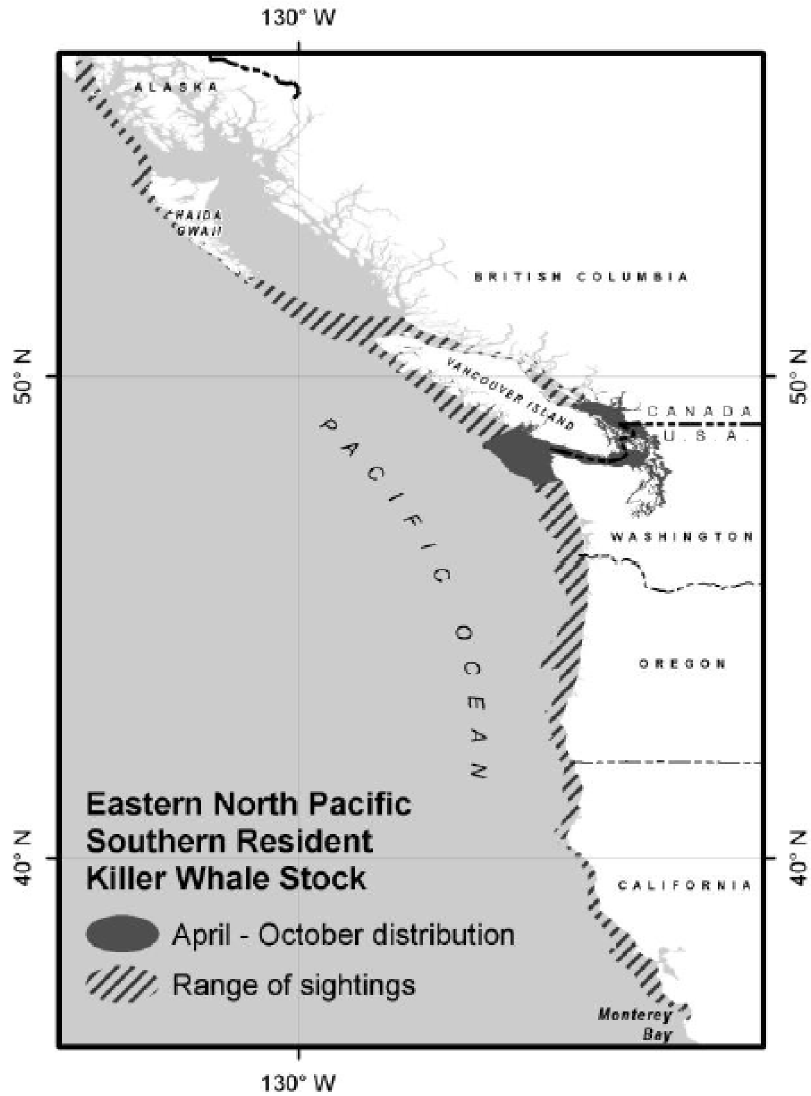

Figure 2.12-1. Southern Killer Whale Range off the Pacific West Coast

Most of the coastal sightings have occurred within 10 miles of shore (NMFS 2006), and there is no evidence that Southern Residents travel more than about 31 miles offshore (Ford et al. 2005). Although new evidence shows that Southern Residents range spans the coastal waters of Washington, Oregon, and California during the winter months (Ford 2013). A tracking study found that Southern Residents traveled extensively between Cape Flattery, Washington, and Point Reyes, California, from December 2012 to March 2013. Whales during this period generally confined their offshore movements to the continental shelf and slope, ranging to a maximum distance of 76 km offshore (Michael J. Ford, NMFS). Southern Resident Killer Whales live in highly cohesive matrilineal groups consisting of older females and one or two generations of their offspring of both sexes (Bigg et al. 1990; Baird and Whitehead 2000). Matrilineal groups tend to associate consistently in pods that travel together most of the time. Pods that share a common range and frequently associate form a community, for example, the Southern Resident community (Bigg et al. 1990; Parsons et al. 2009). The Southern Resident Killer Whale community is considered a single population. Different Southern Resident pods frequently intermingle for brief periods, likely for socialization and breeding (Hauser et al. 2007).

Increased sightings in the Strait of Juan de Fuca in late fall suggest that activity shifts to the outer coasts. Most sightings along the outer coast from 1975 to the present have been in the period of January through April. Given that Southern Resident Killer Whales occur during winter months as far south as Monterey Bay and that Central Valley Chinook Salmon compose a large percentage of the Chinook Salmon available south of the Columbia River, it is reasonable to expect that the whales could be affected by a change in the availability of Central Valley Chinook Salmon (Reclamation and DWR 2016).

2.12.4 Limiting Factors, Threats, and Stressors

As discussed in the original listing notice (70 FR 69903), the three main human-caused factors that have affected the Southern Resident Killer Whale population and may continue to impede the recovery of this species are contaminants, vessel traffic, and reductions in prey availability. Reductions in prey availability are caused by a large number of factors, including entrainment, predators, and climate change.

Exposure to contaminants may result in harm to the species. The presence of high levels of persistent organic pollutants, such as polychlorinated biphenyls (PCBs) and DDT, have been documented in Southern Resident Killer Whales (Ross et al. 2000; Ylitalo et al. 2001; and Herman et al. 2005). Many organochlorines are highly fat soluble (lipophilic) and accumulate in the fatty tissues of animals (Ross et al. 2000,). Some are highly persistent in the environment and resistant to metabolic degradation. These and other chemical compounds have the ability to induce immune suppression, impair reproduction, and produce other adverse physiological effects, as observed in studies of other marine mammals. High levels of “newly emerging” contaminants that may have similar negative effects, such as flame retardants, have been documented in Killer Whales, and are also becoming more prevalent in the marine environment (Rayne et al. 2004). Although contaminants enter marine waters and sediments from numerous sources, these chemical compounds enter Killer Whales through their prey. Because of their long life span, position at the top of the food chain, and their blubber stores, Killer Whales are capable of accumulating high concentrations of contaminants (Ylitalo et al. 2001; Grant and Ross 2002).

Commercial shipping, whale watching, ferry operations, and recreational boat traffic have increased in recent decades. Several studies have linked vessels with short-term behavioral changes in Northern and Southern Resident Killer Whales (Kruse 1991; Williams et al. 2002; Foote et al. 2004). Although the potential impacts from vessels and the sounds they generate are poorly understood, these activities may affect foraging efficiency, communication, and/or energy expenditure through their physical presence, increased underwater sound level, or both. Collisions with vessels are another potential source of serious injury and mortality and have been recorded, although rarely, for both Southern and Northern Resident Killer Whales.

Healthy Killer Whale populations are dependent on adequate prey levels. Reductions in prey availability may force whales to spend more time foraging and might lead to reduced reproductive rates and higher mortality rates. The Southern Resident Killer Whale prey base is composed primarily of salmonids, particularly Chinook Salmon between late spring and early fall (Ford and Ellis 2005; Hanson et al. 2010). Salmon populations available as prey to Southern Resident Killer Whale have declined because of degradation in aquatic ecosystems from modern land use changes, harvest, and hatchery practices, and 27 ESUs of Salmon and Steelhead in Washington, Oregon, Idaho, and California have been listed as threatened or endangered under the ESA (NMFS 2008). Reductions in prey availability may increase the amount of time whales must spend foraging, reduce reproductive output, and lead to higher mortality.

Chinook Salmon stocks that are important to Southern Resident Killer Whales have been identified by NOAA Fisheries and Washington Department of Fish and Wildlife. A framework was developed by including three factors that contribute to the identification of priority Chinook Salmon populations: (1)

observed part of Southern Resident Killer Whale diet, (2) consumed during reduced body condition or diversified Southern Resident Killer Whale diet, and (3) degree of spatial and temporal overlap (NOAA/WDFW 2018). These three factors were evaluated and scored (zero to five) to develop the priority list of Chinook Salmon populations (see details in Chapter 5, Effects Analysis).

2.12.5 Water Operations Management

While neither the CVP nor SWP operate for Killer Whales, occasional modifications to operations are made for Fall-Run Chinook Salmon, the commercial fishery of importance as prey to Southern Resident Killer Whales in California. For example, when cold water pool considerations are minimal, Reclamation keeps Keswick Dam releases high in the fall to avoid dewatering the last Fall-Run Chinook Salmon redds.

2.12.6 Recovery and Management

The final *Recovery Plan for Southern Resident Killer Whales (*Orcinus orca*)* was issued in January 2008 (NMFS 2008). The ultimate goal of the recovery plan is to achieve the recovery of the Southern Resident Killer Whale DPS and its ecosystem to a level sufficient to warrant its removal from the Federal List of Endangered and Threatened Wildlife and Plants under the ESA. The intermediate goal is to reclassify the DPS from endangered to threatened. The recovery plan also provides a recovery program that includes a set of specific management, research, and monitoring actions intended to reduce threats and restore the population to long-term sustainability.

Because NMFS determined that the Southern Resident stock was below its optimum sustainable population, the DPS was designated as “depleted” under the Marine Mammal Protection Act (MMPA) in 2003 (68 FR 31980) and a Proposed Conservation Plan was announced in 2005 (70 FR 57565). The plan provides a strategy to conserve and restore Southern Resident Killer Whales so that they are no longer considered depleted under the MMPA.

The following conservation measures were included in the final recovery plan for the DPS’s ESA listing (NMFS 2008):

- Support salmon restoration efforts in the region including habitat, harvest, and hatchery management considerations and continued use of existing NMFS authorities under the ESA and Magnuson-Stevens Fishery Conservation and Management Act to ensure an adequate prey base.
- Clean up existing contaminated sites, minimize continuing inputs of contaminants harmful to Killer Whales, and monitor emerging contaminants.
- Continue evaluating and improving guidelines for vessel activity near Southern Residents and evaluate the need for regulations or protected areas.
- Prevent oil spills and improve response preparation to minimize effects on Southern Residents and their habitat in the event of a spill.
- Continue agency coordination and use of existing MMPA mechanisms to minimize potential impacts from anthropogenic sound.
- Enhance public awareness, educate the public on actions they can participate in to conserve Killer Whales, and improve reporting of Southern Resident sightings and strandings.
- Improve responses to live and dead Killer Whales to implement rescues, conduct health assessments, and determine causes of death to learn more about threats and guide overall conservation efforts.

- Coordinate monitoring, research, enforcement, and complementary recovery planning with international, federal, and state partners.
- Conduct research to facilitate and enhance conservation efforts. Continue the annual census to monitor trends in the population, identify individual animals, and track demographic parameters

2.12.7 Monitoring and Research Programs

Many programs are conduct monitoring and research with Southern Resident Killer Whales. These are a few of the programs:

- NMFS—Northwest Fisheries Science Center—Marine Mammal Ecology Team conducts research to understand the factors that may limit this population including studies about their taxonomy, behavior, ecology, health, and human-caused impacts. (https://www.westcoast.fisheries.noaa.gov/protected_species/marine_mammals/killer_whale/rpi_monitoring_research.html)
- The Center for Whale Research has been collecting detailed demographic data on the Southern Resident Killer Whale population, recording all observed births and deaths and also conducted an aerial observation study (Center for Whale Research 2018)
- The National Marine Fisheries Service Southwest Fisheries Science Center and SR3: SeaLife Response, Rehabilitation and Research utilize photogrammetry to make quantitative measurements (particularly aerial photographs) of length, growth and body condition/nutritional status and to describe the abundance and structure of pods (https://www.westcoast.fisheries.noaa.gov/stories/2018/25_09252018_give_whales_space.html)
- The University of Washington—Center for Conservation Biology conducts research by acquiring scat samples to extract and assay DNA stress, reproductive and nutritional hormones, as well as toxicants (http://conservationbiology.uw.edu/research-programs/killer-whales/)
- The National Fish and Wildlife Foundation - Killer Whale Research and Conservation Program supports projects to help study and protect Killer Whales in the wild (https://www.nfwf.org/killerwhales/Pages/home.aspx)

2.13 Killer Whale Critical Habitat

Critical habitat for the Southern Resident DPS was designated under the Endangered Species Act on November 29, 2006 (71 FR 69054). The critical habitat designation encompasses the Summer Core Area in Haro Strait and the waters around the San Juan Islands, the Strait of Juan de Fuca and all of Puget Sound (Figure 2.13-1), but does not include any areas in California.

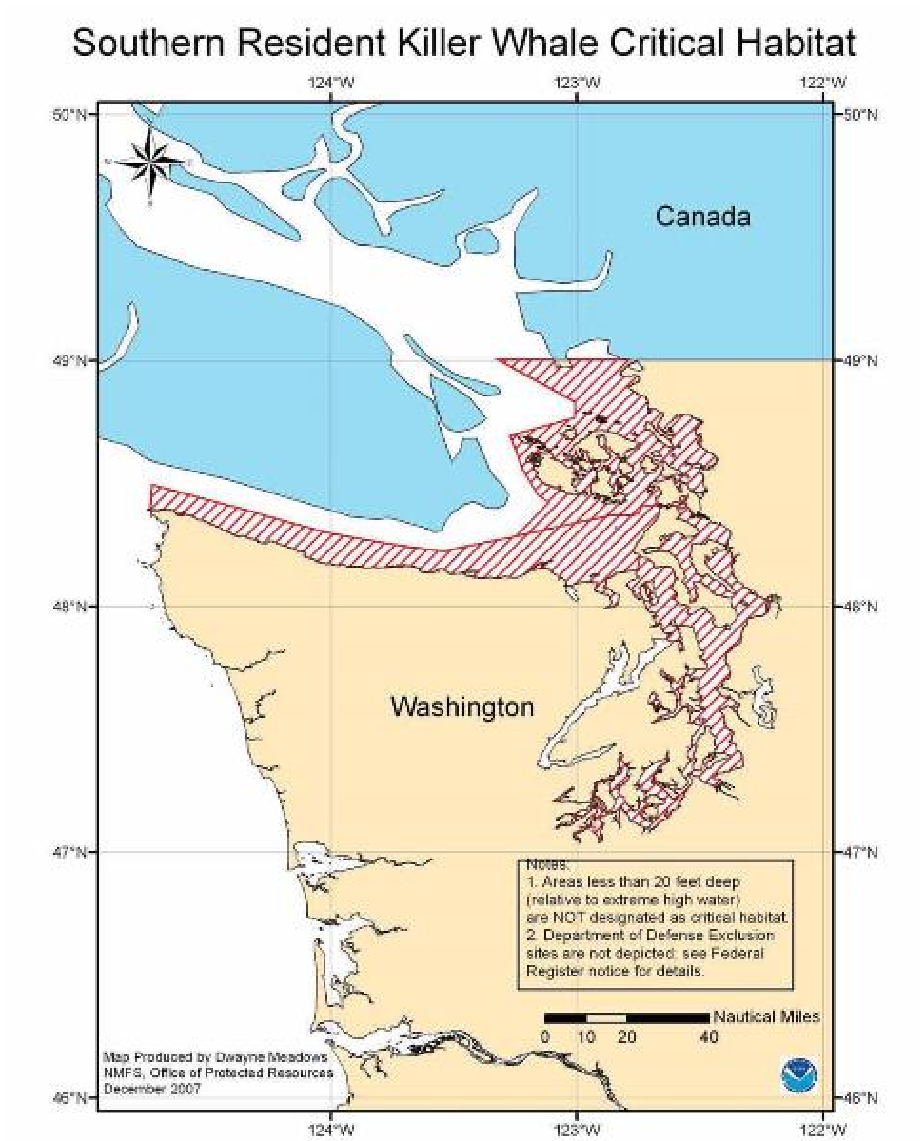

Figure 2.13-1. Southern Killer Whale Designated Critical Habitat

The critical habitat designation identified the following primary constituent elements considered essential for the conservation of the ESU.

1. Water quality to support growth and development;
2. Prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and
3. Passage conditions to allow for migration, resting, and foraging.

The Center for Biological Diversity proposes that the critical habitat designation be revised and expanded to include the addition of the Pacific Ocean region between Cape Flattery, WA, and Point Reyes, CA, extending approximately 47 miles (76 km) offshore. Based on new information, NOAA Fisheries intends to proceed with the petitioned action to revise critical habitat for Southern Resident Killer Whales (80 FR 9682).

2.14 Delta Smelt

2.14.1 ESA Listing

The USFWS proposed to list the Delta Smelt (*Hypomesus transpacificus*) as threatened with proposed critical habitat on October 3, 1991 (USFWS 1991). The USFWS listed the Delta Smelt as threatened on March 5, 1993 (USFWS 1993), and designated critical habitat for the species on December 19, 1994 (USFWS 1994). The Delta Smelt was one of eight fish species addressed in the *Recovery Plan for the Sacramento–San Joaquin Delta Native Fishes* (USFWS 1996). A 5-year status review of the Delta Smelt was completed on March 31, 2004 (USFWS 2004). The 2004 review concluded that Delta Smelt remained a threatened species. A subsequent 5-year status review recommended uplisting Delta Smelt from threatened to endangered status (USFWS 2010a). A 12-month finding on a petition to reclassify the Delta Smelt as an endangered species was completed on April 7, 2010 (USFWS 2010b). After reviewing all available scientific and commercial information, the USFWS determined that reclassifying the Delta Smelt from threatened to endangered was warranted but was precluded by other higher priority listing actions (USFWS 2010c). The USFWS annually reviews the status and uplisting recommendation for Delta Smelt during its Candidate Notice of Review (CNOR) process. Each year, the CNOR has recommended the uplisting from threatened to endangered. Electronic copies of these documents are available at http://ecos.fws.gov/docs/five_year_review/doc3570.pdf and http://www.gpo.gov/fdsys/pkg/FR-2013-11-22/pdf/2013-27391.pdf (USFWS 2010a; USFWS 2010b; USFWS 2012b).

2.14.2 General Life History and Habitat Requirements

The Delta Smelt is endemic to the San Francisco Bay–Delta where it primarily occupies open-water habitats in Suisun Bay and marsh and the Sacramento–San Joaquin Delta. The Delta Smelt is primarily an annual species, meaning that it completes its life cycle in 1 year, which typically occurs from April to the following April plus or minus 1 or 2 months. In captivity, Delta Smelt can survive to spawn at 2 years of age (Lindberg et al. 2013), but this appears to be rare in the wild (Bennett 2005; Damon et al. 2016), where very few individuals reach lengths over 3.5 inches (90 mm).

Delta Smelt spawning likely occurs at night with several males attending a female that broadcasts her eggs onto bottom substrate (Bennett 2005). Although preferred spawning substrate is unknown, spawning habits of Delta Smelt's closest relative, the Surf Smelt (*Hypomesus pretiosus*), as well as unpublished experimental trials, suggest that sand or small pebbles may be the preferred substrate (Bennett 2005). Hatching success peaks at temperatures of 15°C to 16°C (59°F to 61°F) and decreases at cooler and warmer temperatures. Hatching success nears 0 percent as water temperatures exceed 20°C (68°F) (Bennett 2005). Water temperatures suitable for spawning occur most frequently during the months of March to May, but ripe female Delta Smelt have been observed as early as January and larvae have been collected as late as July. Delta Smelt spawn in the estuary and have one spawning season for each generation, which makes the timing and duration of the spawning season important every year. Freshwater flow affects how much of the estuary is available for Delta Smelt to spawn (Hobbs et al. 2007), but water temperature controls how long Delta Smelt can spawn each season.

Although adult Delta Smelt can spawn more than once, mortality is high during the spawning season and most adults die by May (Polansky et al. 2018). The egg stage averages about 10 days before the embryos hatch into larvae. The larval stage averages about 30 days. Metamorphosing "post-larvae" appear in monitoring surveys from April into July of most years. By July, most Delta Smelt have reached the juvenile life stage. Delta Smelt collected during the fall are called "subadults," a stage which lasts until winter when fish disperse toward spawning habitats. This winter dispersal usually precedes sexual maturity (Sommer et al. 2011).

Most Delta Smelt complete their entire life cycle within or immediately upstream of the estuary's low-salinity zone. The low-salinity zone is frequently defined as waters with a salinity range of about 0.5 to 6 parts per thousand (ppt) (Kimmerer 2004). The 0.5 to 6 ppt and similar salinity ranges reported by various authors were chosen based on analyses of historical peaks in phytoplankton and zooplankton abundance, but recent physiological and molecular biological research has indicated that the salinities typical of the low-salinity zone are also within the tolerance range (0 to 18 ppt) for Delta Smelt (Komoroske et al. 2016). Komoroske et al. (2016) also found that acclimating to salinities outside the low-salinity zone (LSZ) could impose energetic costs that constrain the species' ability to exploit these habitats. The low-salinity zone is a dynamic habitat with size and location responding rapidly to changes in tidal and river flows. By local convention the location of the low-salinity zone is described as "X2" in terms of the distance from the 2 ppt isohaline to the Golden Gate Bridge. The low-salinity zone magnitude and dimensions change when river flows into the estuary are high, placing low-salinity water over a larger and more diverse set of nominal habitat types than occurs under low flow conditions. During periods of low outflow, the low-salinity zone contracts and moves upstream.

Delta Smelt mainly occupy an arc of habitat in the north Delta, including Liberty Island and the adjacent reach of the Sacramento Deepwater Shipping Channel (Sommer and Mejia 2013), Cache Slough to its confluence with the Sacramento River, and the Sacramento River from that confluence downstream to Chipps Island, Honker Bay, and the eastern part of Montezuma Slough (see Figure 2.14-5). The reasons Delta Smelt are believed to permanently occupy this part of the estuary are the year-round presence of fresh- to low-salinity water that is comparatively turbid and of a tolerable water temperature. These appropriate water quality conditions overlap an underwater landscape featuring variation in depth, tidal current velocities, edge habitats, and food production (Sweetnam 1999; Nobriga et al. 2008; Feyrer et al. 2011; Murphy and Hamilton 2013; Hammock et al. 2015; Bever et al. 2016). Field observations are increasingly supported by laboratory research on the physiological response of Delta Smelt to variation in salinity, turbidity, water temperature, and environmental variables associated with changes in climate, freshwater flow, and estuarine bathymetry (Hasenbein et al. 2014, 2016; Komoroske et al. 2014, 2016).

2.14.3 Historical and Current Distribution

The 2018 (WY 2019) CDFW Fall Midwater Trawl (FMWT) Index was zero, the lowest on record. The CDFW Spring Kodiak Trawl (SKT) monitors the adult spawning stock of Delta Smelt and serves as an indication for the relative number and distribution of spawners in the system. The 2018 SKT Abundance Index was 2.1, the second lowest on record. All CDFW relative abundance indices show a declining trend since the early 2000s (Figure 2.14-1).

In 2016, the USFWS began calculating an absolute abundance estimate using January and February SKT catch data, which have been available since 2002. This calculation was modified in 2017 and resulting estimates and ranges are shown in Figure 2.14-1.

The 2018 absolute abundance estimate is the second lowest; however the confidence intervals overlap so strongly that it cannot be stated that 2018 actually had higher adult abundance than 2016. The January through February 2016 point estimates are the lowest since the SKT survey began in 2002 and suggest Delta Smelt experienced increased natural mortality during the extreme drought conditions occurring during 2013–2015. While the estimate may have increased slightly in 2017, it appears to have decreased again in 2018. The continued low spawning stock of Delta Smelt relative to historical numbers suggests the population would continue to be vulnerable to stochastic events and operational changes occurring in response until successive years of increased population growth results in a substantial increase in abundance.

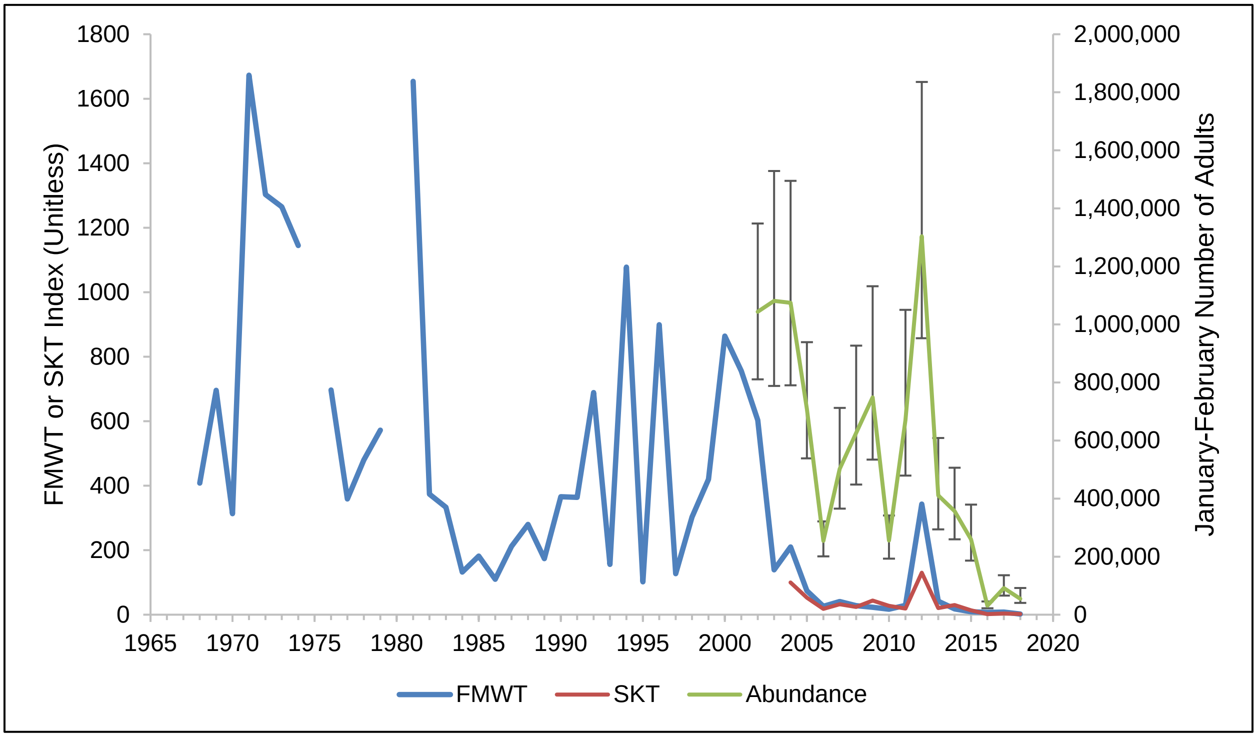

Figure 2.14-1. Fall Midwater Trawl Index, Spring Kodiak Trawl Index, and January-February Spring Kodiak Trawl Abundance Estimate (With 95% Confidence Interval), Water Years 2002-2018.

In addition to these abundance estimates, the CDFW conducts four fish surveys from which it develops indices of Delta Smelt's relative abundance. Each survey has variable capture efficiency (Mitchell et al. 2017), and in each, the frequency of zero catches of Delta Smelt is very high, largely due to the species' rarity (Latour 2016; Polansky et al. 2018).

The Townet Survey (TNS) and FMWT abundance indices for Delta Smelt have documented the species' long-term decline, while the newer 20-mm and SKT abundance indices have generally confirmed the recent portions of the trends implied by the older surveys (Figures 2.14-2 and 2.14-3). During the period of record, Delta Smelt relative abundance has declined from peak levels observed during the 1970s. The TNS and FMWT abundance indices both declined rapidly during the early 1980s, increased somewhat during the 1990s, and then collapsed in the early 2000s. Since 2005, the TNS and the FMWT have produced indices that reflect less year to year variation than their 20-mm and SKT analogs, but overall, the trends in both sets of indices are similar. During the past decade, the index has continued to decrease and currently reflects 0.5 percent of the relative abundance recorded in 1970–1971.

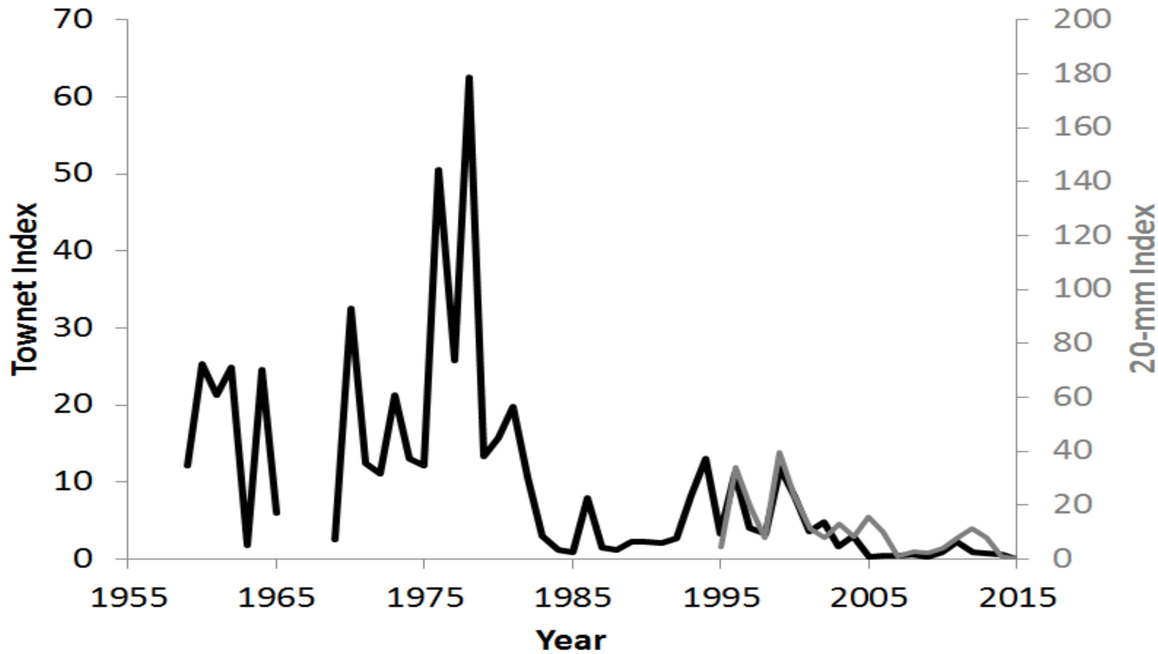

Figure 2.14-2. Time Series of the CDFW's Summer TNS (black line; primary y-axis) and 20-mm Survey (gray line; secondary y-axis) Abundance Indices for Delta Smelt

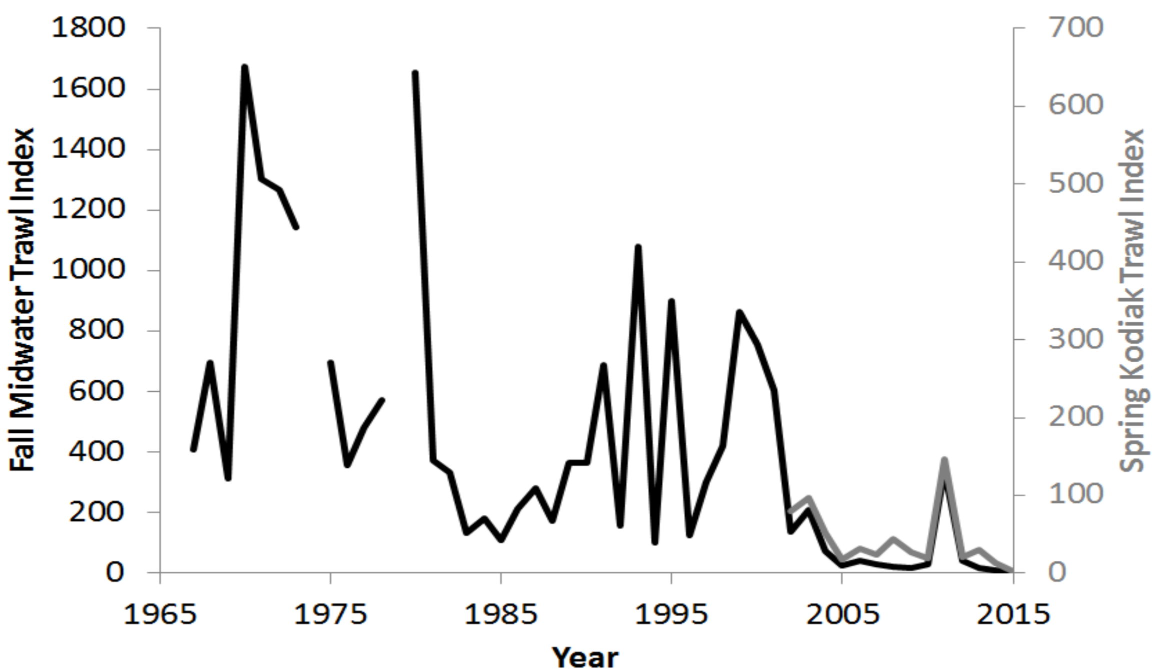

Figure 2.14-3. Time Series of the CDFW's FMWT (black line; primary y-axis) and SKT (gray line; secondary y-axis) Abundance Indices for Delta Smelt

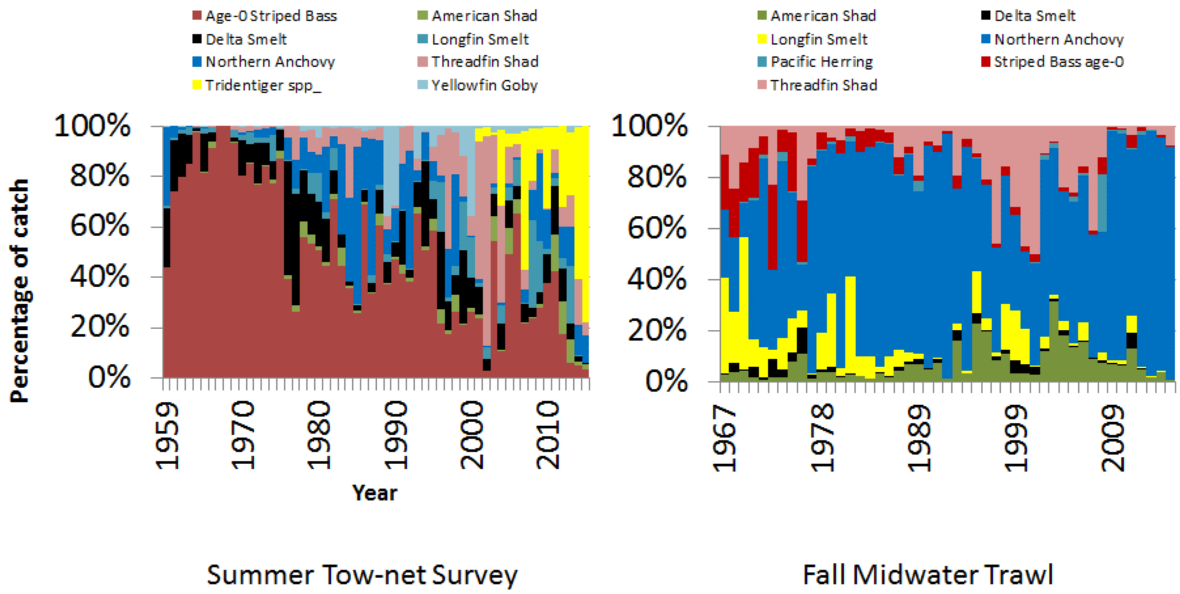

Figure 2.14-4. Fractional Compositions of the Eight Most Frequently Collected Fish Species in the CDFW's Summer TNS (1959–2015), and the Seven Most Frequently Collected Fish Species in the FMWT (1967–2015)

The general distribution of Delta Smelt is well known partly due to its limited geographic distribution (Moyle et al. 1992; Bennett 2005; Hobbs et al. 2006, 2007; Feyrer et al. 2007; Nobriga et al. 2008; Kimmerer et al. 2009; Merz et al. 2011; Murphy and Hamilton 2013; Sommer and Mejia 2013). The suitable habitat for Delta Smelt is a geographically limited area (e.g., one example is Sacramento River around Decker Island) and has low-salinity conditions. The additional seasonally suitable habitats utilized for spawning and migration are identified as occasional seasonal use habitats. Distribution extremes do not yield Delta Smelt in most sampling years.

Delta Smelt have been observed as far west as San Francisco Bay, as far north as Knights Landing on the Sacramento River, as far east as Woodbridge on the Mokelumne River and Stockton on the Calaveras River, and as far south as Mossdale on the San Joaquin River (Figure 2.14-5). This distribution represents a range of salinity from essentially 0 ppt to about 20 ppt, which includes brackish water exceeding 2 ppt salinity. However, most Delta Smelt that have been collected in the extensively surveyed San Francisco Estuary have been collected from locations within the defined ranges of the critical habitat rule. In addition, all habitats known to be occupied year-around by Delta Smelt occur within the conditions defined in the critical habitat rule.

Each year, the distribution of Delta Smelt seasonally expands when adults disperse in response to winter flow increases, increases in turbidity, and decreases in water temperature (Figure 2.13-1). The annual range expansion of adult Delta Smelt extends up the Sacramento River to about Garcia Bend in the Pocket neighborhood of Sacramento, up the San Joaquin River from Antioch to areas near Stockton, up the lower Mokelumne River system, and west throughout Suisun Bay and Suisun Marsh. Some Delta Smelt seasonally and transiently occupy Old and Middle River in the south Delta each year, but face a high risk of entrainment when they do (Grimaldo et al. 2009).

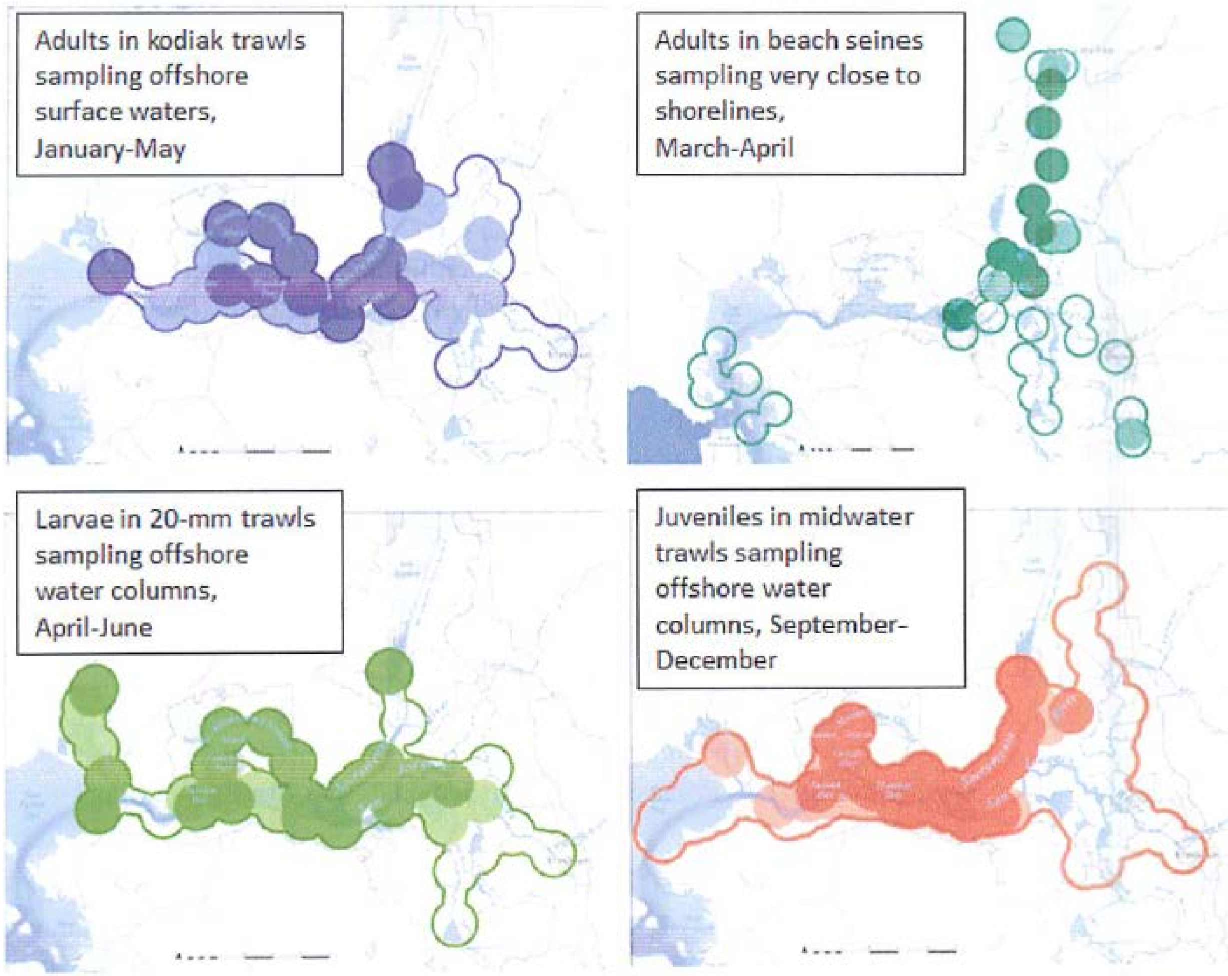

Figure 2.14-5. Maps of Multi-Year Average Distributions of Delta Smelt Collected in Four Monitoring Programs

The sampling regions covered by each survey are outlined. The areas with dark shading surround sampling stations in which 90 percent of the Delta Smelt collections occurred, the areas with light shading surround sampling stations in which the next 9 percent of Delta Smelt collections occurred (Murphy and Hamilton 2013).

The distribution of Delta Smelt occasionally expands beyond this area (Figure 2.14-5). For instance, during high outflow winters, adult Delta Smelt also disperse west into San Pablo Bay and up into the Napa River (Hobbs et al. 2007). Similarly, Delta Smelt have occasionally been reported from the Sacramento River north of Garcia Bend up to Knights Landing (Merz et al. 2011; Vincik and Julienne 2012).

The relative abundance of Delta Smelt has declined substantially for a small forage fish in an ecosystem the size of the San Francisco Estuary. The recent relative abundance reflects decades of habitat change and marginalization by nonnative species that prey on and outcompete Delta Smelt. The anticipated effects of climate change on the San Francisco Estuary and watershed, such as warmer water temperatures, greater salinity intrusion, lower snowpack contribution to spring outflows from the Delta, and the potential for frequent extreme drought, indicate challenges to maintaining a sustainable Delta

Smelt population. A rebound in relative abundance during the very wet and cool conditions during 2011 indicated that Delta Smelt retained some population resilience (IEP 2015). However, since 2012, declines to record low population estimates have been broadly associated with the 2012–2015 drought, and wetter conditions in 2017 and 2018 have not produced a rebound similar that seen in 2011.

2.14.4 Limiting Factors, Threats, and Stressors

Limiting factors for Delta Smelt are: SWP and CVP exports due to entrainment of larvae and juveniles and the effects of low flow on the location and function of the estuary mixing zone (now called the low-salinity zone) (Moyle et al. 1992; USFWS 1993; USFWS 2010); drought; in-Delta water diversions; reduction in food supplies by nonindigenous aquatic species, specifically overbite clam and nonnative copepods; toxicity due to agricultural and industrial chemicals; increasing water transparency; and Brazilian waterweed (*Egeria densa*). Predation was considered a low-level threat linked to increasing waterweed abundance and increasing water transparency. Additional threats considered potentially significant by the USFWS in 2010 were entrainment into power plant diversions, contaminants, and reproductive problems that can stem from small population sizes.

The long-term rarity of the Delta Smelt has had a consequence for understanding the reasons for their population decline, which adds challenges for implementing effective resource management strategies. Some pelagic fishes have shown long-term relationships between Delta inflow, Delta outflow, or X2 and abundance or survival (Stevens and Miller 1983; Jassby et al. 1995; Kimmerer 2002b; Kimmerer et al. 2009). A predictive correlation between freshwater flow conditions and relative abundance has not been established (Stevens and Miller 1983; Jassby et al. 1995; Kimmerer 2002b; Kimmerer et al. 2009). Since 2010, several conceptual models (Interagency Ecological Program [IEP] 2015) and empirical models (Thomson et al. 2010; Maunder and Deriso 2011; Miller et al. 2012; Rose et al. 2013a; Hamilton and Murphy 2018) have explored life cycle models for the Delta Smelt in an attempt to describe the reasons for the population decline. Some of these models have recreated the trend observed in abundance indices (Figure 2.14-6), but each model has applied different methodology and variables. Collectively, these modeling efforts generally support water temperature and changes in the estuary's trophic dynamics as "universally supported" factors affecting Delta Smelt. However, they have varying conclusions regarding the effectiveness of alterations in water operations as management strategies to increase the likelihood of population increases for Delta Smelt.

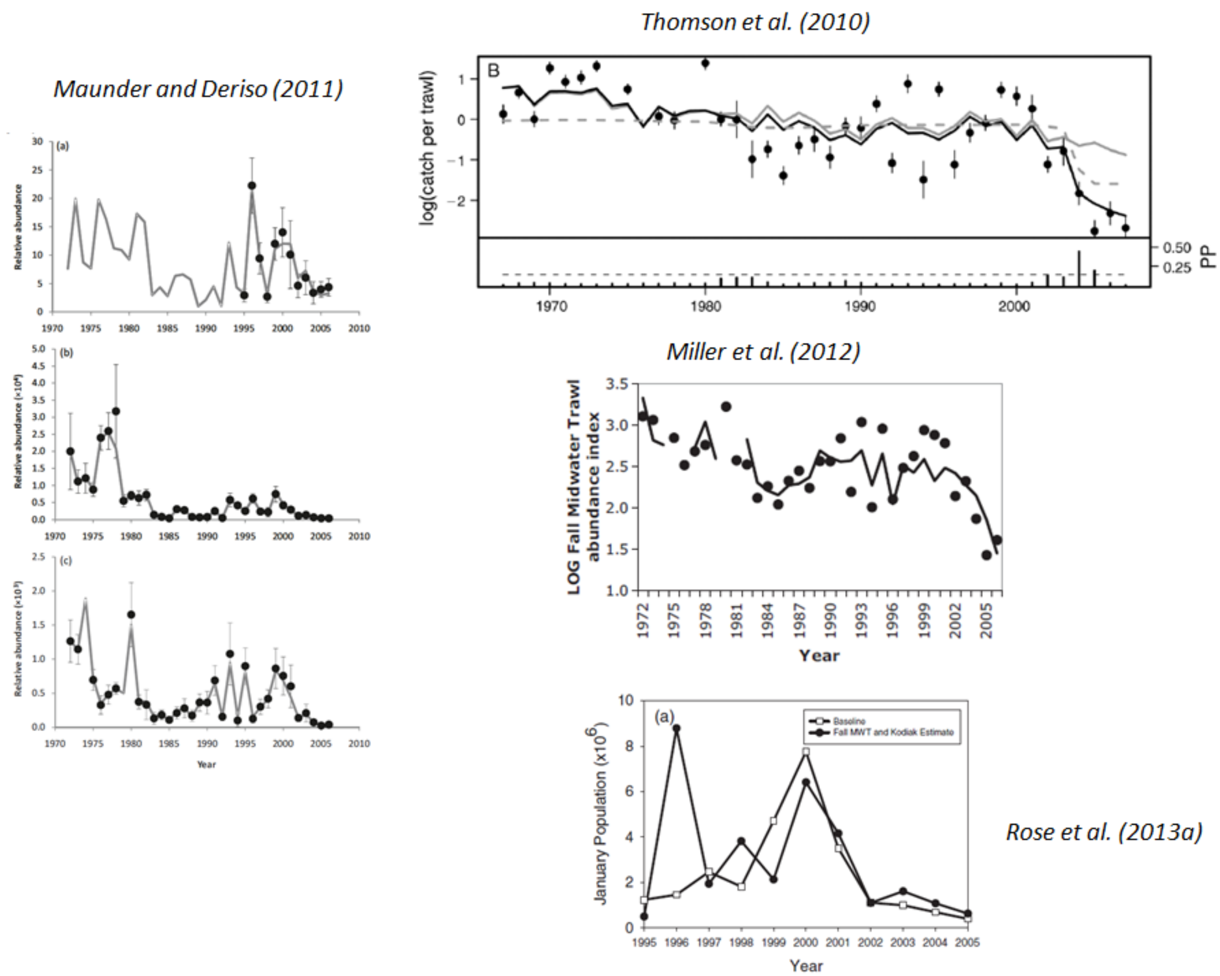

Figure 2.14-6. Examples of Recent Published Model Fits to Time Series of Delta Smelt Relative Abundance Data

The source of each is referenced above or alongside each time series. In each plot, observed catches are depicted as black dots and model predictions of the data as gray or black lines. Model predictions from Rose et al. (2013a) are a black line with open symbols. In Maunder and Deriso (2011), the three panels represent the 20-mm Survey, summer TNS, and FMWT Survey from top to bottom, respectively. The other three studies are fit to estimates of adult Delta Smelt relative abundance (Thomson et al. 2010; Miller et al. 2012) or absolute abundance (Rose et al. 2013a). See each study for further details on methods, results, and the authors' interpretations of their results.

The ecological function of the low-salinity zone can vary depending on Delta outflow (Jassby et al. 1995; Kimmerer 2002a; Kimmerer 2004). During the past four decades, the low-salinity zone ecosystem has undergone substantial changes in turbidity (Schoellhamer 2011) and foodweb function (Winder and Jassby 2011) that cannot be undone solely by increasing Delta outflow. These habitat changes, which extend into parts of the Delta where water is fresher than 0.5 ppt, are hypothesized to have also decreased the ability of the low-salinity zone and adjacent habitats to support the production of Delta Smelt (Thomson et al. 2010; Rose et al. 2013b; IEP 2015).

At all life stages, numerous small planktonic crustaceans, especially a group called calanoid copepods, make up most of the Delta Smelt diet (Nobriga 2002; Slater and Baxter 2014). Small crustaceans are ubiquitously distributed throughout the estuary, but the prey species present at particular times and locations has changed dramatically over time (Winder and Jassby 2011; Kratina et al. 2014). This has likely affected Delta Smelt feeding success, particularly during central California's warm summers.

Climate projections for the San Francisco Bay–Delta and its watershed indicate that temperature and precipitation changes would reduce snowpack in the Sierra Nevada, changing the timing and availability of natural water supplies (Knowles and Cayan 2002; Dettinger 2005). Temperature increases may result in more precipitation falling as rain and less water stored in spring snowpacks. Increased frequency of rain-on-snow events would increase winter runoff and an associated decrease in runoff for the remainder of the year (Hayhoe et al. 2004). Overall, these and other storm track changes may lead to increased frequency of flood and drought cycles during the 21st century (Dettinger et al. 2015).

Central California's warm summers are already a source of energetic stress for Delta Smelt and warm springs already compress the duration of their spawning season (Rose et al. 2013a, 2013b; Moyle et al. 2016). Central California's climate is anticipated to get warmer (Cayan et al. 2009). Warmer estuary temperatures to present a significant conservation challenge for Delta Smelt (Brown et al. 2013, 2016). Mean annual water temperatures within the Delta are expected to increase steadily during the second half of this century (Cloern et al. 2011). Warmer water temperatures could further reduce Delta Smelt spawning opportunities, decrease juvenile growth during the warmest months, and increase mortality via several foodweb pathways including: increased vulnerability to predators, increased vulnerability to toxins, and decreased capacity for Delta Smelt to successfully compete in an estuary that is energetically more optimal for warm-water tolerant fishes.

2.14.5 Water Operations Management

Currently, in addition to D-1641, Reclamation operates to reduce entrainment risk and for Delta Smelt fall habitat in wet and above normal water years through releases of water from storage for Fall X2.

2.14.6 Recovery and Management

The USFWS (2010) recommended the following conservation actions: establish Delta outflows proportionate to proposedproposed action flows to set outflow targets as fractions of runoff in the Central Valley watersheds; minimize net reverse flow in Old and Middle River; and, establish a genetic management plan with the goals of minimizing the loss of genetic diversity and limiting risk of extinction caused by unpredictable catastrophic events. The USFWS (2012b) added climate change to the list of threats to the Delta Smelt.

Continued protection of the Delta Smelt from excessive entrainment, improving the estuary's flow regime, suppressing nonnative species, increasing zooplankton abundance, and improving water quality are among the actions recommended to aid the recovery of Delta Smelt (USFWS 2010).

2.14.7 Monitoring and Research

The Enhanced Delta Smelt Monitoring (EDSM) program began in November 2016 to acquire finer temporal resolution information than existing surveys provided about the spatial distribution and abundance of Delta Smelt. EDSM is a year-round weekly sampling program that samples randomly selected locations using a probabilistic procedure aimed at providing a spatially dispersed sample. This is a significant improvement on existing surveys, which sample in the same locations again and again, and may find no fish. EDSM sampling is repeated until a fish is caught or an upper limit on the number of tows is reached. EDSM methodology attempts to lower the probability of a "False Zero," that is, failing to catch fish when fish are present, while aiming to minimize the "take" of a threatened species. EDSM is the only survey that allowed agencies to measure where Delta Smelt are located in 2018, given their increasingly low abundance.

2.15 Delta Smelt Critical Habitat

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

The primary objective in designating critical habitat was to identify the key components of Delta Smelt habitat that support successful completion of the life cycle, including spawning, larval and juvenile transport, rearing, and adult migration back to spawning sites. Delta Smelt are endemic to the San Francisco Bay/Sacramento–San Joaquin Delta (Bay-Delta) and the vast majority only live 1 year. Thus, regardless of annual hydrology, the estuary must provide suitable habitat all year, every year. The primary constituent elements essential to the conservation of the Delta Smelt are physical habitat, water, river flow, and salinity concentrations required to maintain Delta Smelt habitat for spawning, larval and juvenile transport, rearing, and adult migration (USFWS 1994). The USFWS recommended in its designation of critical habitat for the Delta Smelt that salinity in Suisun Bay should vary according to water year type. For the months of February through June, this element was codified by the State Water Resources Control Board's "X2 standard" described in D-1641 and the Board's current Water Quality Control Plan.

Table 2.15-1 compares the original descriptions of the primary constituent elements with current scientific understanding.

Table 2.15-1. Comparison of Delta Smelt Primary Constituent Elements of Critical Habitat between the 1994 Publication of the Rule and the Present

Primary Constituent Element	1994 Critical Habitat Rule	2018 State of Scientific Understanding
Spawning Habitat	Shallow fresh or slightly brackish edgewaters.	No change.
	Backwater sloughs.	Possible, never confirmed. Most likely spawning sites have sandy substrates and need not occur in sloughs. Backwater sloughs in particular tend to have silty substrates that would suffocate eggs.
	Low concentrations of pollutants.	No change.
	Submerged tree roots, branches, emergent vegetation (tules).	Not likely. Unpublished observations of spawning by captive Delta Smelt suggest spawning on substrates

Primary Constituent Element	1994 Critical Habitat Rule	2018 State of Scientific Understanding
		oriented horizontally and a preference for gravel or sand that is more consistent with observations of other osmerid fishes. (Bennett 2005, p.17).
	Suspected spawning locations: Sacramento River "in the Delta," Barker Slough, Lindsey Slough, Cache Slough, Prospect Slough, Georgiana Slough, Beaver Slough, Hog Slough, Sycamore Slough, Suisun Marsh.	All of the locations listed in 1994 may be suitable for spawning, but based on better monitoring from the Spring Kodiak Trawl Survey, most adult fish have since been observed to aggregate around Grizzly Island, Sherman Island, and in the Cache Slough complex, including the subsequently flooded Liberty Island, Sacramento River Deep Water Ship Channel (SRDWSC), and Montezuma Slough.
	Adults could spawn December–July.	Adults are virtually never fully ripe and ready to spawn before February and most spawning is completed by May (in warm years) or June (in cool years). However, new research confirms that Delta Smelt are capable of multiple batch spawning (Kurobe et al. 2016).
Larval and juvenile transport	Larvae require adequate river flows to transport them from spawning habitats in backwater sloughs to rearing habitats in the open waters of the low-salinity zone.	Not likely. Most Delta Smelt that survive to the juvenile life stage do eventually inhabit water that is in the 0.5 to 6 ppt range, due to either downstream movement or decreasing outflow, or both. However, Delta Smelt larvae can feed in the same habitats they were hatched in and juvenile fish can rear in water less than 0.5 ppt salinity.
	Larvae require adequate flow to prevent entrainment.	No change.
	Larval and juvenile transport needs to be protected from physical disturbances like sand and gravel mining, diking, dredging, riprapping.	No change, but these disturbances seem likely to have more impact on spawning habitat than larval transport.
	2 ppt isohaline (X2) must be west of the Sacramento–San Joaquin River confluence to support sufficient larval and juvenile transport.	X2 is generally west of the confluence during February–June due to State Water Resources Control Board X2 standard. Movement downstream of larval or other life stages is likely to have less of an energy cost with more outflow versus less.
	Maturation must not be impaired by pollutant concentrations.	Developmental contaminants are present at benchmark levels in the Delta (Fong et al. 2016; Jabusch et al. 2018).
	Additional flows might be required in the period July–August to protect Delta Smelt that were	July–August outflow augmentations may be helpful, but not to mitigate entrainment. Habitat changes in the central and south Delta have rendered it seasonally

Primary Constituent Element	1994 Critical Habitat Rule	2018 State of Scientific Understanding
	present in the south and central Delta from being entrained in export pumps.	unsuitable to Delta Smelt during the summer; entrainment is seldom observed past June.
Rearing habitat	2 ppt isohaline (X2) should remain between Carquinez Strait in the west, Three-Mile Slough on the Sacramento River and Big Break on the San Joaquin River in the east. This was determined to be a range for 2 ppt salinity (including its tidal time scale excursion into the Delta).	X2 is generally in this area during February–June due to State Water Resources Control Board X2 standard; however the standard does have a drought off-ramp. Most juvenile Delta Smelt still rear in this area but it is now recognized that a few remain in the Cache Slough complex as well. It has not been verified that the X2 isohaline must be between Carquinez and the confluence for biotic factors regarding Delta Smelt.
Adult migration	Adults require unrestricted access to spawning habitat from December–July.	Adults disperse faster than was recognized in 1994; most spawning is finished by the time Spring Kodiak Trawls start in January.
	Unrestricted access results from adequate flow, suitable water quality, and protection from physical disturbance.	Biotic factors can also restrict access.

2.15.1 Primary Constituent Element 1

Physical habitat is defined as the structural components of habitat (USFWS 1994). The ancestral Delta was a large tidal marsh–floodplain habitat totaling approximately 300,000 acres. During the late 1800s and early 1900s, most of the wetlands were diked and reclaimed for agriculture or other human use. The physical habitat modifications of the Delta and Suisun Bay were mostly due to land reclamation and urbanization. Water conveyance projects and river channelization have had some influence on the regional physical habitat by armoring levees with riprap, building conveyance channels like the DCC, storage reservoirs like CCF, and by building and operating temporary barriers in the south Delta and permanent gates and water distribution systems in Suisun Marsh.

Between the 1930s to 1960s, the shipping channels were dredged deeper (about 12 meters) to accommodate shipping traffic from the Pacific Ocean and San Francisco Bay to ports in Sacramento and Stockton. These changes left Suisun Bay and the Sacramento–San Joaquin River confluence region as the largest places with the greatest depth variation in the typical range of the low-salinity zone. This region remained a highly productive nursery for many decades (Stevens and Miller 1983; Moyle et al. 1992; Jassby et al. 1995). However, the deeper landscape created to support shipping and flood control requires more freshwater outflow to maintain the low-salinity zone in the large Suisun Bay/river confluence region than was once required. The shipping itself has historically provided a source of nonnative organisms, that, along with lower Delta outflow and deep channelization, have contributed to the changing ecology of the upper estuary (Winder and Jassby 2011; Kratina et al. 2014; Andrews et al. 2017).

Although the Delta Smelt is a generally pelagic or open-water fish, depth variation of open-water habitats is an important habitat attribute (Moyle et al. 1992; Hobbs et al. 2006). In the wild, Delta Smelt are most frequently collected in water that is somewhat shallow (4 to 15 feet deep) where turbidity is often elevated and tidal currents exist but are not excessive (Moyle et al. 1992; Bever et al. 2016). In Suisun Bay, the deep shipping channels are poor quality habitat because tidal velocity is very high (Bever et al. 2016), but in the north Delta where tidal velocity is slower, the Sacramento Deepwater Shipping Channel is used to a greater extent. Adult Delta Smelt also use edge habitats as tidal current refuges and corridors to spawning habitats (Bennett and Burau 2015).

2.15.2 Primary Constituent Element 2

Water is defined as water of suitable quality to support various Delta Smelt life stages that allow for survival and reproduction (USFWS 1994). Certain conditions of temperature, turbidity, and food availability characterize suitable pelagic habitat for Delta Smelt and are discussed in detail below. Contaminant exposure can degrade this primary constituent element even when the basic habitat components of water quality are otherwise suitable (Hammock et al. 2015).

Turbidity: Delta Smelt require turbidity. Even in captivity, clear water is a source of physiological stress (Lindberg et al. 2013; Hasenbein et al. 2016). The small plankton that Delta Smelt larvae eat are nearly invisible in clear water. The sediment (or algal) particles that make turbid water turbid, provide a dark background that helps Delta Smelt larvae see their translucent prey (Baskerville-Bridges et al. 2004). Older Delta Smelt are less reliant on turbidity to see their prey, but juvenile fish still feed more effectively in water of moderate turbidity (Hasenbein et al. 2016) and probably need turbid water to help disguise themselves from predators (Ferrari et al. 2014). The turbidity of the Delta and Suisun Bay has been declining for a long time due to dams and riprapped levees, both of which cut off sources of sediment from rivers flowing into the estuary (Arthur et al. 1996; Wright and Schoellhamer 2004), and due to the spread of Brazilian waterweed (Hestir et al. 2016) which filters the water, increasing clarity. Water exports from the south Delta may also have contributed to the trend toward clearer water by removing resuspended sediment in the exported water (Arthur et al. 1996). The primary turbid areas that remain in the upper estuary are the semi-shallow embayments in northern Suisun Bay (Bever et al. 2016) and the lower Yolo Bypass region that includes Liberty Island and the upper reach of the Sacramento Deepwater Shipping Channel (Morgan-King and Schoellhamer 2013). Both tidal and river flows, as well as wind speed, affect turbidity in these locations (Bever et al. 2018). Many of the estuary’s deeper channels tend to have somewhat lower turbidity because water velocity and wind cannot resuspend sediment that sinks into deep water (Ruhl and Schoellhamer 2004).

Water temperature: Water temperature is the primary driver of the timing and duration of the Delta Smelt spawning season (Bennett 2005). Water temperature also affects Delta Smelt’s growth rate which in turn can affect their readiness to spawn (Rose et al. 2013a). Water temperature is not strongly affected by variation in Delta outflow; the primary driver of water temperature variation in the Delta Smelt critical habitat is air temperature (Wagner et al. 2011). Very high flows can transiently cool the upper estuary (e.g., flows in the upper 10th percentile, Kimmerer 2004) during the early part of the year, but the system rapidly re-equilibrates once air temperatures begin to warm.

Older laboratory based research suggested an upper water temperature limit for Delta Smelt of about 25°C, or 77°F (Swanson et al. 2000). Newer laboratory research suggests Delta Smelt temperature tolerance decreases as the fish age, but is a little higher than previously reported, up to 28°C or 82°F in the juvenile life stage (Komoroske et al. 2014). It should be kept in mind that these are upper acute water temperature limits, meaning temperatures in this range will kill, on the average, one of every two fish.

In the laboratory and the wild, Delta Smelt appear to have a physiological optimum temperature near 20°C or 68°F (Nobriga et al. 2008; Rose et al. 2013a; Jeffries et al. 2016); most of the upper estuary exceeds this water temperature from June through September (Wagner et al. 2011). Thus, many parts of the estuary are energetically costly and stress Delta Smelt. Generally speaking, spring and summer water temperatures are cooler to the west and warmer to the east due to the differences in overlying air temperatures between the Bay Area and the warmer Central Valley (Kimmerer 2004). In addition, there is a strong water temperature gradient across the Delta with cooler water in the north and warmer water in the south. The higher flows from the Sacramento River probably explain this north-south gradient. Note that water temperatures in the north Delta near Liberty Island and the lower Yolo Bypass are also typically warmer than they are along the Sacramento River (Sommer et al. 2001; Nobriga et al. 2005).

Food: Food and water temperature are strongly interacting components of Delta Smelt health and habitat because the warmer the water, the more food Delta Smelt require (Rose et al. 2013a). If the water gets too warm, then no amount of food is sufficient. The more food Delta Smelt eat (or must try to eat) the more they will be exposed to predators and contaminants. Water exports can limit the flux of phytoplankton production from the Delta into Suisun Bay (Jassby and Cloern 2000), but the effect of water exports on phytoplankton production appears to be lower than grazing by clams (Jassby et al. 2002) and ammonium inhibition of phytoplankton growth from Sacramento’s urban wastewater inputs (Dugdale et al. 2007).

Historically, prey production peaked when the low-salinity zone was positioned over the shoals of Suisun Bay during late spring through the summer, but this function has been depleted due to grazing by overbite clams (Kimmerer and Thompson 2014), high ammonium concentrations in critical habitat (Dugdale 2012; Dugdale et al. 2016), and water diversions (Jassby and Cloern 2000). Recent research suggests Delta Smelt occupying Suisun Bay may experience poor nutritional health (Hammock et al. 2015). Delta Smelt occupying the Cache Slough region in the north Delta are in better nutritional health, but have shown evidence of relatively high contaminant impacts. The southern Delta is among the more productive areas remaining in the upper estuary (Nobriga et al. 2005), but Delta Smelt cannot remain in this habitat during the warmer months of the year (Nobriga et al. 2008) and may face a high risk of entrainment when they occupy it during cooler months (Kimmerer 2008; Grimaldo et al. 2009). Extensive blooms of the toxin-producing cyanobacteria Microcystis in the central and southern Delta became abundant around 1999 and depending on flow, and temperature, blooms can extend westward into the low-salinity zone where Delta Smelt are rearing (Brooks et al. 2012). In one recent study, Delta Smelt that occupied Suisun Marsh fared better both in terms of nutrition and in experiencing a lower level of contaminant impacts (Hammock et al. 2015).

2.15.3 Primary Constituent Element 3

“River flow” was originally defined as transport flow to facilitate spawning migrations and transport offspring to low-salinity zone rearing habitats (USFWS 1994), currently called tidal surfing (Bennett and Burau 2015). Both the flood and ebb tide influence the Delta Smelt distribution and dispersal.

The spawning microhabitats of Delta Smelt are not known, but it is likely there is more available suitable spawning habitat when Delta outflow is high during spawning than when it is low because more of the estuary is covered in fresh- and low-salinity water when outflow is high (Jassby et al. 1995). An examination of the adults found that a majority were using fresh to low salinity water. Most spawning occurs between February and May. Delta outflow during February through May is mainly driven by the climatic effect on the amount and form of precipitation in the watershed, the storage and diversion of water upstream of the Delta, and CVP and SWP water operations in the Delta (Jassby et al. 1995; Kimmerer 2002a). Thus far, the 21st Century has tended to be pretty dry and warm and that could have resulted in some chronic reduction in spawning habitat availability or suitability.

2.15.4 Primary Constituent Element 4

Older laboratory research suggested that Delta Smelt have an upper acute salinity tolerance of about 20 ppt (Swanson et al. 2000) which is about 60 percent of seawater's salt concentration of 32 to 33 ppt. Newer laboratory-based research suggests that some individuals can acclimate to seawater, but that comes at a high energetic cost that is lethal to about one in four individuals (Komoroske et al. 2014, 2016). In the wild, Delta Smelt are nearly always collected at very low salinities, which recent laboratory research has confirmed is nearer to the physiological optimum (Komoroske et al. 2016). Few individuals are collected at salinities higher than 6 ppt (about 20 percent of seawater salt concentration) and very few are collected at salinities higher than 10 ppt (about 30 percent of seawater salt concentration) (Bennett 2005). This well documented association with fresh to low salinity water is a reason for the scientific emphasis on X2 as a Delta Smelt habitat indicator (Dege and Brown 2004; Feyrer et al. 2011). Recent research combining long-term monitoring data with three-dimensional hydrodynamic modeling shows that the spatial overlap of several of the key habitat attributes described above increases as Delta outflow increases (Bever et al. 2016). This means that higher outflow, which lowers the salinity of Suisun Bay and Suisun Marsh, increases the suitability of habitat in the estuary by increasing the overlap of some, but not necessarily all, needed elements.

2.16 Coho Salmon, Southern Oregon/Northern California Coastal ESU

2.16.1 ESA Listing Status

Southern Oregon/Northern California Coast (SONCC) Coho Salmon were listed as threatened under the ESA on May 6, 1997 (62 FR 24588). Subsequent to the Alsea Valley decsions (Alsea Valley Alliance v. Evans, 161 F. Supp. 2d 1154 (D. Or. 2001)), which provided guidance on the appropriate compositon of an ESU, this listing status was reaffirmed on June 28, 2005 (70 FR 37160). This ESU consists of populations from Cape Blanco, Oregon, south to Punta Gorda, California, including Coho Salmon in the Trinity River. NMFS designated critical habitat for SONCC Coho Salmon on May 5, 1999 (64 FR 24049) as accessible reaches of all rivers (including estuarine areas and tributaries) between the Elk River in Oregon and the Mattole River in California, inclusive).

2.16.2 General Life History and Habitat Requirements

Coho salmon exhibit a 3-year life cycle in the Trinity River and are dependent on freshwater habitat conditions year round because they spend a full year residing in freshwater. Most Coho Salmon enter rivers between August and January with some more northerly populations entering as early as June. Coho salmon river entry timing is influenced by a number of factors including genetics, stage of maturity, river discharge, and access past the river mouth. Spawning is concentrated in riffles or in gravel deposits at the downstream end of pools with suitable water depth, velocity, and substrate size. Spawning in the Trinity River occurs mostly in November and December.

Coho salmon eggs incubate from 35 to more than 100 days depending on water temperature, and emerge from the gravel 2 weeks to 7 weeks after hatching. Coho eggs hatch after an accumulation of 400 to 500 temperature units measured in degrees Celsius (°C) and emerge from the gravel after 700 to 800 temperature units. After emergence, fry move into areas out of the main current. As Coho grow they spread out from the areas where they were spawned.

During the summer, juvenile Coho prefer pools and riffles with adequate cover such as large woody debris with smaller branches, undercut banks, and overhanging vegetation and roots. Juvenile Coho overwinter in large mainstem pools, beaver ponds, backwater areas, and off-channel pools with cover such as woody debris and undercut banks. Most juvenile Coho Salmon spend a year in freshwater with some northerly populations spending 2 full years in freshwater. Coho in the Trinity River are thought be exclusively 3-year lifecycle fish (1 year in freshwater). Because juvenile Coho remain in their spawning stream for a full year after emerging from the gravel, they are exposed the full range of freshwater conditions. Most smolts migrate to the ocean between March and June with most leaving in April and May.

2.16.3 Historical and Current Distribution and Abundance

According to NMFS (2014), all nine Coho Salmon population units in the Klamath-Trinity Basin have declined dramatically in abundance relative to historical levels, including the three populations within the Trinity River Basin. These three populations are including (1) the Upper Trinity (North Fork Trinity River to Ramshorn Creek inclusive), (2) the Lower Trinity (Weitchpec to just below North Fork Trinity River confluence and tributaries excluding South Fork Trinity River), and (3) the South Fork Trinity sub-population (NMFS 2012). Coho Salmon were not likely the dominant species of salmon in the Klamath Trinity River before dam construction. They were, however, widespread in the Trinity Basin ranging as far upstream in the Trinity River as Stuarts Fork above Trinity Dam. Wild Coho in the Trinity River today are not abundant and the majority of the fish returning to the river are of hatchery origin. Returns to the Trinity River are monitored at the Willow Creek Weir, (typically sited within a few miles of the town of Willow Creek) (see figure 2.16-1 below). Run size estimates include Coho Salmon from all or part of the three Trinity River Coho Salmon populations. The proportion of Coho Salmon from each population is unknown, though most are thought to be of the Upper Trinity River Population Unit. Few juveniles or adults are observed in the Lower or South Fork Trinity Population Units. Adult return numbers to the TRH provide rough estimates of the hatchery-origin coho salmon return numbers.

Data from this monitoring program indicates the Trinity River portion of the Southern Oregon/Northern California Coast Coho Salmon ESU is predominately of hatchery origin (Figure 2-16.1). NMFS (2012) views such a high proportion of hatchery fish is a population to be a high level risk factor for continued existence of the populations in the Trinity Basin. NMFS, USBR, and CDFW are working to develop a Hatchery and Genetics Management Plan to mitigiate the adverse effects of the hatchery program on production of wild Coho Salmon in the Trinity River (NMFS 2017, USBR and CDFW 2017).

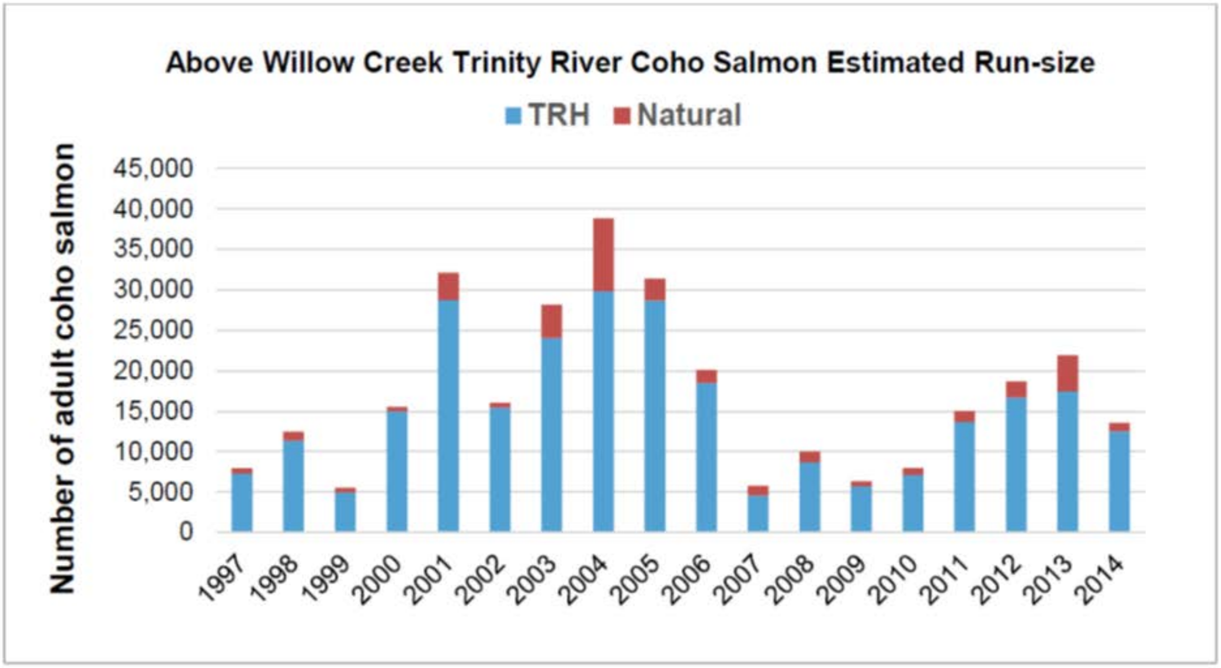

Figure 2.16-1. Estimated run-size of hatchery-origin and natural-origin Coho salmon in the Trinity River based on data collected at the Willow Creek weir (Reclamation and CDFW 2017).

2.16.4 Limiting Factors, Threats, Stressors

A number of interrelated factors affect Coho abundance and distribution in the Trinity River. These include water temperature, water flow, habitat suitability, habitat availability, hatcheries, predation, competition, disease, ocean conditions, and harvest. Current CVP operations affect primarily water temperature, water flow, and habitat suitability in the Trinity River. Climate change also affects water temperature, water flow, and habitat suitability in the Trinity River.

Juvenile Coho Salmon in the Trinity River spend up to a full year in freshwater before migrating to the ocean. Their habitat preferences change throughout the year and are highly influenced by water temperature. During the warmer summer months when Coho are most actively feeding and growing, they spend more time closer to main channel habitats. Coho tend to use slower water than Steelhead or Chinook Salmon. Coho juveniles are more oriented to submerged objects such as woody debris while Chinook and Steelhead tend to select habitats in the summer based largely on water movement and velocities, although the species are often intermixed in the same habitat. Juvenile Coho tend to use the same habitats as pikeminnows, a possible reason that Coho are not present in Central Valley watersheds. Juvenile Coho would be highly vulnerable to predation from larger pikeminnows during warm-water periods. Pikeminnow are limited to only a few SONCC Coho streams and they are not present in the Klamath River basin. When the water cools in the fall, juvenile Coho move further into backwater areas or into off-channel areas and beaver ponds if available. There is often no water velocity in the areas inhabited by Coho during the winter. These same off-channel habitats are often dry or unsuitable during summer because temperatures get too high.

Lewiston Dam blocks access to 109 miles of upstream habitat (U.S. Department of the Interior 2000). Trinity River Hatchery produces Coho Salmon with a production goal of 300,000 yearlings to mitigate for the upstream habitat loss. Habitat in the Trinity River has changed since flow regulation with the encroachment of riparian vegetation restricting channel movement and limiting fry rearing habitat (Trush et al. 2000). According to the Trinity River Restoration Plan, higher peak flows are needed to restore attributes of a more alluvial river such as alternate bar features and more off-channel habitats. These are projected in the restoration plan to provide better rearing habitat for Coho Salmon than the dense riparian vegetation currently present. Physical habitat manipulations have been implemented providing better juvenile rearing in selected sites along the river.

2.16.5 Water Operations

Reclamation makes releases from Lewiston Dam in accordance with the Trinity ROD, which considers requirements for Coho in the Trinity River. Increases in Trinity River releases in the late summer and fall result in lower storage in Trinity Reservoir at the end of the water year. The decreases in storage accumulate from water year to water year when the reservoir does not refill resulting in lower end-of-summer storages, negative impacts on cold water pool, and warmer stream temperatures for Coho and Fall-Run Chinook Salmon spawning in the Trinity River.

2.16.6 Recovery and Management

Reclamation is currently working on a Hatchery Genetics Management Plan for Trinity River Hatchery Coho.

2.17 Coho Critical Habitat

The critical habitat designation includes all waterways, substrate, and adjacent riparian zones, excluding: (1) areas above specific dams identified in the *Federal Register* notice (including Lewiston Dam); (2) areas above longstanding, natural impassable barriers (i.e., natural waterfalls in existence for at least several hundred years); and (3) Indian tribal lands.

2.18 Eulachon

2.18.1 ESA Listing Status

The southern DPS of Eulachon was listed as threatened under the ESA on March 18, 2010 (75 FR 13012) and the listing was reaffirmed on April 1, 2016. Critical habitat was designated on October 20, 2011 (76 FR 65324). The listing encompassed all spawning populations in rivers south of the Nass River in British Columbia to, and including, the Mad River in California (Gustafson et al. 2010).

2.18.2 General Life History and Habitat Requirements

Eulachon are an anadromous fish, meaning adults spend most of their life in the ocean but migrate into freshwater to spawn. Although they spend 95 to 98 percent of their lives at sea (Hay and McCarter 2000), current data only provides an incomplete picture concerning their saltwater existence. Their offspring hatch in freshwater but are carried to the estuary/ocean as larvae by the flow of the natal creek or river. The species is endemic to the northeastern Pacific Ocean, ranging from Northern California to the southeastern Bering Sea in Bristol Bay, Alaska (McAllister 1963; Scott and Crossman 1973; Willson et al. 2006). This distribution coincides closely with the distribution of the coastal temperate rainforest ecosystem on the west coast of North America (with the exception of populations spawning west of the Cook Inlet in Alaska).

Eulachon eggs can vary considerably in size but typically are approximately 1 mm (0.04 in) in diameter and average about 43 mg (0.002 oz) in weight (Hay and McCarter 2000). Eggs are enclosed in a double membrane; after fertilization in the water, the outer membrane breaks and turns inside out, creating a sticky stalk that acts to anchor the eggs to the substrate (Hart and McHugh 1944; Hay and McCarter 2000). Eulachon eggs hatch in 20–40 days with incubation time dependent on water temperature (Howell 2001). Shortly after hatching, the larvae are carried downstream and dispersed by estuarine, tidal, and ocean currents. It is not known how long larval Eulachon remain in the estuary before entering the ocean. Similar to salmon, juvenile Eulachon are thought to imprint on the chemical signature of their natal river basins. However, because juvenile Eulachon spend less time in freshwater environments than do juvenile salmon, researchers hypothesize that this short freshwater residence time may cause returning Eulachon to stray between spawning sites at higher rates than salmon (Hay and McCarter 2000).

Once juvenile Eulachon enter the ocean, they move from shallow nearshore areas to deeper areas over the continental shelf. Larvae and young juveniles become widely distributed in coastal waters, where they are typically found near the ocean bottom in 9 waters 20–150 m deep (66-292 ft) (Hay and McCarter 2000) and sometimes as deep as 182 m (597 ft) (Barraclough 1964). There is currently little information available about Eulachon movements in nearshore marine areas and the open ocean. However, Eulachon occur as bycatch in the pink shrimp fishery (Hay et al. 1999; Olsen et al. 2000; NWFSC 2008; Hannah and Jones 2009), which indicates that the distribution of these organisms overlaps in the ocean.

Eulachon typically spend several years in salt water before returning to fresh water to spawn from late winter through early summer. Spawning grounds are typically in the lower reaches of larger rivers fed by snowmelt (Hay and McCarter 2000). Willson et al. (2006) concluded that the age distribution of Eulachon in a spawning run varies considerably, but typically consists of fish that are 2-5 years old. Eulachon eggs commonly adhere to sand (Langer et al. 1977) or pea-sized gravel (Smith and Saalfeld 1955), though eggs have been found on a variety of substrates, including silt, gravel to cobble sized rock, and organic detritus (Smith and Saalfeld 1955; Langer et al. 1977; Lewis et al. 2002). Eggs found in areas of silt or organic debris reportedly suffer much higher mortality than those found in sand or gravel (Langer et al. 1977). The sexes must synchronize their activities closely, unlike some other group spawners such as herring, because Eulachon sperm remain viable for only a short time, perhaps only minutes (Hay and McCarter 2000). Eulachon are semelparous, meaning that they spawn once and then die. In many rivers, spawning is limited to the part of the river that is influenced by tides (Lewis et al. 2002), but some exceptions exist. In the Berners Bay system of Alaska, the greatest abundance of Eulachon is observed in tidally-influenced reaches, but some fish ascend well beyond the tidal influence (Willson et al. 2006). Eulachon once ascended more than 160 km (100 mi) in the Columbia River system (Smith and Saalfeld 1955). There is some evidence that water velocity greater than 0.4 meters/second (1.3 ft/second) begins to limit the upstream movements of Eulachon (Lewis et al. 2002).

Entry into the spawning rivers appears to be related to water temperature and the occurrence of high tides (Ricker et al. 1954; Smith and Saalfeld 1955; Spangler 2002). Spawning generally occurs in January, February, and March in the Columbia River, the Klamath River, and the coastal rivers of Washington and Oregon, and April and May in the Fraser River. Eulachon runs in central and northern British Columbia typically occur in late February and March or late March and early April. Eulachon typically spawn when water levels are lower and prior to spring freshets (Lewis et al. 2002). Rivers that experience Eulachon spawning generally have the characteristics of spring freshets caused by melting snow packs or glaciers (Hay and McCarter 2000). However, attempts to characterize Eulachon run timing are complicated by marked annual variation in timing. Willson et al. (2006) give several examples of spawning run timing varying by a month or more in rivers in British Columbia and Alaska. Water temperature at the time of spawning varies across the distribution of the species. Although spawning generally occurs at temperatures from 4 to 7°C (39 to 45º F) in the Cowlitz River (Smith and Saalfeld 1955), and at a mean temperature of 3.1°C (37.6º F) in the Kemano and Wahoo Rivers, peak Eulachon runs occur at noticeably colder temperatures (between 0 and 2°C [32 and 36º F]) in the Nass River. The Nass River run is also earlier than the Eulachon run that occurs in the Fraser River, which typically has warmer temperatures than the Nass River (Langer et al. 1977). Water temperatures between 4 and 10°C is preferred for adults entering the Columbia River (WDFW and ODFW, 2001). Sudden increases in temperatures above this range can lead to adult mortality and spawning failure (Blahm and McConnell 1971).

Eulachon larvae and juveniles eat a variety of prey items, including phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, and worm larvae (Barraclough 1967, Barraclough and Fulton 1967, Robinson et al. 1968a, 1968b). Eulachon adults feed on zooplankton, chiefly eating crustaceans such as copepods and euphausiids (Hart 1973; Scott and Crossman 1973; Hay 2002; Yang et al. 2006), unidentified malacostracans (Sturdevant 1999), and cumaceans (Smith and Saalfeld 1955). Adults and juveniles commonly forage at moderate depths (20 to 150 m [66 to 292 ft]) in nearshore marine waters (Hay and McCarter 2000). Eulachon adults do not feed during spawning (McHugh 1939; Hart and McHugh 1944).

2.18.3 Historical and Current Abundance

Eulachon spawn in rivers from southwestern Alaska to Northern California. The southern DPS encompasses spawning populations in rivers south of the Nass River in British Columbia to, and including, the Mad River in California (Gustafson et al. 2010). Historically, the only large river basins in the contiguous United States with large, consistent spawning runs were the Klamath River in Northern California and the Umpqua River in Oregon. However, Eulachon have been found both frequently and infrequently in other coastal rivers within this range, including the Mad River, Redwood Creek, and Humboldt Bay in California (Monaco et al. 1990; Willson et al. 2006, as cited in Gustafson et al. 2010).

There are no reliable historical abundance estimates for Eulachon. Available information (based largely on commercial fishery records) indicates that, starting in 1994, the southern DPS of Eulachon experienced an abrupt decline in abundance throughout its range (Gustafson et al. 2010). Since the 2010 listing, improved monitoring of Eulachon in several rivers detected general increases in adult spawning abundance, especially in 2013-2015 (NMFS 2016). However, sharp declines in Eulachon abundance occurred in 2016 and 2017 likely in response to poor conditions in the north east Pacific Ocean (NMFS 2017). The likelihood that these conditions will persist into the near future suggest that declines may again be widespread in upcoming years (NMFS 2017).

2.18.4 Limiting Factors, Threats, and Stressors

Factors that have been identified as major threats to southern DPS Eulachon include climate change impacts on marine and freshwater habitat, bycatch in offshore shrimp and groundfish fisheries, changes in flow quantity due to dams or water diversions, and predation (Gustafson et al. 2010). Because of similar trends in abundance across their range, large-scale oceanic and atmospheric patterns in the northeast Pacific Ocean associated with both natural climate variability and anthropogenic-forced climate change is likely the principal threat to Eulachon (NMFS 2017). The relationship between ocean conditions and population dynamics of Eulachon suggests that marine survival, most likely during the first weeks or month in the ocean, may have a large influence on overall survival and adult recruitment (NMFS 2016). Consequently, anomalously warm marine and freshwater conditions combined with below average precipitation may have contributed to poor returns of spawning adults in recent years (NMFS 2017). In 2010, an analysis of these threats, together with large declines in abundance, indicated that the southern DPS of Eulachon was at moderate risk of extinction throughout its range (Gustafson et al. 2010, 2012). NMFS's recent threats analysis indicated that the collective risk to the persistence of Eulachon has not changed significantly since the listing determination (NMFS 2016).

2.18.5 Water Operations Management

Reclamation currently does not manage for Eulachon, although they benefit from TRRP ROD flows and other releases in the Trinity River.

2.18.6 Recovery and Management

The Recovery Plan for Southern DPS of Eulachon (NMFS 2017) established recovery goals, objectives, and delisting criteria that NMFS will use in future ESA status reviews. The recovery goals for Eulachon are to (1) increase abundance and productivity of Eulachon, and (2) protect and enhance the genetic, life history, and spatial diversity of Eulachon throughout its geographical range, and reduce existing threats to warrant delisting of the species. Conservation actions that have been implemented in the U.S. and Canada to support recovery efforts include state regulations requiring the use of bycatch reduction devices in ocean shrimp fisheries, commercial and recreational fishery closures and catch prohibitions, seasonal dredging restrictions, dam removal, and Salmon and Steelhead habitat restoration projects (NMFS 2017). In recent years, the states of Oregon and Washington opened a limited-opportunity Eulachon fishery to better understand trends and variability in Eulachon abundance, fill critical information gaps on species biology and distribution, support cultural traditions of Northwest tribes, and provide limited public and commercial opportunities for Eulachon harvest to promote public engagement in Eulachon conservation and recovery (NMFS 2107).

2.18.7 Monitoring and Research

NMFS proposes to advance the conservation of Eulachon by working with stakeholders to continue to implement actions that further reduce the severity of threats to Eulachon, as well as develop a comprehensive research program to improve understanding of Eulachon population abundance and demographics, and understanding of large-scale threats (e.g., climate change) on Eulachon productivity, recruitment, and persistence (NMFS 2017). Specific research and monitoring needs include implementation of annual in-river spawning stock surveys and distribution surveys (e.g., environmental DNA), and identification of data and assessment needs to monitor annual variability and long-term trends in abundance, productivity, and viability of Eulachon across their range.

2.19 Eulachon Critical Habitat

Critical habitat was designated under the ESA for southern DPS Eulachon on October 20, 2011 (76 FR 65324). Critical habitat extends from the Elwha River in Washington to the Mad River in California. In California, designated critical habitat includes the Klamath River, Redwood Creek and Mad River. NMFS identified the following physical or biological features as essential for conservation of the southern DPS of Eulachon:

1. Freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation.

2. Freshwater and estuarine migration corridors free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted.
3. Nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.

Critical habitat does not include any Indian lands of the following federally-recognized tribes in the States of California, Oregon, and Washington: Lower Elwha Tribe, Washington; Quinault Tribe, Washington; Yurok Tribe, California; and Resighini Rancheria, California.

2.20 Riparian Brush Rabbit

2.20.1 ESA Listing Status

The USFWS listed riparian brush rabbit (*Sylvilagus bachmani riparius*) as an endangered species under the Endangered Species Act (ESA) on February 23, 2000 (65 FR 8881).

2.20.2 Critical Habitat Designation

No critical habitat rules have been published for the riparian brush rabbit.

2.20.3 General Life-History and Habitat Requirements

Riparian brush rabbits prefer dense, brushy areas of valley riparian forests, marked by extensive thickets of wild rose (*Rosa* spp.), blackberries (*Rubus* spp.), and willows (*Salix* spp.). Riparian brush rabbits prefer to remain hidden under protective shrub cover and seldom venture more than a few feet from cover. A typical response to danger is to retreat back into cover rather than to be pursued in open areas (USFWS 1998).

Riparian brush rabbits feed at the edges of shrub cover rather than in large openings (e.g., along trails, fire breaks, edges of thickets). Their diet consists of herbaceous vegetation such as grasses, sedges, clover, forbs, buds, bark and leaves of woody plants, and vines (USFWS 1998).

The approximate breeding season of riparian brush rabbits is from January to May. In favorable years, females may produce three or four litters. The young are born in a shallow burrow or cavity lined with grasses and fur and covered by a plug of dried vegetation. Although these rabbits have a high reproductive rate, five out of six rabbits typically do not survive to the next breeding season (USFWS 1998).

2.20.4 Historical and Current Distribution and Abundance

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 (USFWS 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 (USFWS 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. 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 (USFWS 2007).

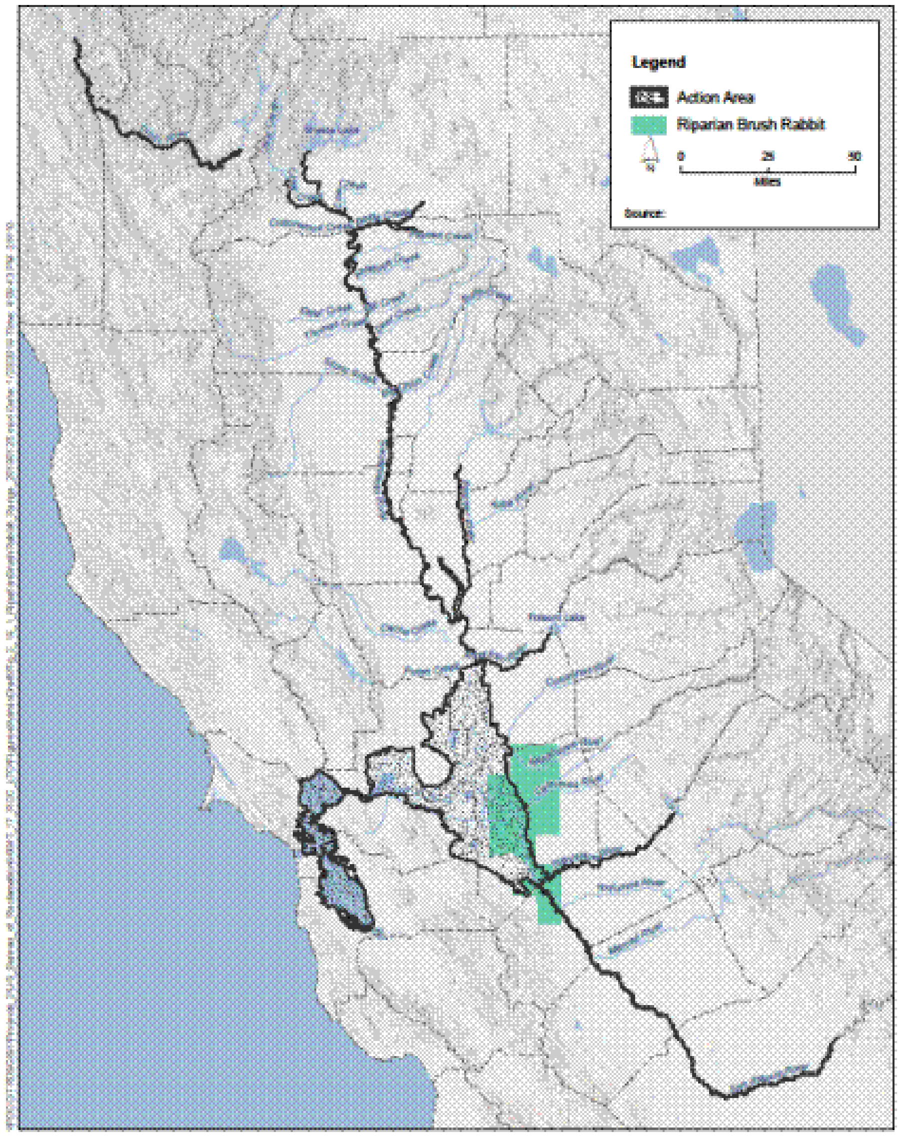

Figure 2.20-1. Riparian Brush Rabbit Range

2.20.5 Limiting Factors, Threats, and Stressors

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.

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; USFWS 1998). Climate change likely to increase the severity of flooding, impacting riparian brush rabbit.

Periodic flooding still occurs along all major rivers in the Central Valley (Kindel 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 (USFWS 1998).

Wildfire also pose a major threat. Long-term fire suppression combined with prolonged drought conditions can result in 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.

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; USFWS 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.

2.20.6 Recovery Considerations

The USFWS finalized the recovery plan for upland species of the San Joaquin Valley in 1998, which includes the riparian brush rabbit. Additionally, the riparian brush rabbit has limited coverage under the *San Joaquin County Multi-Species Habitat Conservation and Open Space Plan* (SJMSCP 2000).

The following are important components of riparian brush rabbit habitat when considering recovery actions:

- 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.

2.20.7 Monitoring and Research Programs

The San Joaquin River National Wildlife Refuge encompasses approximately 7,000 acres located where the Tuolumne, Stanislaus, and San Joaquin Rivers join, creating a mix of habitats for terrestrial wildlife and plant species. Initially established to protect and manage habitat for the Aleutian Cackling Goose, the refuge is currently managed to provide habitat for migratory birds and endangered wildlife species (USFWS 2012). River Partners have been working on increasing riparian brush rabbit population size; their restoration actions continue today and are expected to be completed in 2025. Over 500,000 native trees and shrubs such as willow, cottonwood, oak, blackberry, and rose have been planted across 2,200 acres of river floodplain within the San Joaquin River National Wildlife Refuge, creating the largest block of contiguous riparian woodland in the San Joaquin Valley. Endangered riparian brush rabbits have been reintroduced to this restored habitat from captive-reared populations. The goal is to have increased the available habitat for the riparian brush rabbit by more than 30 times its 1997 extent. The restored habitat will protect the population from nearing extinction in inevitable future flood events.

In 2015, Reclamation provided additional funds to the River Partners to restore 175 acres of historic floodplain forest that are now degraded back to riparian floodplain habitat at Dos Rios Ranch to benefit riparian brush rabbit, riparian woodrat, valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*), least Bell’s vireo, and western yellow-billed cuckoo in Stanislaus County. After successful pilot studies, two berms were strategically notched and removed from the landscape in 2018, which reconnected the endangered riparian brush rabbit and nine other endangered species to seasonally flooded land (River Partners 2018).

2.21 Riparian Woodrat

2.21.1 ESA Listing Status and Critical Habitat Designation

The USFWS listed riparian woodrat (*Neotoma fuscipes riparia*) as an endangered species under the ESA on February 23, 2000 (65 FR 8881).

2.21.2 Critical Habitat Designation

No critical habitat rules have been published for the riparian woodrat.

2.21.3 General Life-History and Habitat Requirements

Riparian woodrats are most numerous where shrub cover is dense and least abundant in open areas. In riparian areas, highest densities of riparian woodrats and their houses are often encountered in willow thickets with an oak overstory. They are common where there are deciduous valley oaks, but few live oaks. Mostly active at night, the riparian woodrat’s diet is diverse and principally herbivorous. Their diet consists of leaves, fruits, terminal shoots of twigs, flowers, nuts, and fungi (USFWS 2000).

Riparian woodrats are well known for their large terrestrial stick houses some of which can last for 20 or more years after being abandoned. At Caswell Memorial State Park, riparian woodrats construct houses of sticks and other litter. Houses are typically placed on the ground or against/straddling a log or exposed roots of a standing tree and are often located in dense brush. Nests also are placed in the crotches and cavities of trees and in hollow logs. Sometimes arboreal nests are constructed, but this behavior seems to be more common in habitat with evergreen trees such as live oak. With their general dependence on terrestrial stick houses, riparian woodrats can be vulnerable to flooding.

Riparian woodrats live in loosely cooperative societies and have a matrilineal (mother-offspring associations; through the maternal line) social structure. Unlike males, adjacent females are usually closely related and, unlike females, males disperse away from their birth den and are highly territorial and aggressive, especially during the breeding season. Consequently, populations are typically female-biased and, because of pronounced polygyny (mating pattern in which a male mates with more than one female in a single breeding season), the effective population size (i.e., successful breeders) is generally much smaller than the actual population size. This breeding system in combination with the small size of the only known extant population suggests that the riparian woodrat could be at an increased risk of extinction because of inbreeding depression.

2.21.4 Historical and Current Distribution and Abundance

Historical records for the riparian woodrat are similarly distributed along the San Joaquin, Stanislaus, and Tuolumne Rivers, and Corral Hollow, in San Joaquin, Stanislaus, and Merced Counties (Hooper 1938; Williams 1986). Thus, prior to the statewide reduction of riparian communities by nearly 90 percent (Katibah 1984), the riparian woodrat probably ranged throughout the extensive riparian forests along major streams flowing onto the floor of the northern San Joaquin Valley.

The range of the riparian woodrat is far more restricted today than it was in 1938 (Williams 1986). The only population that has been verified is the single, known extant population restricted to about 100 ha (250 acres) of riparian forest on the Stanislaus River in Caswell Memorial State Park. Williams (1993) estimated the size of this population at 437 individuals. Analysis of California Department of Water Resources land use maps indicate that there were approximately 50 acres (20 hectares) of "natural vegetation" present along the San Joaquin River near the type locality in 1988, though no woodrats have been seen in that area. Today there is no habitat for woodrats around El Nido, which is located about 5.5.miles (8.9 kilometers) east of the San Joaquin River, the closest possible riparian habitat.

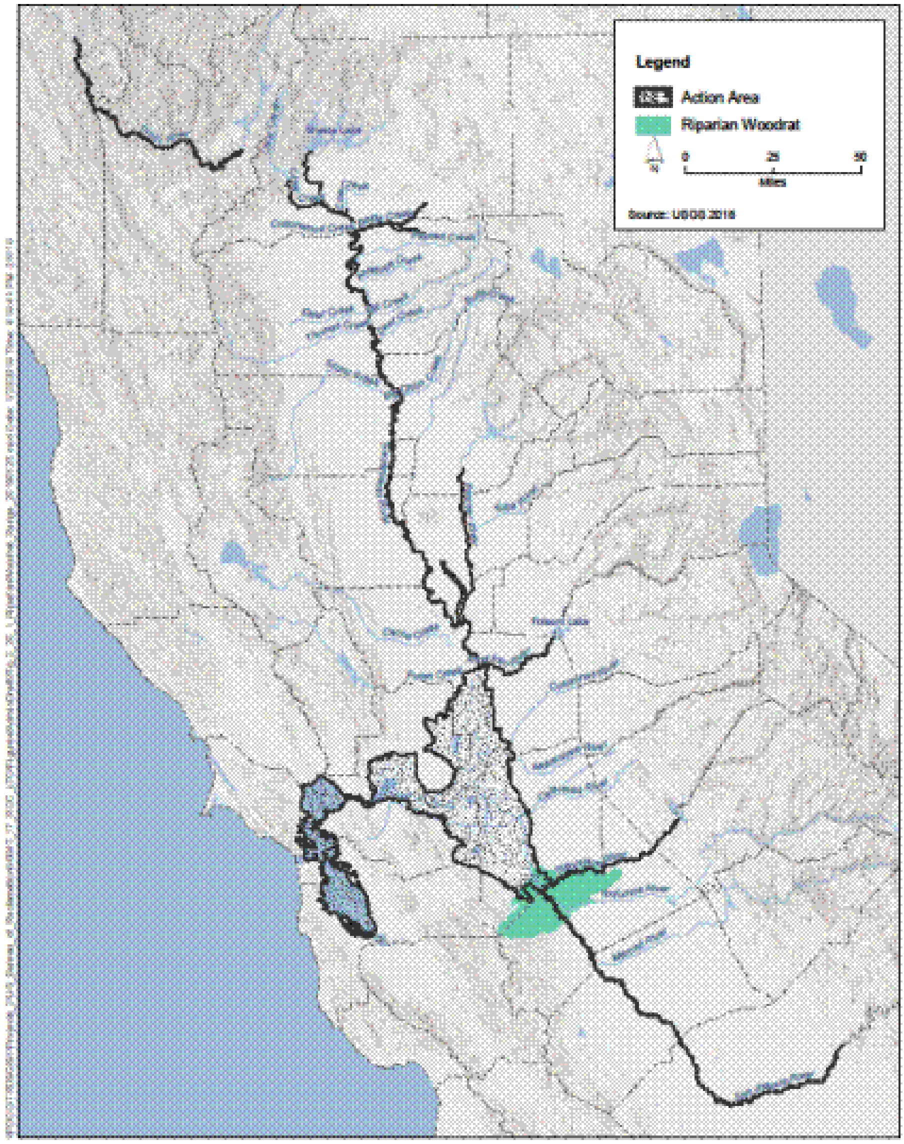

Figure 2.21-1. Riparian Woodrat Range

2.21.5 Limiting Factors, Threats, and Stressors

Loss, fragmentation, and degradation of habitat are the principal reasons for the decline of the riparian woodrat (USFWS 2000).

The most immediate threats to the single, small population of the species include naturally occurring events, such as drought, flooding of Caswell Memorial State Park lands, and wildfires. All of these environmental stressors are likely to increase in severity with climate change as California's snowpack decreases and watersheds move toward more precipitation driven hydrology (i.e., more variable). In addition, riparian woodrats are threatened by disease, predation, competition, clearing of riparian vegetation, use of rodenticide, and loss of genetic variability.

2.21.6 Recovery Considerations

The USFWS finalized the recovery plan for upland species of the San Joaquin Valley in 1998, which includes the riparian woodrat.

No specific conservation measures for the riparian woodrat are in place, but the species does receive some protection through the management plan for the riparian brush rabbit at the Caswell Memorial State Park.

2.21.7 Monitoring and Research Programs

The California Department of Parks and Recreation has supported some general small-mammal studies and woodrat population studies at the Caswell Memorial State Park (Cook 1992; Williams 1993).

In 2000, San Joaquin County developed a multispecies habitat conservation plan that considers habitat for the riparian woodrat. Some of the measures suggested under the plan may benefit or minimize negative impacts on the woodrat. A fire management plan has also been initiated for the Caswell Memorial State Park to protect habitat, but fires from outside sources still pose a threat.

In 2015, Reclamation provided additional funds to the River Partners to restore 175 acres of historic floodplain forest that are now degraded back to riparian floodplain habitat at Dos Rios Ranch to benefit riparian brush rabbit, riparian woodrat, valley elderberry longhorn beetle, least Bell's vireo, and western yellow-billed cuckoo in Stanislaus County. After successful pilot studies, two berms were strategically notched and removed from the landscape in 2018, which reconnected the endangered riparian brush rabbit and nine other endangered species to seasonally flooded land (River Partners 2018).

Section 4(c)(2)(A) of the ESA requires that the USFWS conduct a review of listed species at least once every 5 years. The USFWS announced review of 34 species in California and Nevada on May 21, 2010 which included review of the riparian woodrat (75 FR 28636).

2.22 Salt Marsh Harvest Mouse

2.22.1 ESA Listing Status and Critical Habitat Designation

The USFWS listed salt marsh harvest mouse (*Reithrodontomys raviventris*) as an endangered species under the ESA on October 13, 1970 (35 FR 16047).

2.22.2 Critical Habitat Designation

Critical habitat has not been designated for the salt marsh harvest mouse.

2.22.3 General Life-History and Habitat Requirements

Salt marsh harvest mice are critically dependent on dense cover and their preferred habitat is pickleweed (*Salicornia virginica*). However, harvest mice can use a broader source of food and cover, including salt grass (*Distichlis spicata*) and other vegetation typically found in the salt and brackish marshes of the region. Salt marsh harvest mice are seldom found in cordgrass or alkali bulrush (*Scirpus americanus* and *S. maritimus*). In marshes with an upper zone of peripheral halophytes (salt-tolerant plants), they use this vegetation to escape the higher tides, and may even spend a considerable portion of their lives there. They also move into the adjoining grasslands during the highest winter tides. During the spring and summer months, some individuals will move from pickleweed marsh to bordering grasslands.

Breeding occurs from March through November. The salt marsh harvest mouse does little nest building, and nest structures are generally composed of a loose arrangement of grass. One or two litters may be produced annually with three to four young per litter.

2.22.4 Historical and Current Distribution and Abundance

The salt marsh harvest mouse is endemic to the marshes of the San Francisco Bay. There are two subspecies: the southern subspecies (*R. raviventris*) is found in the South San Francisco Bay (South Bay), the Corte Madera area, and Richmond area in the Central Bay; and the northern subspecies (*R. halicoetes*) is found in the Marin Peninsula, as well as in the tidal and brackish marshes of San Pablo and Suisun Bays (USFWS 2013).

In most of its range, the salt marsh harvest mouse is found in the upper half of tidal salt marshes, where shallow flooding, high tide cover, and escape habitat are available (Shellhammer and Barthman-Thompson 2015; USFWS 2013). They species also occurs in some of the South Bay brackish marshes (Shellhammer et al. 2010).

Differences in population sizes for the two subspecies can likely be attributed to differences in available marsh habitat and ecotones throughout the species' range (Shellhammer and Barthman-Thompson 2015; USFWS 2013). For example, due to loss of marsh habitat, population numbers are low throughout the range of the southern subspecies (Shellhammer et al. 2010). Conversely, population numbers are higher in the brackish marshes of northern and western Suisun Marsh, where there is both a higher quantity and quality of available habitat. A study conducted by Sustaita et al. (2011) found a positive correlation between the density and height of mixed vegetation and salt marsh harvest mouse numbers. Sustaita and colleagues (2011) reported large populations in both pickleweed-dominant (*Salicornia virginica*) areas and areas with mixed halophytes, such as fat hen (*Atriplex triangularis*), alkali heath (*Frankenia salina*), Baltic rush (*Juncus balticus*), Olney's threesquare bulrush (*Schoenoplectus americanus*) and other halophytic species. Additionally, the results showed that areas with mixed vegetation that were not dominated by pickleweed were often as productive as the pickleweed-dominant areas (Sustaita el al. 2011).

CDFW and DWR conducted a 2-year mark-recapture study to investigate demographic performance and habitat use of the northern subspecies of the endangered salt marsh harvest mouse in the Suisun Marsh. The studies examined the effects of different wetland types and microhabitats on three demographic variables: density, reproductive potential, and persistence. The results indicate that microhabitats dominated by mixed vegetation or pickleweed supported similar salt marsh harvest mouse densities, reproductive potential, and persistence throughout much of the year. The studies showed that densities were higher in diked wetlands, whereas post-winter persistence was higher in tidal wetlands. The results emphasize the importance of mixed vegetation, where at least some vegetation is taller, and suggests that both diked and tidal wetlands support salt marsh harvest mouse populations by promoting different demographic attributes as well as adequate habitat. The southern subspecies, *R. raviventris*, occupies South San Francisco Bay marshes. Marshes in the South Bay generally lack the attributes that contribute to relatively high densities of mice in Suisun. South Bay marshes have lost most of their high marsh and upland ecotones to development, so mice have little escape cover during high tides.

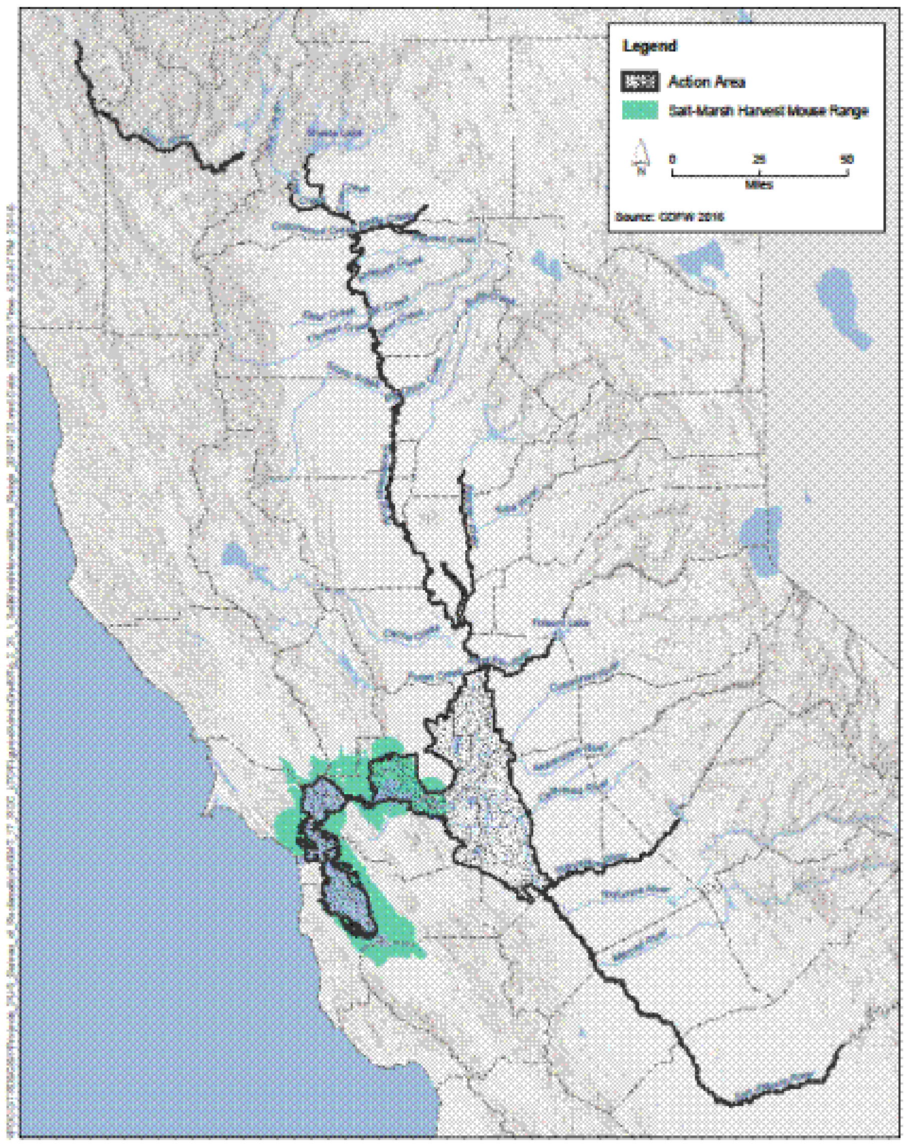

Figure 2.22-1. Salt Marsh Harvest Mouse Range

2.22.5 Limiting Factors, Threats, and Stressors

Salt marsh harvest mouse habitat loss can be ascribed to extensive urban and industrial development, particularly in the South Bay (USFWS 2013). Tidal marshes in the San Francisco Bay have lost the upper half of their mid-marsh zones, most of the high marsh zones, as well as most of the marsh/upland ecotones (Shellhammer and Barthman-Thompson 2015). Loss of the two latter areas means loss of escape cover during high tides. Many of the remaining South Bay marshes are very narrow, have poor vegetative cover, and reduce or prevent the movement of the salt marsh harvest mouse (Shellhammer and Duke 2010). Decreased sediment loads will likely result in narrowing of marshes (Cloern and Jassby 2012). The results are more and smaller populations that experience higher random genetic drift (Shellhammer and Duke 2010).

Climate change, particularly sea level rise due to climate change, will have a significant and negative impact on this species, especially in the South Bay where marshes are already narrow. For most of the mouse’s range, marshes are bordered by developed land. Areas of undeveloped upland available as habitat and ecotone still exist, including the Coyote Hills in the South Bay and the Sears Point area in the San Pablo Bay. There are also protected areas along the eastern side of the Marin Peninsula, but unfortunately, they are vulnerable to steeply-rising waters. The Suisun Marsh, which is further inland, is not subjected to the same high tides as the San Francisco Bay. With less intense flooding, development, and infrastructure, Suisun Marsh provides better migration and survival rates (Shellhammer and Barthman-Thompson 2015).

Increased salinity from sea level rise coupled with lower precipitation from climate change could lead to vegetation loss, specifically pickleweed, and changes to vegetation composition (Padgett-Flohr and Isakson 2003; Shellhammer and Barthman-Thompson 2015). Intense flooding and storm events could eliminate cover and refugia, thereby increasing predation (Johnston 1957) and destroying nests (Hadaway and Newman 1971).

2.22.6 Recovery and Management

The *Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California* covers five endangered species, including the salt marsh harvest mouse. The overall goal of the recovery plan is comprehensive restoration and management of tidal marsh ecosystems. (USFWS 2013)

Population resilience can be increased in tidal salt marshes by acquiring existing, historic, and restorable tidal marsh in order to increase marsh size, connectivity, and expand high marsh and ecotone areas (Shellhammer and Barthman-Thompson 2015; USFWS 2013). One significant and successful endeavor for the southern subspecies is the South Bay Salt Pond Restoration Project, which began in 2003. Projects in San Pablo Bay and Suisun Marsh, where tidal action was restored, have benefited the northern subspecies. Improving marsh connectivity increases genetic exchange and avoids inbreeding depression (Shellhammer and Barthman-Thompson 2015). Increased marsh size, complexity, and the possibility for extension landward are critical to the species’ survival.

Benefits to recreating large marshes include the development of raised overflow berms along their intermediate channels, which provide areas for marsh gumplant (*Grindelia robusta* var. *angustifolia*) to grow and offer escape cover from high tides (Shellhammer and Barthman-Thompson 2015). Restored marshes should have sloping upper edges where high marsh and transition zone vegetation can develop, even though the slopes will be narrow. One recurring dilemma with tidal restoration is that some areas to be diked have existing salt marsh harvest mouse populations (USFWS 2013). Because of this, unoccupied or unsuitable habitats have higher priority for tidal marsh restoration.

Salt marsh harvest mouse use of managed wetlands has been documented to be as high, or higher than, tidal wetland use (Sustaita et al. 2011). Downlisting of the salt marsh harvest mouse in the Suisun Bay Recovery Unit is achievable through 1,000 or more acres of muted or tidal marsh in the Western Suisun/Hill Slough Marsh Complex, 1,000 or more acres of muted or tidal marsh in the Suisun Slough/Cutoff Slough Marsh Complex, 1,500 or more acres of diked or tidal marsh in the Grizzly Island Marsh Complex, 1,000 or more acres of muted or tidal marsh in the Nurse Slough/Denverton Slough Marsh Complex, and 500 or more acres of muted or tidal marsh in the Contra Costa County Marsh Complex.

It is recommended that habitat management, restoration, and enhancement efforts include areas containing mixed vegetation, pickleweed in both diked and tidal wetlands, and areas that will accommodate sea level rise.

2.23 California Clapper Rail

2.23.1 ESA Listing Status

The USFWS listed California Ridgway's rail (*Rallus obsoletus obsoletus*), formerly California clapper rail (*Rallus longirostris obsoletus*), as an endangered species under the ESA on October 13, 1970 (35 FR 16047). Recent genetic analyses of rail species resulted in a change in the common name and taxonomy of the large, "clapper-type" rails (Rallus longirostris) of the west coast of North America to Ridgway's rail (Rallus obsoletus) (Maley and Brumfield 2013; Chesser et al. 2014). However, the change does not change the current listing status of the species. The USFWS will continue to recognize the species as the California clapper rail until the change in common name and taxonomy of the California clapper rail to Ridgway's rail is officially entered into the Federal Register.

2.23.2 Critical Habitat Designation

No critical habitat rules have been published for the species.

2.23.3 General Life-History and Habitat Requirements

The California Ridgway's rail is a year-round resident of tidally influenced salt and brackish marshes in the San Francisco Estuary. Areas used by California Ridgway's rails are dominated by pickleweed, Pacific cordgrass (*Spartina foliosa*), and salt grass in the lower tidal zone and taller pickleweed, gumplant, and wrack (the area where debris is deposited) in the upper tidal zone. They also can occupy habitats with other vegetative components, including bulrush, cattails (*Typha* spp.), and Baltic rush. Shrubby areas adjacent to or within the marsh may be important for predator avoidance during high tides. Nesting also occurs in this habitat.

California Ridgway's rails are most active in early morning and late evening, when they forage in marsh vegetation in and along creeks and mudflat edges. They are highly opportunistic feeders; principal food items include crabs, mussels, spiders, clams, snails, aquatic insects, isopods, pickleweed and Pacific cordgrass vegetation, seeds, and small fish. They often roost at high tide during the day.

The breeding season begins by February. Nesting starts in late March and extends into August. The end of the breeding season is typically defined as the end of August, which corresponds with the time when eggs laid during re-nesting attempts have hatched and young are mobile. Clutch sizes range from 5 to 14 eggs.

Both parents share in incubation and rearing. Nests are placed to avoid flooding by tides, yet in dense enough cover to be hidden from predators, generally on raised ground near tidal sloughs in low marsh habitats. The young are semiprecocial, incapable of moving from the nest for at least 1 hour after hatching and are brooded by the adults for several days. The young follow the adults during foraging and are able to forage independently on small prey soon after hatching.

2.23.4 Historical and Current Distribution and Abundance

The California Ridgway's rail is endemic to tidally influenced salt and brackish marshes of California. Historically, the California Ridgway's rail occurred in tidal marshes along California's coast from Morro Bay, in San Luis Obispo County, to Humboldt Bay, in Humboldt County. Thousands of California Ridgway's rails were eliminated by market hunters from the time of the Gold Rush until the passage of the Weeks-McLean Law in 1913, which was a precursor to the Migratory Bird Treaty Act of 1918 and was designed to stop commercial market hunting and illegal shipment of migratory birds from one state to another. Since that time, diking and filling for conversion to agriculture, urban development, and salt production have reduced the San Francisco Bay tidal marshes by 84 percent or more.

Currently, California Ridgway's rails are known to occur in tidal marshes in the San Francisco Estuary (estuary) (San Francisco, San Pablo, Grizzly, and Suisun Bays) (Olofson Environmental, Inc. 2011; CDFG 2011). California Ridgway's rails are typically found in the intertidal zone and sloughs of salt and brackish marshes dominated by pickleweed, Pacific cordgrass, Grindelia, saltgrass, jaumea, and adjacent upland refugia. They may also occupy habitats with other vegetative components, which include, but are not limited to, bulrush, cattails, and Baltic rush. In northern San Francisco Bay, California Ridgway's rails also occur in tidal brackish marshes that vary significantly in vegetation structure and composition, ranging from salt-brackish marsh to fresh-brackish marsh transitions (USFWS 2010a). Use of brackish marshes by California Ridgway's rails is largely restricted to major sloughs and rivers of San Pablo Bay and western Suisun Marsh, and along portions of Coyote Creek in the South Bay (USFWS 2010a). California Ridgway's rails were also found in nearly pure stands of alkali bulrush along Guadalupe Slough in 1990 and 1991 (H. T. Harvey & Associates 1990a, l990b and 1991). On rare occasions, California Ridgway's rails have been recorded even farther upstream, in brackish/freshwater transition marshes, particularly during the non-breeding season.

The California Ridgway's rail population was first estimated (between 1971-1975) at 4,200 to 6,000 birds, of which 55 percent occurred in the South Bay and 38 percent in the Napa Marshes (Gill 1979). Although the population was estimated at only 1,500 between 1981–1987 (Harvey 1988), the difference between these two estimates is believed to be partially due to survey intensity. Breeding season density data indicate that populations remained stable during the 1970s (Gill 1979; Harvey 1988), but reached an estimated all-time historical low of about 500 birds in 1991, with about 300 California Ridgway's rails in the South Bay (Harding et al. 1998). California Ridgway's rail numbers have rebounded between the 1990s and 2007. However, substantial increases in population may be difficult to achieve due to the current disjunct distribution of their habitat (Albertson and Evens 2000). Bay-wide California Ridgway's rail numbers in the estuary have been declining overall since 2007, and the decline is highly correlated with efforts to eradicate invasive Spartina in the estuary. U.S. Geological Survey (USGS) data suggest that Bay-wide California Ridgway's rail call count numbers declined by as much as 50 percent between 2007 and 2011.

Point Blue Conservation Science (formerly PRBO Conservation Science) conducted estuary-wide surveys of the San Francisco Bay for California Ridgway's rail between 2005 and 2010. Results of the 2008 survey indicated only 543 rails, compared to 938 rails detected in 2007 (PRBO Conservation Science

2009a). In both years, the South Bay accounted for the majority of California Ridgway's rails. Between 2005 and 2008, the estimated estuary-wide total population of California Ridgway's rails decreased by about 21 percent (Liu et al. 2009). The South Bay population of California Ridgway's rails decreased by 54 percent between 2007 and 2008 (Liu et al. 2009). Invasive Spartina Project (ISP) California Ridgway's rail survey data collected at 30 sites from 2004 to 2010 also shows an overall decline in California Ridgway's rails. The population increased by 25 percent between 2005 and 2006 and by 25 percent again between 2006 and 2007. Then count numbers decreased by 35 percent between 2007 and 2008, by 32 percent from 2008 to 2009 and by 13 percent from 2009 to 2010.

Data collected by ISP from 2004 to 2010 at 30 sites within the San Leandro Bay, the Hayward region, the San Francisco Peninsula, and the Newark region showed a decline in California Ridgway's rail numbers from 519 in 2007 to 202 in 2010. USGS data suggest that, estuary-wide California Ridgway's rail call count numbers declined by approximately 50 percent between 2007 and 2011. According to the California Ridgway's Rail Population Monitoring Report: 2005–2008, the estuary-wide California Ridgway's rail population showed an overall negative trend (-20.6 percent, $P<0.0001$) from 2005 to 2008, which can be mostly attributed to the 57 percent decline seen in the South Bay from 2007 to 2008 (PRBO Conservation Science 2009b). This decrease in the population of California Ridgway's rails in 2008 is highly correlated with large scale Spartina eradication during this period which resulted in the loss of cover. No new cover was created or enhanced for California Ridgway's rail to offset this loss.

In 20l0, Point Blue Conservation Science detected an increase of California Ridgway's rails in San Pablo Bay and South San Francisco Bay, while ISP detected a decline at other locations. This difference suggests that mature marshes (surveyed by Point Blue Conservation Science) which received a high degree of hybrid Spartina control still provided enough native habitat to support stable California Ridgway's rail population, while young marshes (surveyed by ISP), where hybrid Spartina was a more significant component of marsh vegetation cover, no longer provided habitat for California Ridgway's rails because California Ridgway's rails in these marshes were dependent on the hybrid Spartina for cover. It is unknown if the increased number of California Ridgway's rails detected at some locations is due to high breeding success or is a result of immigration from marshes where Spartina treatment resulted in a loss of high tide refugia habitat. In addition, high tide surveys conducted by East Bay Regional Parks District showed decreases in California Ridgway's rail numbers in San Leandro Bay since 2007. An extreme decline on East Bay Regional Parks District land occurred at Arrowhead Marsh which decreased from 112 California Ridgway's rails in 2007 to 35 in 2010.

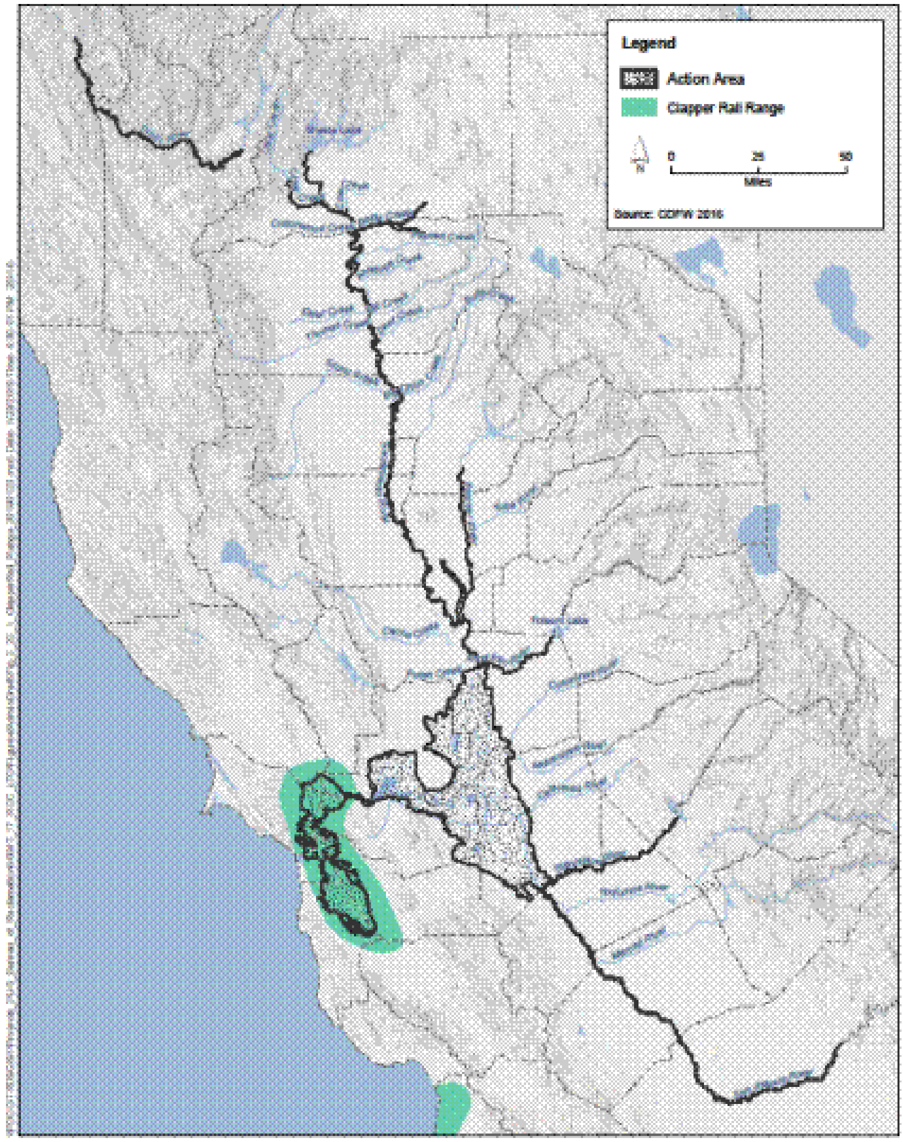

Figure 2.23-1. California Clapper Rail Range

2.23.5 Limiting Factors, Threats, and Stressors

California Ridgway's rail populations are limited by their small size, habitat fragmentation, and lack of tidal channel systems and other micro-habitat features. These limitations render much of the remaining tidal marsh acreage unsuitable or of low value for the species. Habitat loss has dramatically slowed since the California Ridgway's rail was listed in 1970, but ongoing disturbance and degradation precludes or reduces occupation of much of the remaining potential habitat by California Ridgway's rails. Remaining habitat has been fragmented by levee systems that reduce and isolate patches of habitat, reduce or eliminate high marsh and refugial habitat, and make habitat accessible to predators and human disturbance. Habitat has been filled, subjected to many contaminants, converted to less suitable vegetation conditions by fresh wastewater discharges, and submerged by land subsidence caused by agricultural practices and groundwater overexploitation. Loss of upper marsh vegetation has greatly reduced available habitat throughout the range of the California Ridgway's rail.

In addition to the problems associated with landscape alteration caused by development, California coastal wetlands are expected to be subject to the effects of global sea level rise and climate change due to global warming. The effects of past subsidence of marsh plain relative to mean tidal level, particularly in the South Bay (Atwater et al. 1979), are likely to be amplified by rising tidal levels.

California Ridgway's rails vary in their sensitivity to human disturbance, both individually and between marshes. California Ridgway's rails have been documented nesting in areas with high levels of disturbance, including areas adjacent to trails, dikes, and roads heavily used by pedestrian and vehicular traffic (USFWS 2013; Baye in litt. 2008). In contrast, Albertson (1995) documented a California Ridgway's rail abandoning its territory in the Laumeister Tract shortly after a repair crew worked on a nearby transmission tower. California Ridgway's rail reactions to disturbance may vary with season; however, both breeding and non-breeding seasons are critical times. Public trails that run along a narrow marsh transition zone may be particularly hazardous to California Ridgway's rails that depend on this habitat for refuge during high tides.

Throughout the estuary, the remaining California Ridgway's rail population is impacted by a suite of mammalian and avian predators and is exacerbated by at least 12 native and 3 nonnative predator species known to prey on various life stages of the California Ridgway's rail (Albertson 1995).

Mercury accumulation in eggs is perhaps the most significant contaminant problem affecting California Ridgway's rails in the estuary, with the South Bay containing the highest mercury levels. Mercury is extremely toxic to embryos and has a long biological half-life. Schwarzbach and colleagues (2006) found high mercury levels and low hatching success (due both to predation and, presumably, mercury) in California Ridgway's rail eggs throughout the estuary. California Ridgway's rail habitat is also at risk of contamination due to oil spills (Cosco Busan Oil Spill Trustees 2012).

2.23.6 Recovery and Management

The Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California (Recovery Plan; USFWS 2013) is an expansion and revision of the California Clapper Rail and Salt Marsh Harvest Mouse Recovery Plan (USFWS 1984). The Recovery Plan features the California Ridgway's rail (formerly California clapper rail) along with four other endangered species. The Recovery Plan identifies high priority areas for tidal marsh and ecotone restoration including restoring tidal action to many of the salt ponds and other diked baylands along San Francisco Bay. Thousands of acres of former salt ponds and other diked baylands along San Francisco Bay have been restored or are proposed to be restored to tidal action (Service file number 81420-2008-F-0621; USFWS 2008b); however, it may take decades before many of the heavily subsided areas within the former salt ponds accumulate enough sediment to become suitable tidal marsh habitat for California Ridgway's rails.The USFWS, on June 18, 2018 initiated 5-year status reviews for 50 species in California, Nevada, and the Klamath Basin of Oregon under the Endangered Species Act of 1973, as amended. A 5-year review is based on the best scientific and commercial data available at the time of the review; therefore, the USFWS is requesting submission of any new information on species that has become available since the last review.

2.23.7 Monitoring and Research Programs

The Don Edwards San Francisco Bay National Wildlife Refuge with assistance from the U.S. Department of Agriculture Wildlife Services currently manages mammalian and avian predators within California clapper rail habitat on its refuge lands in the South Bay and on DFW lands; however, the Predator Management Program is underfunded.

Although it has been suggested that habitat quality may be lower in brackish marshes than in salt marshes (Shuford 1993), further studies comparing reproductive success in different marsh types are necessary to determine the value of brackish marshes to California clapper rails.

2.24 Least Bell's Vireo

2.24.1 ESA Listing Status and Critical Habitat Designation

The least Bell's vireo (*Vireo bellii pusillus*) was listed as an endangered species by the USFWS on May 2, 1986 (51 FR 16474).

2.24.2 Critical Habitat Designation

Critical habitat, designated on February 2, 1994 (59 FR 4845 4867), is located in Santa Barbara, Ventura, Los Angeles, Riverside, San Bernardino, and San Diego Counties. No critical habitat for least Bell's vireo is present in the project Action Area or vicinity.

2.24.3 General Life-History and Habitat Requirements

Least Bell's vireo is an obligate riparian species during the breeding season, inhabiting structurally diverse woodlands along watercourses, including cottonwood-willow forests, oak woodlands, and mule fat (*Baccharis salicifolia*) scrub (USFWS 1998). Preferred breeding habitat generally consists of early successional, dense, low, shrubby vegetation in riparian areas, or young second-growth forest or woodland. This vireo is a subtropical migrant, typically arriving at breeding grounds in California from mid-March to early April. It can be highly territorial, and individuals have been known to return to the same breeding site, drainage, territory, and even nest tree each year, but birds may also disperse to new breeding sites (USFWS 1998). Birds may leave their breeding grounds as early as late July but are generally present until late September. Least Bell's vireos winter on the Baja California peninsula, where they occupy a variety of habitats, including mesquite scrub within arroyos, palm groves, and hedgerows bordering agricultural and residential areas (USFWS 1998).

Least Bell's vireos build their nest in dense cover in and along the edges of riparian habitat 3 to 6 feet off the ground, and within a dense, stratified canopy for foraging. Plant species composition and age are not important factors in nest site selection. Although least Bell's vireos nest in riparian habitat, they have also been observed foraging in adjacent upland habitats (USFWS 1998). Within a few days after pair formation, least Bell's vireos begin building a nest, with both parents constructing the cup-shaped nest composed of leaves, bark, willow catkins, spider webs, and other materials. Both parents incubate the eggs, which usually takes 14 days. Clutch size is typically three to four eggs, but may be as few as two or, rarely, up to five eggs. Fledging occurs approximately 10 to 12 days after the eggs hatch, with the adults continuing to care for the fledglings for 2 weeks. Pairs may attempt as many as five nests in a breeding season, but typically do not start nests after mid-July (USFWS 1998). Least Bell's vireos average between 1.1 and 2.4 young fledged per year (USFWS 1998).

2.24.4 Historical and Current Distribution and Abundance

The least Bell's vireo is a small, insectivorous bird of the southwestern United States. Historically, least Bell's vireo was widespread and abundant, ranging from the interior of Northern California near Red Bluff in Tehama County southward through the Sacramento and San Joaquin Valleys and Sierra Nevada Foothills, and in the Coast Ranges from Santa Clara County south to San Fernando, Baja California, Mexico.

By the early 1980s, least Bell's vireo was extirpated from the Sacramento and San Joaquin Valleys, with the species restricted to two locations in the Salinas River Valley in Monterey and San Benito Counties, one location along the Amargosa River in Inyo County, and numerous small populations in southern California south of the Tehachapi Mountains and in northwestern Baja California, Mexico (USFWS 1998). At the time of listing in 1986, over 99 percent of the least Bell's vireo population was found south of Santa Barbara County (USFWS 2006a). Since 1986, there has been a tenfold increase in the recorded least Bell's vireo population, largely due to efforts to control brown-headed cowbird (*Molothrus ater*) (USFWS 2006a). Breeding pairs have been observed in Monterey, San Benito, Inyo, Santa Barbara, San Bernardino, Ventura, Los Angeles, Orange, Riverside, and San Diego Counties, with the highest concentration of birds in San Diego County along the Santa Margarita River (USFWS 2006a). Pairs have also been observed exhibiting nesting behaviors in 2010 and 2011 in Yolo County within the Action Area (CDFW 2018). Although the breeding records do not yet support that least Bell's vireo has recolonized its historical breeding range in the San Joaquin and Sacramento Valleys (USFWS 2006a), the USFWS is including the small area around the 2010–2011 Yolo County unsuccessful nesting in the current range of the species (USFWS 2018).

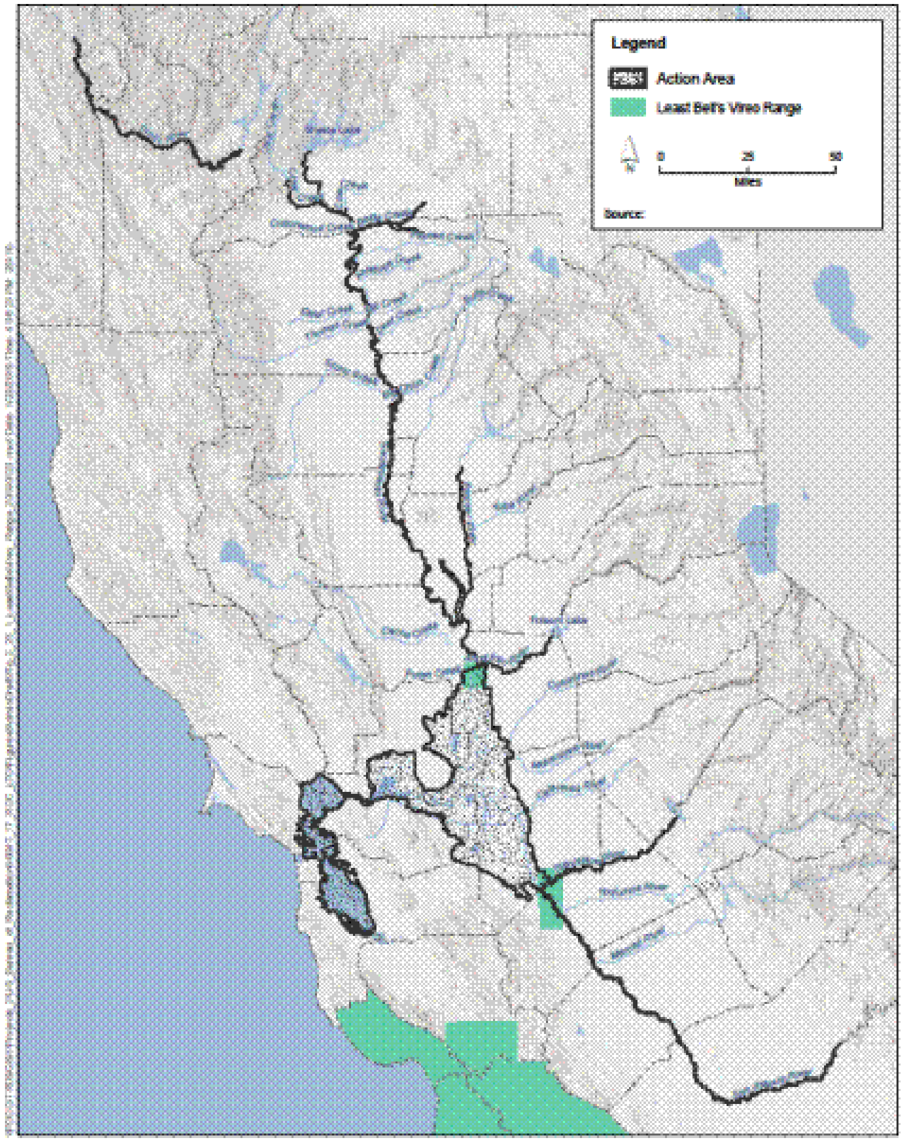

Figure 2.24-1. Least Bell's Vireo Range

2.24.5 Limiting Factors, Threats, and Stressors

The leading causes of least Bell's vireo decline include habitat loss and fragmentation resulting from stream channelization, flood control, water impoundment, water diversion, intensive recreation, agricultural conversion, livestock grazing, and urban development (Riparian Habitat Joint Venture [RHJV] 2004). Alteration of riparian landscapes can narrow or eliminate important population dispersal corridors. In addition, the degradation of riparian habitat resulting from construction of dams, levees, and diversions, clearing associated with farming and development, overgrazing, and invasion by exotic species can lead to disruption of natural hydrological conditions. Some of these factors may be exacerbated by climate change.

Another major threat to least Bell's vireo populations is the expansion of the range of the brown-headed cowbird, which acts as a brood parasite. Agricultural and livestock grazing areas located near riparian zones provide brown-headed cowbirds with ample foraging habitat close to songbird breeding grounds. Cowbird parasitism contributes to lowered productivity in host species through direct destruction of host eggs, competition between cowbirds and host chicks resulting in increased mortality, and nest abandonment in some species, all of which lower overall fecundity within a season (RHJV 2004). In addition, agricultural expansion and urbanization tend to reduce favorable conditions for top predators through habitat fragmentation and increased mortality from roadkill. The elimination of top predators, such as mountain lions (*Puma concolor*) and coyotes (*Canis latrans*), often results in an increased population of mid-level predators, such as raccoons, skunks, and domestic and feral cats, which are well documented nest predators (RHJV 2004).

2.24.6 Recovery and Management

Since its federal listing in 1986, along with intensive cowbird removal programs and riparian habitat protection, the population of Least Bell's Vireo has increased in the southern portion of its historic range, particularly in San Diego and Ventura counties, and is expanding northward (USFWS 1998). The USFWS prepared a Draft Recovery Plan for the Least Bell's Vireo in 1998. This plan details the importance of habitat conservation for Least Bell's Vireo recovery (USFWS 1998). Habitat features that are essential to Least Bell's Vireo conservation include riparian woodland vegetation that contains both canopy and shrub layers as well as associated upland habitat. Current threats to the remaining Least Bell's Vireo habitat that limit the ability for expansion of this habitat beyond its protected critical habitat core areas include stream channelization, water impoundment and extraction, water diversion, intensive recreation, and urbanization. Riparian areas are increasingly bordered by urban areas, whereas historically they were bordered by native upland plant communities. Vireo territories bordering agricultural and urban areas have been demonstrated to be less successful in producing young than territories bordering native upland plant communities (Kus 2002; USFWS 2006a).

2.25 Western Yellow-Billed Cuckoo

2.25.1 ESA Listing Status

The USFWS listed the western DPS of the yellow-billed cuckoo (*Coccyzus americanus*) as threatened on October 3, 2014 (79 FR 59992).

2.25.2 Critical Habitat Designation

Critical habitat, proposed on August 15, 2014 (79 FR 48547), includes sections of the Action Area along the Sacramento River from south of Red Bluff in Tehama County to Colusa, California. No final critical habitat has been designated for this species.

2.25.3 General Life-History and Habitat Requirements

The western yellow-billed cuckoo is a riparian obligate species, using riparian areas along low gradient rivers and streams and in valleys that provide floodplain conditions. Preferred habitat for this species consists of willow-cottonwood (*Salix-Populus*) riparian forest, but other tree species such as white alder (*Alnus rhombifolia*) and box elder (*Acer negundo*) may be important habitat components in some areas, including occupied sites along the Sacramento River (Laymon 1998).Potential habitat may also include valley marshland with willow riparian corridors, such as that found in the Llano Seco area of Butte County. Nesting habitat requires large expanses of willow-cottonwood forests (RHJV 2004). Along the Sacramento River, orchards of English walnut (*Juglans regia*), prune, and almond trees have also reportedly been used for nesting (Laymon 1980).

In western North America, yellow-billed cuckoos begin arriving from their wintering grounds in South America in mid- to late May (Hughes 1999). Nests usually consist of loose platforms of twigs lined with leaves or finer materials and, in the west, are often placed in willows, cottonwoods, and shrubs (Gaines and Laymon 1984). Clutch size ranges from one to five eggs, but is typically two to three (Hughes 1999). The entire period from egg laying to fledgling is one of the shortest among all bird species, lasting only 17 to 18 days, with incubation extending 9 to 11 days and nestlings fledging at 17 to 19 days of age (Hughes 1999). Young can typically fly at about 3 weeks of age. In years with a good food supply, yellow-billed cuckoos may lay two clutches of eggs. Although yellow-billed cuckoos usually raise their own young, they are facultative brood parasites, meaning they occasionally lay their eggs in nests of other yellow-billed cuckoos or of other bird species (Hughes 1999). They depart breeding grounds by early fall.

Yellow-billed cuckoos feed on katydids, caterpillars, cicadas, and other large insects. They forage in areas that are similar to breeding sites, but these areas may be smaller, narrower, and lack understory vegetation. Riparian vegetation is used by adults and young as a movement corridor between foraging sites and post-breeding dispersal areas. Western yellow-billed cuckoo may be found in a variety of vegetation communities during migration, including coastal scrub, secondary growth woodland, hedgerows, humid lowland forests, and forest edges below 8,125 feet above mean sea level, suggesting that the habitat needs of this species during migration are not as restricted as their habitat needs when nesting (Hughes 1999).

2.25.4 Historical and Current Distribution and Abundance

The yellow-billed cuckoo is a neotropical migrant bird that winters in South America and breeds in North America. The breeding range of the entire species formerly included most of North America, extending from southeastern and western Canada to the Greater Antilles in the Caribbean Sea and northern Mexico. At the time of the proposed listing, western yellow-billed cuckoo was not recognized as a separate sub-species and the USFWS determined that the western population segment, which nests in the portion of the United States west of the Continental Divide, is a DPS under the ESA.

The western DPS range has experienced significant reductions over the past 90 years. The northern limit of this species' breeding range along the west coast of the United States is now the Sacramento Valley. A small, potential breeding population exists in coastal Northern California along the Eel River (USFWS

2014b). In California, the yellow-billed cuckoo breeding range once extended from the Mexican border northward along the southern coast, and through the entire Central Valley (Grinnell and Miller 1944). However, its range is now generally restricted to the Sacramento Valley, the Kern River, and the lower Colorado River, with individuals occasionally reported in other areas (Laymon and Halterman 1987). The Sacramento Valley is believed to be a major population center for this species and the Sacramento River represents an area where yellow-billed cuckoo habitat has potentially increased over the last 30 years (Dettling and Howell 2011). The estimated breeding population in California is currently 40 to 50 pairs (78 FR 61639; October 3, 2013).

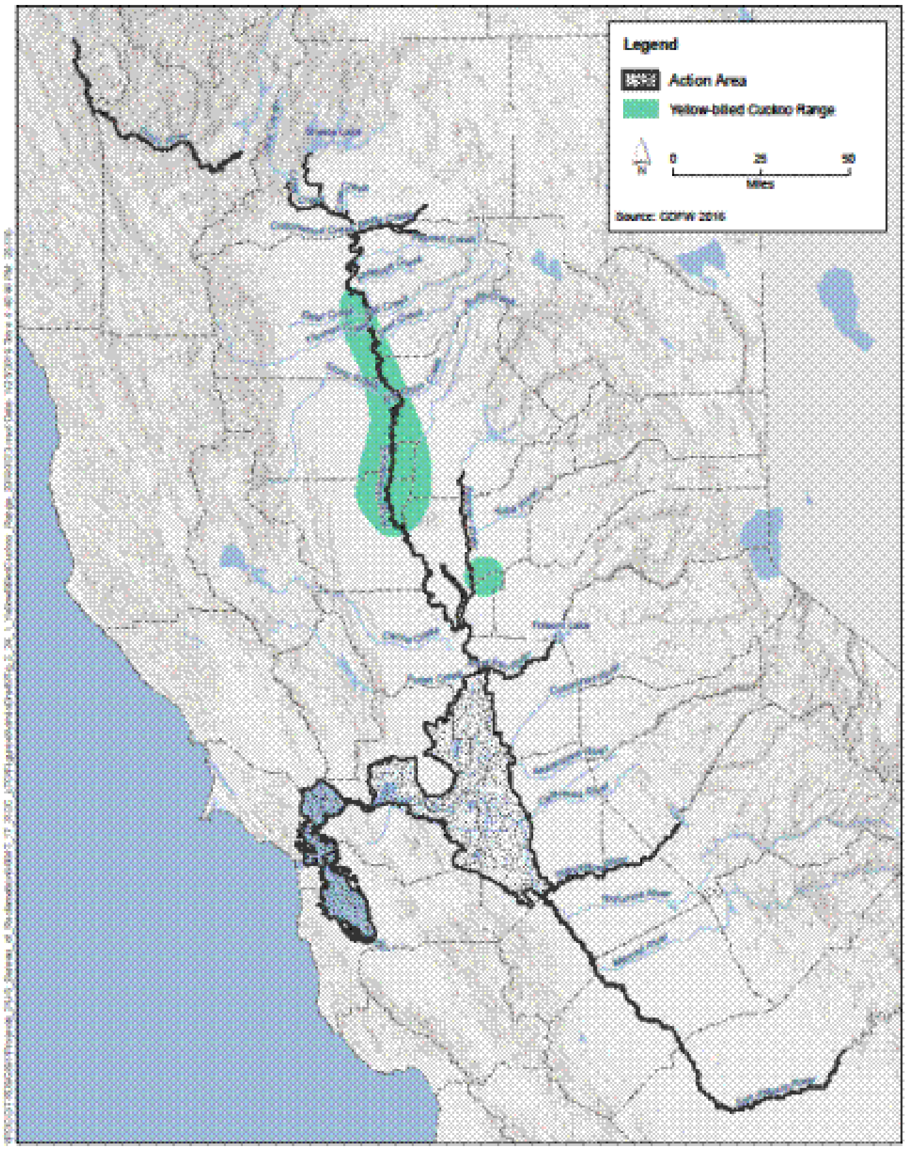

Figure 2.25-1. Western Yellow-Billed Cuckoo Range

2.25.5 Limiting Factors, Threats, and Stressors

Over the past 150 years, land use changes and alterations of river flow regimes have drastically reduced the amount of riparian forest in California and, therefore, the availability of breeding habitat for this neotropical migrant (Laymon and Halterman 1987). Because yellow-billed cuckoo is a riparian obligate, the range of this species in the western United States has become restricted to remaining isolated riparian forest fragments. Similarly, the number of western yellow-billed cuckoos in the western United States has declined substantially. The western population of cuckoos once ranged from northern Mexico to the Canadian border; however, they currently only breed in significant numbers in California, Arizona, New Mexico, and Texas (Gaines and Laymon 1984; Hughes 1999; USFWS 2014b).

Current threats to the yellow-billed cuckoo include habitat loss from flood control projects (including from ongoing maintenance), alterations to hydrology, development of urban and agricultural areas, climate change, and invasive species. The application of pesticides in riparian habitats and adjacent agricultural areas may affect the reproductive success of this species. In addition, a reduction in the availability of suitably sized prey may lead to the abandonment of nesting areas.

2.25.6 Recovery and Management

A recovery plan has not yet been developed for this species, so recovery efforts are based on general conservation needs. For a species like the Western Yellow-billed Cuckoo that has lost much of its former known occupied habitat, recovery would begin with the conservation of much of the remaining occupied and suitable habitat and restoration of suitable habitat that has been disturbed.

2.26 Giant Garter Snake

2.26.1 ESA Listing Status

The USFWS listed the giant garter snake (*Thamnophis gigas*) as a threatened species on October 20, 1993 (58 FR 54053) under the ESA.

2.26.2 Critical Habitat Designation

Critical habitat has not been designated for the giant garter snake.

2.26.3 General Life-History and Habitat Requirements

Endemic to the wetlands of the Sacramento and San Joaquin Valleys of California, giant garter snake historically inhabited tule marshes and seasonal wetlands created by overbank flooding of rivers and streams in the Central Valley. Present populations of giant garter snake inhabit agricultural wetlands and other waterways such as irrigation and drainage canals, sloughs, ponds, small lakes, low-gradient streams, and adjacent uplands. Because of the direct loss of natural perennial wetland habitat, the giant garter snake relies heavily on rice fields in the Sacramento Valley, but also uses managed marsh areas in federal National Wildlife Refuges and state Wildlife Areas.

The giant garter snake is approximately 15 times more active in aquatic habitats (Halstead et al. 2016), but at the same time Halstead et al. (2015) found high frequency use of terrestrial underground habitats to escape hot weather and for brumation. Giant garter snakes have been observed in uplands during the active spring and summer season up to hundreds of meters (hundreds of yards) from water bodies

(USFWS 2017c). The giant garter snake feeds primarily on aquatic prey, including small fish, frogs, and tadpoles.

Habitat requirements consist of adequate water during the active season (typically March through November) to provide food and cover; emergent, herbaceous wetland vegetation, such as cattails and bulrushes, for escape cover from predators and foraging habitat during the active season. Essential habitat components consist of 1) freshwater aquatic habitat with protective emergent vegetation cover where snakes can forage; 2) upland habitat near the aquatic habitat that can be used for thermoregulation and summer shelter (i.e., burrows), and 3) upland refugia outside flood waters that can serve as winter hibernacula (U.S. Fish and Wildlife Service 2017).

Ideal giant garter snake aquatic habitat exhibits the following characteristics.

- Water present from March through November.
- Slow moving or static water flow with mud substrate.
- Presence of emergent and bankside vegetation that provides cover from predators and may serve in thermoregulation.
- Absence of a continuous canopy of riparian vegetation.
- Available prey in the form of small amphibians and small fish.
- Thermoregulation (basking) sites with supportive vegetation such as folded tule clumps immediately adjacent to escape cover.
- Absence of large predatory fish.
- Absence of recurrent flooding, or, where flooding is probable, the presence of upland refugia.

Because of the historic loss of natural wetlands, the preferred aquatic habitat for giant garter snake, rice fields and more importantly their associated canals and drainage ditches have become important habitat for giant garter snakes within agricultural areas. While giant garter snakes are known to use rice fields seasonally, the species is strongly associated with the canals that supply water to and drain water from rice fields; these canals provide much more stable habitat than rice fields because they maintain water longer and support marsh-like conditions for most of the giant garter snake active season (Reyes et. al. 2017). The giant garter snake active season extends approximately April through September. While flooded rice fields provide a component of aquatic habitat for giant garter snakes that occupy rice-growing regions, rice fields only provide adequate cover for the species for approximately one-third of their active season (Halstead et. al. 2016). In the Sacramento Valley, cultivated rice generally emerges from flooded fields in late May or early June, but sufficient growth that provides cover for snakes does not occur until approximately late June. Water is then drawn off the fields to allow them to dry in late August or early September.

In addition to providing foraging and refuge habitat, canals and ditches provide connectivity between occupied habitats. Giant garter snakes rely on canals and ditches as movement corridors through agricultural landscapes. These corridors provide important habitat, and are used during daily movement within a home range. Studies of marked snakes indicated that individuals typically move about 0.25 to 0.5 miles per day and individuals have been documented to move up to 5 miles over the course of a few days. (Wylie et al. 2002).

Throughout the winter dormancy period, giant garter snakes inhabit small mammal burrows and other soil or rock crevices above flood elevations, often as far as 656 to 820 feet (200 to 250 meters) from the edge of summer aquatic habitat. They typically select burrows with sunny exposures along south- and west-facing slopes along canal banks, marshes, or even riprap. The breeding season extends from March into May, with females giving birth to live young from late July through early September (USFWS 2017). Brood size averages 17 to 23 young, but can range from 10 to 46 young. Newborn snakes immediately scatter into dense cover and soon begin feeding on their own. Giant garter snake growth rates are variable, with size typically doubling within a year. Sexual maturity averages 3 years for males and 5 years for females (USFWS 2017).

2.26.4 Historical and Current Distribution and Abundance

Historically, giant garter snake inhabited the Sacramento and San Joaquin Valleys—bounded by the Coast Range to the west and the Sierra Nevada to the east—from the vicinity of Chico in Butte County in the north to Buena Vista Lake in Kern County in the south. Currently, less than 5 percent of the historical 4.5 million acres (1.8 million hectares) of wetlands remain (USFWS 2017). Giant garter snake has been extirpated from the southern one-third of its range in former wetlands associated with the historical Buena Vista, Tulare, and Kern lakebeds. This species now occupies what remains of high-quality fragmented wetlands, including marshes, ponds, small lakes, and low-gradient streams with silt substrates, as well as managed waterways, including irrigation ditches, drainage canals, rice fields, and their adjacent uplands (USFWS 2017).

Occurrence records coincide with the historical distribution of large flood basins, freshwater wetlands, and tributaries of the Central Valley’s Sacramento and San Joaquin watersheds. Recent genetic studies indicate that giant garter snake populations should be grouped by watershed basin. The current population groupings that are genetically and geographically distinct are: Butte Basin, Colusa Basin, Sutter Basin, American Basin, Yolo Basin, Cosumnes-Mokelumne Watershed, Delta Basin, San Joaquin Basin, and Tulare Basin. The Yolo Basin—the Liberty Farms, Burell, and Lanare populations are presumed extirpated (USFWS 2017).

Giant garter snake abundance has decreased throughout its range. The distribution of giant garter snake in the northern part of the range may still reflect its historic distribution; however, distribution in the San Joaquin Valley has been substantially reduced, with only a few recent sightings (USFWS 2017). In the Central Valley, giant garter snake relies heavily on rice fields, but also uses managed marsh areas in National Wildlife Refuges and state Wildlife Areas.

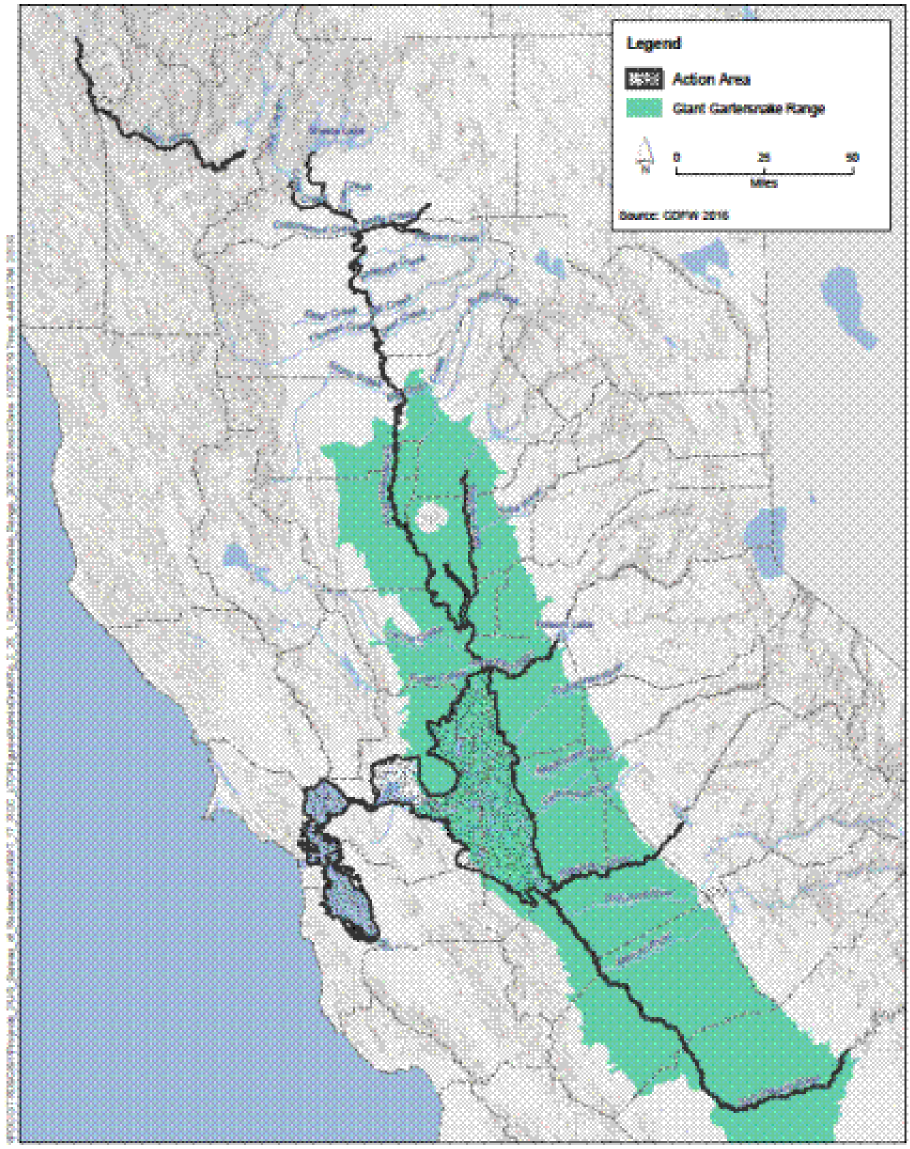

Figure 2.26-1. Giant Garter Snake Range

2.26.5 Limiting Factors, Threats, and Stressors

Habitat loss and fragmentation resulting from urbanization, agricultural conversion, and flood control activities are the main factors that have contributed to the decline of giant garter snake (USFWS 2017). Flood control activities, agricultural practices, and land and water management practices, such as wetland management for waterfowl, nonnative plant management, and water transfers, can alter the availability of summer water, thereby reducing habitat quality for giant garter snake. The loss of wetland ecosystems and suitable habitat has also resulted in giant garter snake using highly modified and degraded habitats among cultivated farm lands, including irrigation ditches, drainage canals, rice fields, and adjacent uplands. Current threats to giant garter snake include habitat loss and fragmentation due to urbanization; changes in the levels and methods of rice production; changes in water availability; levee and canal maintenance, water management, and water deliveries that do not take into account the requirements of the giant garter snake; water transfers (resulting from cropland idling/shifting, reservoir releases, conservation measures, or groundwater substitution); small population sizes; and invasive aquatic species (USFWS 2017).

Flood control and canal maintenance activities can subject snakes to ongoing risks of mortality and injury and can also lead to habitat fragmentation and dispersal barriers. Although giant garter snakes have been observed using rock riprap for thermoregulation, the flood control practice of lining streams and canals with large and extensive quantities of rock can be detrimental to wetland ecosystems and snakes by eliminating a natural thermal mosaic. Flood maintenance activities often include weed management, which destroys surface cover, and rodent eradication, which eliminates the occurrence and abundance of burrows and retreats that are used by giant garter snake for thermoregulation, for cover during shedding, and for over-wintering (USFWS 2012). Additional threats include predation, drought, climate change, roads (resulting in habitat fragmentation and vehicular threats), impaired water quality, selenium contamination, and mosquito abatement (USFWS 2017c).

A “mosaic of cover and water is likely beneficial” to snakes during the active season (Halstead et al. 2016). One study found that a lack of rice production adjacent to occupied canals appears detrimental to giant garter snake survival rates and populations. The study surmises that lower survival rates could be related to lower prey populations, increased predator presence, and a less secure water supply. This study supports the importance of maintaining water in canals adjacent to fallowed rice fields (Reyes et al. 2017). Research results indicate that there is a strong positive association between giant garter snake occupancy, soil classification, elevation, canal density, and the proportion of rice croplands (Hansen et al. 2017).

2.26.6 Recovery and Management

The USFWS finalized its giant garter snake recovery plan in 2017, with the recovery strategy focused on protecting existing, occupied habitat and identifying and protecting areas for habitat restoration, enhancement, or creation, including areas that are needed to provide connectivity between populations (USFWS 2017c). The three habitat components that are important for the giant garter snake include a freshwater aquatic component with protective emergent vegetation cover, nearby uplands that can be used for thermoregulation and summer shelter, and upland winter hibernacula (USFWS 2017c).

Protected waterfowl habitats in wildlife refuges are an important source of habitat for giant garter snake, but do not necessarily provide good habitat when they are flooded in winter and drained in summer. Rice fields and irrigation ditches, which are both flooded in summer, provide good habitat for this species.

2.26.7 Monitoring and Research Programs

In 2009, the California Department of Water Resources (DWR) developed a giant garter snake Baseline Monitoring and Research Strategy to help quantify and evaluate the response of the giant garter snake to rice land idling (USFWS 2015). DWR is working with the USGS Western Ecological Research Center (WERC) on the study of giant garter snake in the Sacramento Valley. The broad objective of this research effort is to provide scientific information to USFWS in support of identifying the effects of rice land idling for the purpose of water transfers on the species. Ultimately, the goal is to design conservation measures that will avoid and minimize effects on giant garter snake from rice land idling for water transfers. Once rice was emergent in the rice fields, giant garter snake used rice fields 39 to 60 percent of the time and canals 40 to 61 percent of the time. These results support that both rice fields and canals are important habitats for the species (Wylie and Casazza 2000a, 2000b).

Restored areas providing summer water were more effective in meeting the habitat needs of giant garter snake in the 2000-2001 study periods; therefore, giant garter snake did not have to venture as far as in previous years to find aquatic habitat during their active period. This was also found to be true for monitoring conducted during 2005. Sampling of the restored areas in Colusa NWR during the summers of 2002 and 2003 continued to document use of the restored wetland area as the habitat quality improves (USFWS 1999; Wylie and Casazza 2000a; Wylie et al. 2002).

The occurrence of rice agriculture, its supporting network of irrigation and drainage canals, and the restoration of marsh habitats currently provide suitable giant garter snake habitat. Research demonstrates, however, giant garter snake have not been able to disperse into all suitable habitats, and are largely restricted to areas near locations at which they were likely historically abundant (Halstead et al. 2016).

Central Valley Project Conservation Program (CVPCP) and Central Valley Project Improvement Act Habitat Restoration Program (HRP)

Developed during the Section 7 consultation process for the CVPIA and renewal of CVP water service contracts, the Central Valley Project Conservation Program (CVPCP) implements actions to protect, restore, and enhance special-status species populations and habitat, especially federally-listed species (USFWS 2015). Since the mid-1990s, the CVPCP and HRP have routinely identified and funded giant garter snake research and habitat improvement as top Priority Actions (USFWS 2015).

2.27 Soft Bird’s Beak

2.27.1 ESA Listing

U.S. Fish and Wildlife Service (USFWS) listed soft bird’s beak as an endangered species under ESA on November 20, 1997 (62 FR 61925).

2.27.2 Critical Habitat Designation

Critical habitat for soft bird's beak was designated in 2007 (72 FR 18536). The designated critical habitat areas contain physical and biological features (primary constituent elements [PCEs]) that are considered essential to the conservation of the species and that may require special management considerations and protection. The PCEs identified for soft bird's beak are: (1) persistent emergent, intertidal, estuarine wetland at or above the mean high-water line (as extended directly across any intersecting channels); (2)

rarity or absence of plants that naturally die in late spring (winter annuals); and (3) partially open spring canopy cover at ground level, with many small openings to facilitate seedling germination. In total, five critical habitat units covering approximately 2,276 acres (921 hectares) were designated. The critical habitat is located within Contra Costa, Napa, and Solano Counties at Fagan Slough Marsh, Hill Slough Marsh, Point Pinole Shoreline, Rush Ranch/ Grizzly Island Wildlife Area, and Southampton Marsh.

2.27.3 General Life-History and Habitat Requirements

Soft bird's beak is an annual herb of the snapdragon family (Scrophulariaceae). It grows 10 to 16 inches tall, branching sparingly from the middle and above. A floral bract (modified leaf) with two to three pairs of lobes occurs immediately below each inconspicuous white or yellowish-white flower. Flowers appear between May and September. Like other members of *Cordylanthus* and related genera, soft bird's beak is partially parasitic on the roots of other plants. Soft bird's beak is found predominantly in the upper reaches of salt grass/pickleweed marshes of the San Francisco Estuary at or near the limits of tidal action. It is associated with pickleweed (*Salicornia pacifica*), saltgrass (*Distichlis spicata*), fleshy or marsh jaumea (*Jaumea carnosa*), alkali seaheath (*Frankenia salina*) and seaside arrowgrass (*Triglochin maritima*).

2.27.4 Historical and Current Distribution and Abundance

Soft bird’s beak grows at the upper margin of tidal brackish high marshes in the San Francisco Estuary, often near the upper marsh–upland boundary (Grewell 2005; Grewell et al. 2007, p. 140). Where the topography is relatively uniform, soft bird’s beak is distributed in bands at the upper margin of the brackish high marsh. In Suisun Marsh, these bands are not correlated with elevation, but with soil pore water salinity during the dry season, which is determined by distance to channel and varies from season to season depending on freshwater flows from creeks draining into the marsh (Culberson 2001). Where the topography is more complex, such as areas with ridges or mounds and on levee banks, soft bird’s beak can be found in a variety of patch shapes (Grewell 2005; Grewell et al. 2007, p. 140). Plant distribution is influenced by a number of factors, including the existence of a persistent seed bank, the dispersal and germination dynamics of its floating seed, the extent of bare soil where seedlings can establish, the presence of appropriate long-lived annual or perennial host species, and the absence of dense populations of large, perennial, nonnative plant species (Grewell et al. 2003; Grewell 2005; Grewell et al. 2007, p. 143–144). The presence of a natural tidal inundation pattern is important, and the more muted the tidal influence is, such as tidal creeks with salt water exclusion gates or marshes with extensive levee systems, the less suitable the habitat is for soft bird’s beak (Grewell et al. 2003; Grewell 2005; Grewell et al. 2007, p. 140). A number of hypotheses have been suggested to explain the effects of the muted tidal influence, including increased rates of seed predation and herbivory by native insects, high densities of inappropriate host species, such as nonnative annual plants, and invasion and displacement by large nonnative plant species, such as perennial pepperweed (*Lepidium latifolium*) (Grewell 2005).

2.27.5 Limiting Factors, Threats, and Stressors

Threats to the subspecies include the destruction of habitat, erosion, the elimination or muting of tidal regimes, overgrazing and trampling by livestock, rooting by feral pigs, invasion of habitat by nonnative annual plants that are inappropriate hosts, recent invasion of its habitat by perennial pepperweed, alteration of salinity regimes, mosquito abatement, and oil spills (Fiedler et al. 2007; Grewell et al. 2003; Grewell 2005). Trampling and disturbance by cattle, feral pigs, and human foot traffic can directly damage plants and also damage the fragile root connections between soft bird's beak and the host plants (U.S. Fish and Wildlife Service 2009a). Seed predation by moth larvae may be an important factor in population declines at sites in Suisun Marsh (Grewell et al. 2003; Fiedler et al. 2007). Climate change and sea level rise may change tidal regimes faster than species can react, leading to increased pressure on the soft bird's beak.

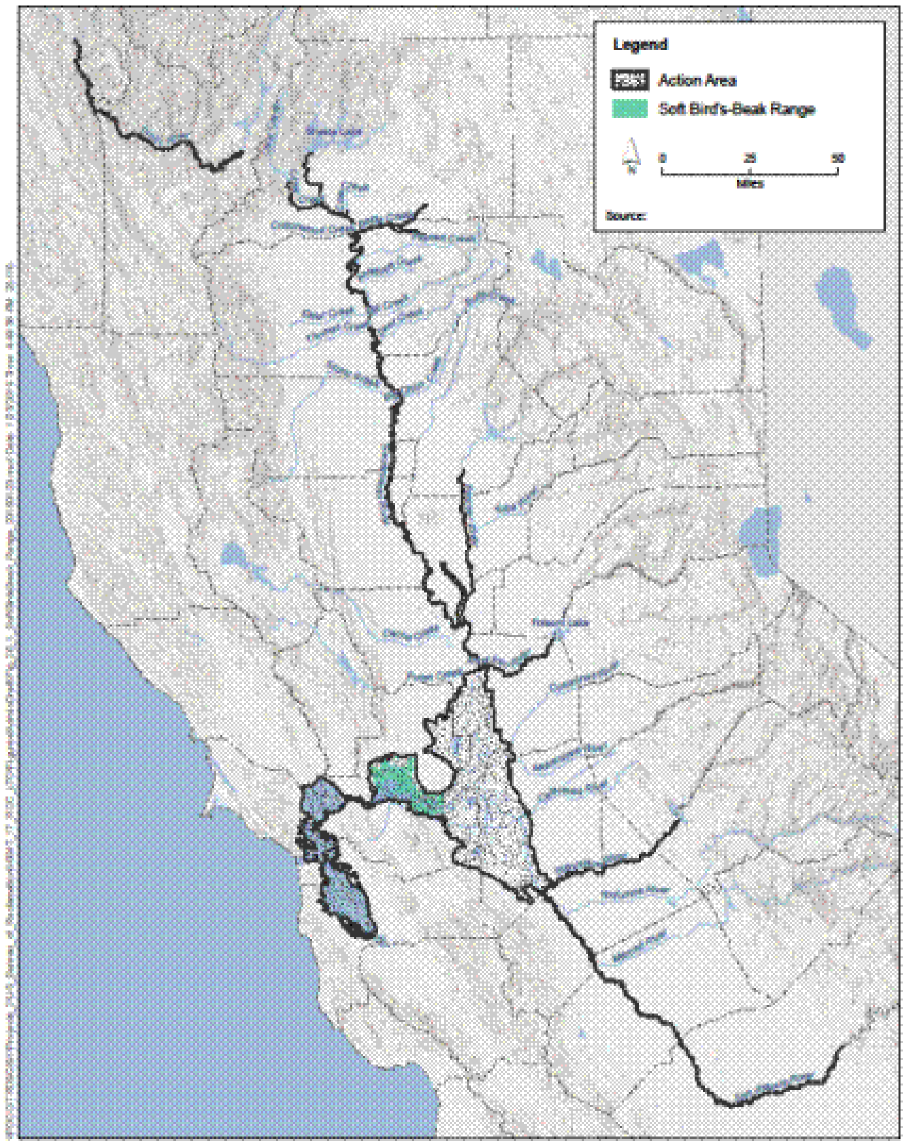

Figure 2.27-1. Soft Bird's Beak Range

2.27.6 Recovery and Management

The status of soft bird's beak and information about its biology, ecology, distribution, and current threats is available in the *Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California* (U. S. Fish and Wildlife Service 2013). The plan features soft bird's beak, along with four other endangered species.

There are two recovery units for this species, the Suisun Recovery Unit and the San Pablo Bay Recovery Unit. Downlisting of soft bird's beak would be achieved if the minimum area inhabited by the species in the Suisun Bay Area Recovery Unit is at least 3,000 acres and at least 1,000 acres in the San Pablo Bay Recovery Unit, over a period of 5 years. A minimum of 5,000 acres of suitable habitat in both recovery units must be permanently established. This must include existing or successfully restored tidal marsh areas with suitable habitat for the species and encompass a minimum of 80 percent of the species. Perennial pepperweed populations must be reduced to less than 10 percent cover in Suisun Marsh for 5 years, natural tidal cycles must be restored at Hill Slough, and the ponded area at Rush Ranch must be returned to periodic tidal flooding. There must be less than 10 percent total cover of other nonnative, invasive perennial or nonnative winter annual grass species.

2.28 Suisun Thistle

2.28.1 ESA Listing Status

USFWS listed Suisun thistle as an endangered species under ESA on November 20, 1997 (62 FR 61925).

2.28.2 Critical Habitat Designation

Critical habitat for Suisun thistle was designated in 2007 (72 FR 18536). The designated critical habitat areas contain physical and biological features (PCEs) that are considered essential to the conservation of the species, and that may require special management considerations and protection. The PCEs identified for Suisun thistle are: (1) persistent emergent, intertidal, estuarine wetland at or above the mean high-water line (as extended directly across any intersecting channels); (2) open channels that periodically contain moving water with ocean-derived salts in excess of 0.5 percent; and (3) gaps in surrounding vegetation to allow for seed germination and growth. In total, three critical habitat units covering approximately 2,052 acres (830 hectares) were designated. The critical habitat is located within Solano County at Hill Slough Marsh, Peytonia Slough Marsh, and Rush Ranch/ Grizzly Island Wildlife Area.

2.28.3 General Life-History and Habitat Requirements

Suisun thistle is a perennial herb in the aster family (Asteraceae). It has slender, erect stems that are 3.0 to 4.5 feet tall and well branched above. Pale, lavender-rose flower heads, 1 inch long, grow singly or in loose groups. Flowers appear between July and September. Suisun thistle grows in the upper reaches of tidal marshes of the San Francisco Estuary, where it is associated with narrowleaf cattail (*Typha angustifolia*), three-square or American bulrush (*Scirpus americanus*), Baltic rush (*Juncus balticus*), and saltgrass.

2.28.4 Historical and Current Distribution and Abundance

This species is known to exist only in Suisun Marsh and typically is found in the middle to high marsh zone along tidal channels and in irregularly flooded estuarine wetlands (Interagency Ecological Program 2001). One population occurs on California Department of Fish and Wildlife's (DFW's) Peytonia Slough Ecological Reserve. The remaining occurrences are associated with the Cutoff Slough tidal marshes and DFW's Joice Island Unit of the Grizzly Island Wildlife Management Area.

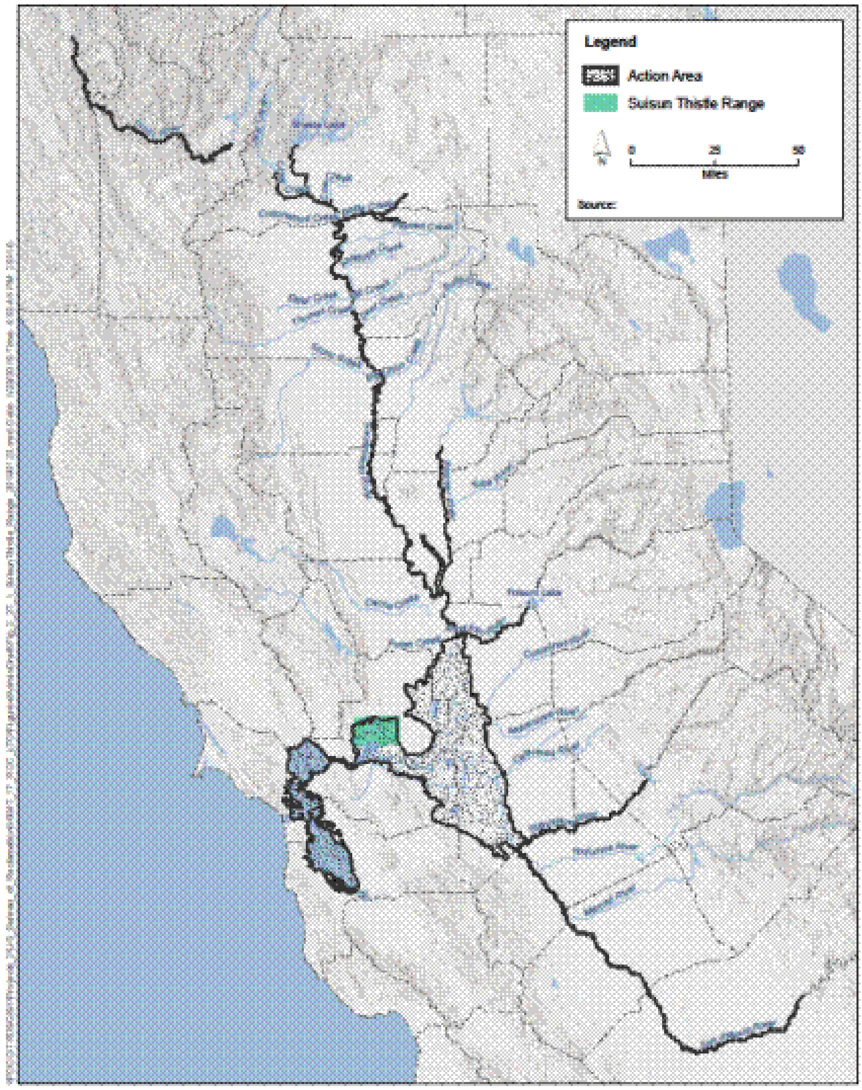

Figure 2.28-1. Suisun Thistle Range

2.28.5 Limiting Factors, Threats, and Stressors

Common threats that may require special management considerations or protections of the PCEs for Suisun thistle in all three critical habitat units include: (1) alterations to channel water salinity and tidal regimes from the operation of the Suisun Marsh Salinity Control Gates that could affect the depth, duration, and frequency of tidal events and the degree of salinity in the channel water column; (2) mosquito abatement activities (dredging, and chemical spray operations), which may damage the plants directly by trampling and soil disturbance, and indirectly by altering hydrologic processes and by providing relatively dry ground for additional foot and vehicular traffic; (3) rooting, wallowing, trampling, and grazing impacts from livestock and feral pigs that could result in damage or loss to Suisun thistle colonies, or in soil disturbance and compaction, leading to a disruption in natural marsh ecosystem processes; (4) the proliferation of nonnative invasive plants, especially perennial pepperweed, leading to the invasives outcompeting Suisun thistle; and (5) programs for the control or removal of nonnative invasive plants, which, if not conducted carefully, can damage Suisun thistle populations through the injudicious application of herbicides, by direct trampling, or through the accidental transport of invasive plant seeds to new areas. An additional threat that may require special management considerations or protection of the PCEs in Units 1 and 2 includes urban or residential encroachment from Suisun City to the north that could increase stormwater and wastewater runoff into these units. Alterations to channel water salinity and tidal regimes may also occur due to climate change and sea level rise.

2.28.6 Recovery and Management

The status of Suisun thistle and information about its biology, ecology, distribution, and current threats is available in the *Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California* (U. S. Fish and Wildlife Service 2013). The plan features Suisun thistle along with four other endangered species. Supplemental or updated information is provided in USFWS's 2009 5-year review for Suisun thistle (U. S. Fish and Wildlife Service 2009b). In 2009, USFWS recommended no change in the classification of Suisun thistle.

USFWS intends to conserve the geographic areas containing the physical and biological features that are essential to the conservation of the species, through the identification of the appropriate quantity and spatial arrangement of the primary constituent elements sufficient to support the life-history functions of the species. Because not all life-history functions require all the primary constituent elements, not all areas designated as critical habitat will contain all the primary constituent elements.

Downlisting of Suisun thistle will be achieved if the median area inhabited by this species is 2,000 acres, a total of 4,000 acres or more is permanently preserved, perennial pepperweed populations are reduced to less than 10 percent cover in Suisun Marsh, natural tidal cycles are restored at Hill Slough, and the ponded area at Rush Ranch is returned to periodic tidal flooding.

2.29 Valley Elderberry Longhorn Beetle

2.29.1 ESA Listing Status

The valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*) was listed as threatened by the USFWS in 1980 (45 FR 52803). A proposed rule to remove the species from federal listing was initiated in 2012 (77 FR 60237), and then withdrawn in 2014 due to habitat loss continuing to threaten the species (79 FR 55879).

2.29.2 Critical Habitat Designation

Critical habitat, designated at the time of listing in 1980 (45 FR 52803), includes two locations in Sacramento County along the American River where the densest known populations of the beetle occur. These areas are within the Action Area.

2.29.3 General Life-History and Habitat Requirements

The valley elderberry longhorn beetle is a small (0.5 to 0.8 inch) wood borer that depends on red or blue elderberry (*Sambucus* spp.) in every phase of its life cycle and is nearly always found on or close to its host plant along rivers and streams. Females are indistinguishable from the more widespread California elderberry longhorn beetle (*Desmocerus californicus californicus*). The elderberry is a common shrub component of riparian forests and adjacent nonriparian vegetation (valley oak and blue oak woodland, and annual grassland) along river corridors of the Central Valley.

Adult beetles feed on elderberry nectar, flowers, and foliage, and are generally active from March through June (77 FR 60237). As elderberry plants begin flowering in the spring, beetles begin to emerge from tunnels they bored as larvae through the shrub’s pith, roaming the shrubs, eating foliage and possibly flowers, until they mate.

Adults live from a few days to a few weeks after emerging, during which time they mate and lay their eggs (Talley et al. 2006). The females lay eggs, singly or in small groups, on the leaves or stems of living elderberry shrubs (Barr 1991). The larvae hatch in a few days and bore into living stems that are at least 1 inch (2.5 cm) in diameter, where they remain within the elderberry stem, feeding on the pith until they complete their development (Talley et al. 2006). Larvae eventually cut an exit hole out of the stem, and then plug the hole up from within using wood shavings. This allows the beetle to eventually exit the stem after it becomes an adult, as adults are not wood borers. Within the stem, the larva becomes a pupa, and finally emerges from its single exit hole as an adult between mid-March and mid-June (77 FR 60237).

Shrub characteristics and other environmental factors appear to have an influence on use by the valley elderberry longhorn beetle, with more exit holes found in shrubs in riparian than in nonriparian habitat types (Talley et. al. 2006). Occupancy of elderberry shrubs varies based on elderberry condition, water availability, elderberry density, and the health of the riparian habitat, indicating that healthy riparian systems supporting dense elderberry clumps are the primary habitat of the beetle (Barr 1991; Talley et al. 2006; Talley et al. 2007). However, some studies have demonstrated that valley elderberry longhorn beetles prefer elderberry shrubs with low to moderate levels of damaged stems (USFWS 2014).

2.29.4 Historical and Current Distribution and Abundance

The valley elderberry longhorn beetle is endemic to the Central Valley of California in moist valley oak woodlands along the margins of rivers and streams in the lower Sacramento and San Joaquin Valleys where its obligate larval host plant, elderberry grows. At the time of listing in 1980, the beetle was known from less than 10 locations on the American River, Putah Creek, and Merced River (USFWS 2009). Subsequent surveys have documented a broader distribution of the species and now it is known to occur from southern Shasta County in the north to Fresno County in the south, including the valley floor and lower foothills, and is generally found below 500 feet (152 meters) above mean sea level (USFWS 2017).

Most of the approximately 270 CNDDB element occurrences are based on observations of exit holes in elderberry stems or branches rather than direct observation of individual beetles; many of these occurrences predate 1997, which was the most recent, comprehensive rangewide survey by observers known to be qualified to detect occupancy of valley elderberry longhorn beetle (CDFW 2018; USFWS 2014). There are approximately 130 known occurrences of valley elderberry longhorn beetle in the San Joaquin and Sacramento Valleys that have been documented since the 1997 comprehensive rangewide survey (CDFW 2018). These occurrences have been found within 18 watersheds at 36 geographic locations (USFWS 2014).

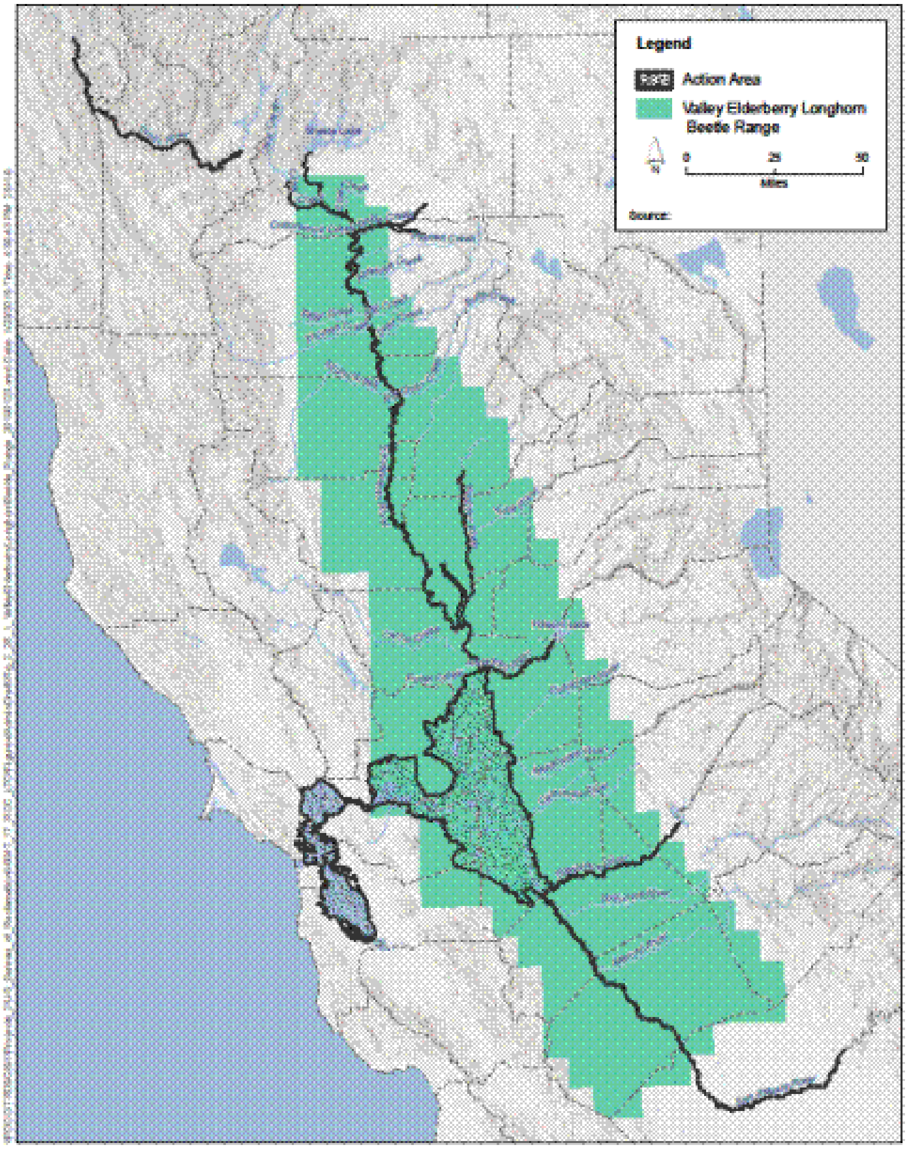

Figure 2.29-1. Valley Elderberry Longhorn Beetle Range

2.29.5 Limiting Factors, Threats, and Stressors

The valley elderberry longhorn beetle, though wide-ranging, is experiencing a long-term decline due to human activities that have resulted in widespread alteration and fragmentation of riparian habitats, and, to a lesser extent, upland habitats that support the beetle.

The primary threats to survival of the beetle include levee construction, stream and river channelization, removal of riparian vegetation, riprapping of shoreline, nonnative animals such as the Argentine ant (*Linepithema humile*), which may eat the early phases of the beetle, and recreational, industrial, and urban development. Insecticide and herbicide use in agricultural areas and along road right-of-ways may also be factors limiting the beetle's distribution.

Over the past 150 years, agricultural expansion and urbanization in the Central Valley increased. The need for water and flood protection spurred water development and reclamation projects, which reduced the expanse of riparian vegetation, including elderberry plants. Riparian vegetation was also removed for or impacted by the building of artificial levees and dams, river channelization, water diversion, and heavy groundwater pumping, thereby reducing these communities to small, isolated fragments.

Based on valley elderberry longhorn beetle research, this species occurs throughout the Central Valley in metapopulations, or discrete subpopulations that exchange individuals through dispersal or migration (Collinge et al. 2001). The subpopulations may shift spatially and temporally within riparian drainages, resulting in a patchwork of occupied and unoccupied habitat (USFWS 2017e). Valley elderberry longhorn beetles have limited dispersal capabilities, making it difficult to colonize unoccupied habitat areas. Therefore, the preservation of contiguous areas of suitable habitat is important for the longevity of this species. Climate change may change riparian flow regimes, which could remove valley elderberry longhorn beetle habitat and create it elsewhere, without the opportunity for the beetle to disperse to the new habitat.

Small population numbers of valley elderberry longhorn beetle host plants, and even lower numbers of occupied host plants, constitute a threat to the beetle at many locations, which, in turn, may result in small beetle population sizes. Additionally, low mobility, very small local populations, and isolation of habitat patches make beetle populations especially susceptible to extirpation with little chance of recolonization (Talley et al. 2006).

2.29.6 Recovery and Management

When the *Valley Elderberry Longhorn Beetle Recovery Plan* was developed (USFWS 1984), little information regarding the beetle’s life history, distribution, and habitat requirements was available to develop specific recovery objectives. The recovery plan did not include recovery criteria, but did include primary interim objectives that have since been at least partially met and include increased surveys, management of additional areas where the beetles have been identified, and some protections afforded to habitat areas (USFWS 2012). The majority of the beetle’s habitat along the Lower American River has been protected as part of the American River Parkway that includes both designated critical habitat and essential habitat (USFWS 2012).

Although riparian vegetation in the Central Valley has declined over time, a number of areas have been restored to accommodate the habitat needs and recovery of the valley elderberry longhorn beetle (that is, riparian vegetation that specifically contains elderberry shrubs). In the years since the time of listing, known locations of the beetle have increased through continued survey efforts, with a resultant significantly greater range size than was originally listed (USFWS 2012). In 2012, the USFWS proposed delisting the beetle from its threatened status under the ESA based on this increase, as well as past and ongoing riparian vegetation restoration and the persistence of elderberry shrubs in restored areas. However, the proposal was withdrawn in 2014 (79 FR 55879) because continued data acquisition indicated that threats to the species and its habitat have not been reduced to the point where the species no longer meets the statutory definition of a threatened species.

2.30 Vernal Pool Fairy Shrimp

2.30.1 ESA Listing Status and Critical Habitat Designation

Vernal pool fairy shrimp (*Branchinecta lynchi*) is listed as threatened under the ESA throughout its range (59 FR 48136). In September 2007, the USFWS published a 5-year review recommending that the species remain listed as threatened. On May 25, 2011, USFWS initiated a new 5-year review to determine if the species should be listed as endangered.

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). There is no designated critical habitat for vernal pool fairy shrimp within the action area.

2.30.2 General Life History and Habitat Requirements

Vernal pool fairy shrimp is entirely dependent 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 and 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 needed to maintain hydrological function of the temporary waters depends on 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). Temporary waters that are habitat for the vernal pool fairy shrimp range from low to moderate alkalinity (King et al. 1996; Eriksen and Belk 1999). Vernal pool fairy shrimp commonly co-occur with other fairy shrimp and vernal pool tadpole shrimp (*Lepidurus packardi*) (USFWS 2005).

Vernal pool fairy shrimp cysts can remain dormant in the soil when their habitats are dry. When the pools refill in the same or subsequent seasons, the cysts may hatch. The cyst bank in the soil may comprise cysts from several years of breeding (USFWS 2005, 2007). 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).

2.30.3 Historical and Current Distribution and Abundance

There is little information on the historical range of vernal pool fairy shrimp. The species currently occurs in a wide range of vernal pool habitats in the southern and Central Valley areas of California, and at two sites in Jackson County, Oregon (USFWS 2005). It is currently found at locations across the Central Valley 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 (USFWS 2005, 2007; CDFW 2019). There are 191 CNDDB element occurrences for vernal pool fairy shrimp in the action area (CDFW 2019).

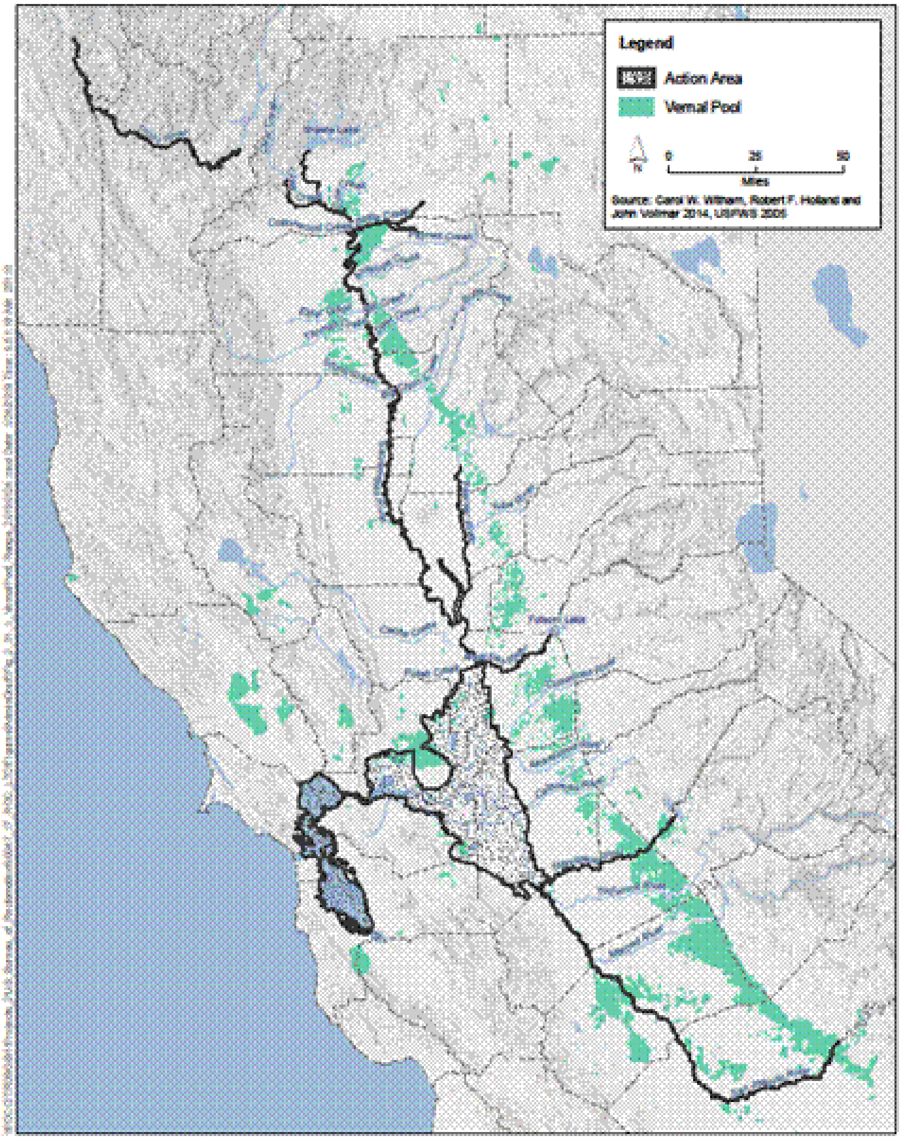

Figure 2.30-1. Vernal Pool Range

2.30.4 Limiting Factors, Threats, and Stressors

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* (USFWS 2005, 2007). Habitat loss and fragmentation are the largest threats to the survival and recovery of vernal pool species. Habitat loss generally is a result of agricultural conversion from rangelands to more developed land uses, while habitat fragmentation results from activities such as road development and other infrastructure projects (USFWS 2005).

Within habitat, grazing practices affect habitat quality. 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*), which, if unchecked, can increase thatch buildup, decrease ponding durations, and decrease the aquatic habitat available to the vernal pool fairy shrimp (USFWS 2007).

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 as habitat for vernal pool species (USFWS 2005).

Climate change affects 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 is causing localized cooling and drying, or warming with higher precipitation. Either scenario might result in adverse 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. A warming trend could increase inundation periods, but could also increase temperatures above levels needed for the species to hatch or reproduce (USFWS 2007).

Specific threats to vernal pool fairy shrimp habitat include the following (USFWS 2005).

- More than half of the known populations 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 (USFWS 2005, 2007).

- 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 State Route 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.

2.30.5 Recovery and Management

The Collinsville, Altamont Hills, and Jepson Prairie core recovery areas, which were developed in part for the recovery of the vernal pool fairy shrimp, encompass both designated critical habitat and essential habitat (USFWS 2005). Other recovery areas include the Western Riverside County and Santa Barbara vernal pool regions. For both regions, the recovery and management strategies involve protecting and reestablishing vernal pool habitat (USFWS 2005).

2.31 Vernal Pool Tadpole Shrimp

2.31.1 ESA Listing Status and Critical Habitat Designation

Vernal pool tadpole shrimp (*Lepidurus packardi*) was listed as endangered throughout its range under the ESA on September 19, 1994 (59 FR 48136). In September 2007, USFWS published a 5-year review recommending that the species remain listed as endangered. On May 25, 2011, USFWS initiated a new 5-year review to determine if the species should remain listed as endangered.

Critical habitat for vernal pool tadpole shrimp was designated on February 10, 2006 (71 FR 7118–7316) and does not occur in the action area.

2.31.2 General Life History and Habitat Requirements

Vernal pool tadpole shrimp occur in a 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 large (more than 100 acres) winter lakes (Helm 1998:134–138; Rogers 2001:1002–1005). These habitats 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).

Vernal pool tadpole shrimp cysts can remain dormant in the soil when their vernal pool habitats are dry. When the pools refill in the same or subsequent seasons, 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, 2007). 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 corresponds 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 is positively correlated with both depth and surface area (King et al. 1996). Vernal pool crustaceans generally have low rates of gene flow between separated sites, which is probably a result of the spatial isolation of their habitats and their reliance on passive dispersal mechanisms. Gene flow between pools within the same vernal pool complex is much higher, indicating that vernal pool tadpole shrimp populations are defined by vernal pool complexes rather than by individual vernal pools (USFWS 2005).

2.31.3 Historical and Current Distribution and Abundance

Historically, vernal pool tadpole shrimp probably did not occur outside of the Central Valley and Central Coast regions (USFWSe 2005). Currently, vernal pool tadpole shrimp occur in the Central Valley of California and in the San Francisco Bay Area. The species has a patchy distribution across the Central Valley from Shasta County southward to northwestern Tulare County (USFWS 2007). 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 (USFWS 2007; CDFW 2019). 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 (USFWS 2005, 2007). There are 136 occurrences of vernal pool tadpole shrimp in the action area (CDFW 2019).

2.31.4 Limiting Factors, Threats, and Stressors

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 (USFWS 2005). Habitat loss and fragmentation are the largest threats to the survival and recovery of vernal pool species. Habitat loss generally is a result of agricultural conversion from rangelands to more developed land uses, while habitat fragmentation results from activities such as road development and other infrastructure projects (USFWS 2005).

Within habitat, grazing practices affect habitat quality. 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*), which if unchecked can increase thatch buildup and decrease ponding durations and decrease the aquatic habitat available to the vernal pool tadpole shrimp (USFWS 2007).

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 vernal pools unsuitable as habitat for vernal pool species (USFWS 2005).

Climate change affects 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 is causing localized cooling and drying, or warming with higher precipitation. Either scenario might result in adverse effects on vernal pool invertebrate species. Cooling and drying trends could adversely affect vernal pool tadpole shrimp through decreased inundation periods that do not allow the species sufficient time to complete its life cycle. A warming trend could increase inundation periods, but could also increase temperatures above levels needed for the species to hatch or reproduce (USFWS 2007).

Specific threats to vernal pool tadpole shrimp habitat include the following (USFWS 2005).

- Encroachment of nonnative annual grasses on the San Francisco Bay National Wildlife Refuge in the Central Coast Region, and urban development 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 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.

2.31.5 Recovery and Management

The Collinsville, Altamont Hills, and Jepson Prairie core recovery areas, which were developed in part for the recovery of the vernal pool tadpole shrimp, encompass both designated critical habitat and essential habitat (USFWS 2005).

2.32 California Tiger Salamander

2.32.1 ESA Listing Status and Critical Habitat Designation

The USFWS listed the Central California DPS of California tiger salamander (which overlaps with the proposed action) as threatened on August 4, 2004 (50 FR 47212–47248). California tiger salamander is also listed as threatened under the California Endangered Species Act (CESA). On August 23, 2005, the USFWS designated approximately 199,109 acres (80,576 hectares) of critical habitat for the Central Valley DPS. The critical habitat is located in 19 California counties (70 FR 49380). No critical habitat overlaps with the action area.

2.32.2 General 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 (USFWS 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 (USFWS 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; USFWS 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, as cited 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 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 to 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 percent 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).

2.32.3 Historical and Current Distribution and Abundance

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 (CDFW 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 (CDFW 2013).

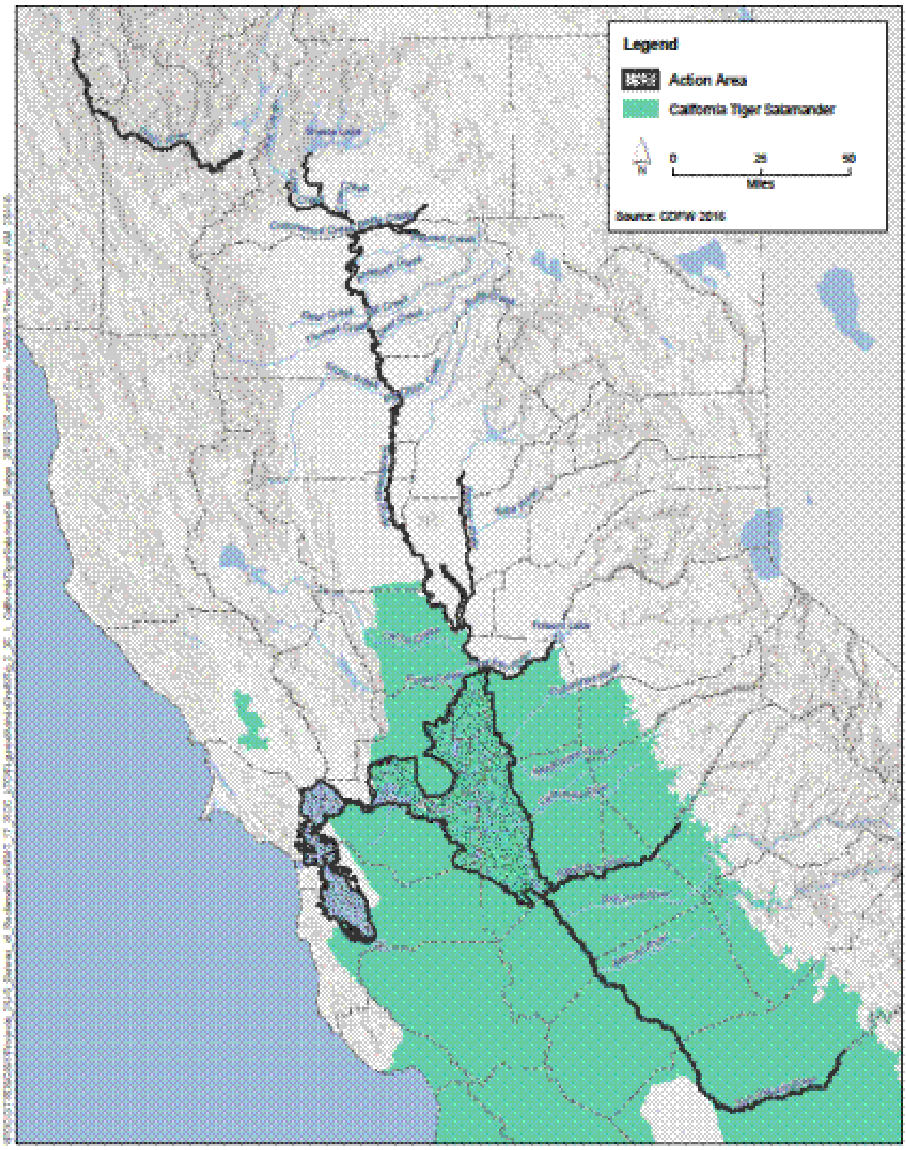

Figure 2.32-1. California Tiger Salamander Range

2.32.4 Limiting Factors, Threats, and Stressors

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; CDFW 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; USFWS 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 (Reclamation and DWR 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 control of burrowing mammals (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).

2.32.5 Recovery and Management

The strategy to recover the Central California tiger salamander focuses on alleviating the threat of habitat loss and fragmentation to increase population resiliency (ensure each population is sufficiently large to withstand stochastic events), redundancy (ensure a sufficient number of populations to provide a margin of safety for the species to withstand catastrophic events), and representation (conserve the breadth of the genetic makeup of the species to conserve its adaptive capabilities) (USFWS 2017). Recovery of this species can be achieved by addressing the conservation of remaining aquatic and upland habitat that provides essential connectivity, reduces fragmentation, and sufficiently buffers against encroaching development and intensive agricultural land uses. Appropriate management of these areas will also reduce mortality by addressing non-habitat related threats, including those from nonnative and hybrid tiger salamanders, other nonnative species, disease, and road mortality (USFWS 2017).

The range of the Central California tiger salamander has been classified into four recovery units: the Central Valley Unit, Southern San Joaquin Valley Unit, Bay Area Unit, and Central Coast Range Unit. The proposed action occurs within the Central Valley Unit, which comprises 12 Management Units across Yolo, Sacramento, Solano, eastern Contra Costa, northeast Alameda, San Joaquin, Stanislaus, Merced, western Amador, western Calaveras, and northwestern Madera Counties. The closest Management Unit to components of the proposed action is the Jepson Prairie Management Unit, located northeast of the Hills Slough Restoration project.

2.33 California Least Tern

2.33.1 ESA Listing Status and Critical Habitat Designation

The USFWS listed the California least tern as endangered on October 13, 1970 (35 FR 8491). California least tern is also listed as endangered under the CESA. On August 23, 2005, the USFWS designated approximately 199,109 acres (80,576 hectares) of critical habitat for the Central Valley DPS. Critical habitat has not been designated for this species.

2.33.2 General 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; USFWS 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 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 (USFWS 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).

2.33.3 Historical and Current Distribution and Abundance

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 (USFWS 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 2016 estimated 3,989 to 4,661 breeding pairs that established 4,746 nests and produced approximately 1,612 to 2,000 fledglings at 50 breeding sites across California (Frost 2017). Of these, only five breeding sites supporting a total of 570 nests (12 percent of the total) were reported from the San Francisco Bay Area (Frost 2017). 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 3,989 pairs in 2016 (Frost 2017), 4,202 in 2015 (Frost 2016), and 4,232 in 2014 (Frost 2015).

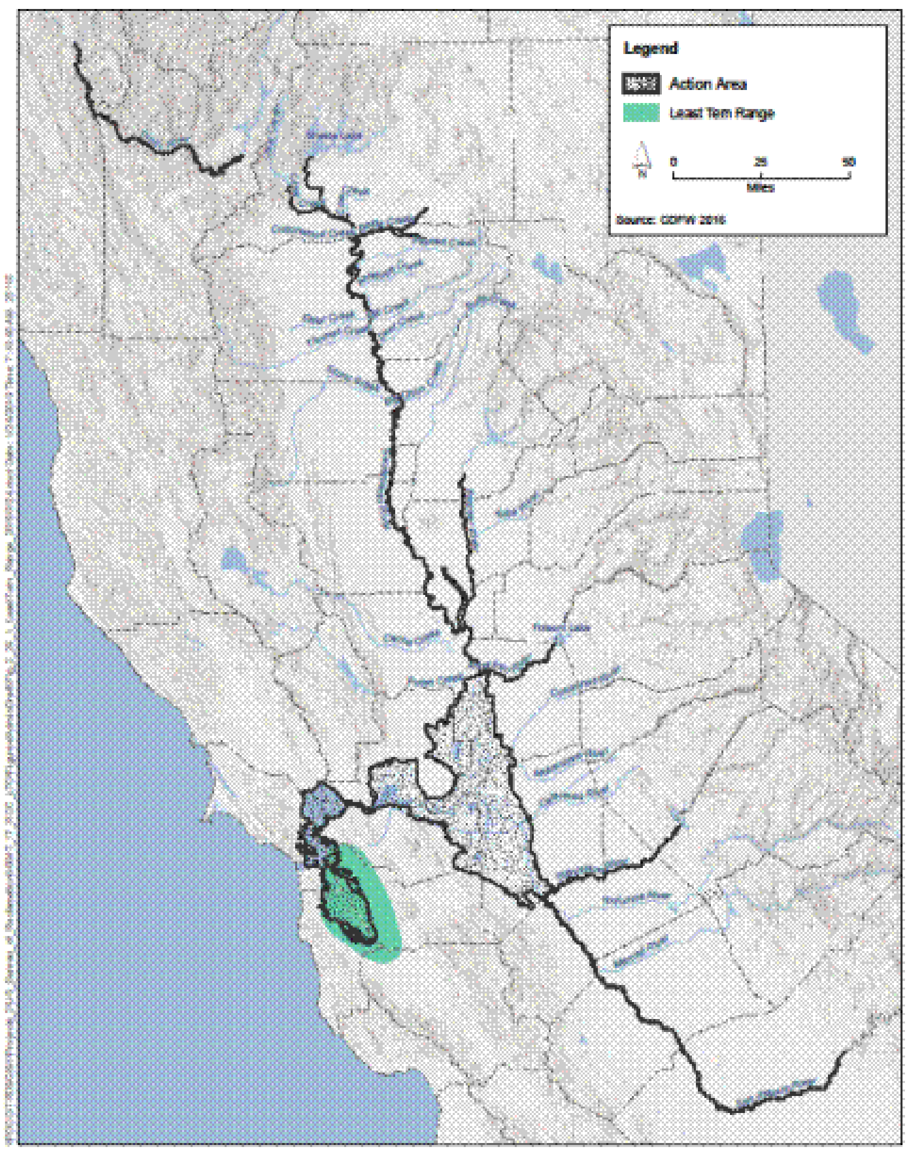

Figure 2.33-1. California Least Tern Range

2.33.4 Limiting Factors, Threats, and Stressors

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 (USFWS 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 (USFWS 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.

2.33.5 Recovery and Management

The 1985 recovery plan recommended developing and implementing management plans and programs for “secure” nesting habitat in Alameda, San Mateo, Santa Barbara, Ventura, and Los Angeles, Orange, and San Diego Counties. Management plans created for long-term site ecological security would focus on reducing perturbation, destruction, or pollution of nesting or foraging habitat (USFWS 1985). No plans have been completed for any of these areas.

2.34 California Red-Legged Frog

2.34.1 ESA Listing and Critical Habitat

The USFWS listed the California red-legged frog as threatened in 1996 (61 FR 25813) and published a final rule to revise the designated critical habitat in 2010 (75 FR 12816). The designated critical habitat areas contain physical and biological features (primary constituent elements) that are considered essential to the conservation of the species, and that may require special management considerations and protection. The primary constituent elements identified for the California red-legged frog are:

1) Aquatic Breeding Habitat. Standing bodies of fresh water (with salinities less than 7.0 parts per thousand), including: natural and constructed (e.g., stock) ponds, slow-moving streams or pools within streams, and other ephemeral or permanent water bodies that typically become inundated during winter rains and hold water for a minimum of 20 weeks in all but the driest of years.

2) Non-Breeding Aquatic Habitat. Freshwater habitats, as described above, that may or may not hold water long enough for the subspecies to hatch and complete its aquatic life cycle but that do provide for shelter, foraging, predator avoidance, and aquatic dispersal for juvenile and adult California red-legged frogs. Other wetland habitats that would be considered to meet these elements include plunge pools within intermittent creeks, seeps, quiet water refugia during high water flows, and springs of sufficient flow to withstand the summer dry period.

3) Upland Habitat. Upland areas within 200 feet (60 meters) of the edge of the riparian vegetation or dripline surrounding aquatic and riparian habitat and comprised of various vegetational series such as grasslands, woodlands, and/or wetland/riparian plant species that provides the frog shelter, forage, and predator avoidance. Upland features are also essential in that they are needed to maintain the hydrologic, geographic, topographic, ecological, and edaphic features that support and surround the wetland or riparian habitat. These upland features contribute to the filling and drying of the wetland or riparian habitat and are responsible for maintaining suitable periods of pool inundation for larval frogs and their food sources, and provide breeding, non-breeding, feeding, and sheltering habitat for juvenile and adult frogs (e.g., shelter, shade, moisture, cooler temperatures, a prey base, foraging opportunities, and areas for predator avoidance). Upland habitat can include structural features such as boulders, rocks and organic debris (e.g. downed trees, logs), as well as small mammal burrows and moist leaf litter.

4) Dispersal Habitat. Accessible upland or riparian dispersal habitat within designated units and between occupied locations within 0.7 mile (1.2 kilometers) of each other that allows for movement between such sites. Dispersal habitat includes various natural habitats and altered habitats such as agricultural fields, which do not contain barriers to dispersal. (An example of a barrier to dispersal is a heavily traveled road constructed without bridges or culverts.) Dispersal habitat does not include moderate to high density urban or industrial developments with large expanses of asphalt or concrete, nor does it include large reservoirs over 50 acres (20 hectares) in size, or other areas that do not contain those features identified in primary constituent elements 1, 2, or 3 as essential to the conservation of the subspecies.

In total, 34 critical habitat units covering approximately 450,288 acres (182,225 hectares) were designated. The critical habitat is located in Alameda, Butte, Contra Costa, El Dorado, Kern, Los Angeles, Marin, Merced, Monterey, Napa, Nevada, San Benito, San Luis Obispo, San Mateo, Santa Barbara, Santa Clara, Santa Cruz, Solano, Ventura and Yuba Counties.

2.34.2 General Life History and Habitat Requirements

The California red-legged frog is the largest native frog in the western United States. It is endemic to California and Baja California, Mexico. This species uses a variety of aquatic, riparian, and upland habitats, including ephemeral ponds, intermittent streams, seasonal wetlands, springs, seeps, permanent ponds, perennial creeks, constructed aquatic features, marshes, dune ponds, lagoons, riparian corridors, blackberry thickets, annual grasslands, and oak savannas. The common factor in all habitats used by California red-legged frogs is an association with a permanent water source.

Breeding sites have been documented in a wide variety of aquatic habitats. Larvae, juveniles, and adults have been observed inhabiting streams, creeks, ponds, marshes, sag ponds, deep pools and backwaters within streams and creeks, dune ponds, lagoons, estuaries, and artificial impoundments such as stock ponds. Breeding has been documented in these habitat types irrespective of vegetative cover. They often breed in artificial ponds with little or no emergent vegetation. The importance of riparian vegetation for this species is not well understood. It is thought that the riparian plant community may provide good foraging habitat and may facilitate dispersal in addition to providing pools and backwater aquatic areas for breeding.

California red-legged frogs disperse upstream and downstream of their breeding habitat to forage and seek shelter. Sheltering habitat for red-legged frogs potentially includes all aquatic, riparian, and upland areas within the range of the species and any landscape features that provide cover, such as existing animal burrows, boulders or rocks, organic debris such as downed trees and logs, and industrial debris. Agricultural features such as drains, watering troughs, spring boxes, abandoned sheds, or hay ricks may also be used. California red-legged frogs breed from November through March with earlier breeding records occurring in southern localities. Individuals occurring in coastal drainages are active year-round, whereas those found in interior sites are normally less active during the cold season. Females attach egg masses to emergent vegetation such as tule stalks, grasses, or willow roots just below the water surface. Larvae hatch 6 to 14 days following fertilization and spend most of their time concealed in submergent vegetation or detritus. Most larvae metamorphose into juvenile frogs 4 to 7 months after hatching, generally between July and September.

The diet of California red-legged frogs is highly variable. Similar to other frog species, larvae most likely consume diatoms, algae, and detritus (USFWS 2002). Invertebrates are the most common food items of adults. Vertebrates, such as Pacific tree frogs (*Hyla regilla*) and California mice (*Peromyscus californicus*), are frequently eaten by larger frogs. Feeding activity likely occurs along the shoreline and on the surface of the water.

2.34.3 Historical and Current Distribution and Abundance

The California red-legged frog has sustained a 70-percent reduction in its geographic range as a result of several factors acting singly or in combination (Jennings et al. 1992). Only a few drainages are currently known to support California red-legged frogs in the Sierra Nevada foothills, compared to more than 60 historical records. In southern California, the California red-legged frog has essentially disappeared from the Los Angeles area south to the Mexican border; the only known population in Los Angeles County is in San Francisquito Canyon on the Angeles National Forest (USFWS 2011).

Based on the best available information at the time of listing, the historic range of the California red-legged frog was described as extending along the coast from the vicinity of Point Reyes National Seashore in Marin County, and inland from the vicinity of Redding in Shasta County, southward to northwestern Baja California, Mexico (61 FR 25814). The listing rule described an intergrade zone between the California red-legged frog and the closely related (and nonlisted) northern red-legged frog (*Rana aurora*; formerly *Rana aurora aurora*) that extended approximately from the Walker Creek watershed in Marin County north to southern Mendocino County. Recent research on the genetics of red-legged frogs indicates that the intergrade zone between the California red-legged frog and the northern red-legged frog likely occurs within a narrower geographic area than previously known, and that the range of the California red-legged frog extends about 60 miles (100 kilometers) farther north (USFWS 2011). The California red-legged frog was probably extirpated from the floor of the Central Valley prior to 1960: the last record of a reproducing population on the valley floor is from the vicinity of Gray Lodge Wildlife Area (Butte County) around 1947, although this record is unverified (USFWS 2002). The species is therefore unlikely to occur in the action area. Reclamation has conducted surveys for California red-legged frog in the action area, and survey results were negative.

2.34.4 Limiting Factors, Threats and Stressors

Factors associated with declining populations of the California red-legged frog include degradation and loss of its habitat through agriculture, urbanization, mining, overgrazing, recreation, timber harvesting, nonnative plants, impoundments, water diversions, degraded water quality, use of pesticides, and introduced predators. The reason for decline and degree of threats vary by geographic location. California red-legged frog populations are threatened by more than one factor in most locations (USFWS 2011).

2.34.5 Recovery and Management

In 2002, USFWS published a recovery plan for the California red-legged frog (USFWS 2002). USFWS initiated a 5-year Status Review of California red-legged frog in June 2018 (USFWS 2018).

2.34.6 Monitoring and Research Programs

Monitoring of the California Red-legged Frog, Rana aurora draytonii, within Properties of the Los Baños Wildlife Area Complex, 2008, by CDFG December 2008

California Department of Fish and Wildlife has been conducting California red-legged frog surveys on the San Luis Reservoir and Upper Cottonwood Creek Wildlife Areas since 2001. Between January and July of 2008, they performed frog surveys on these properties, and additionally at Lower Cottonwood Creek Wildlife Area and Little Panoche Reservoir Wildlife Area at a total of 24 sites. Monitoring consisted primarily of daytime visual surveys and a limited number of night surveys. They were able to confirm frog presence and breeding activity at several sites on Upper Cottonwood Creek Wildlife Area, and observed frog calls during breeding season at Little Panoche Reservoir Wildlife Area. Habitat quality, restoration possibilities, future monitoring, and frog health continue to be key factors in CDFW's monitoring efforts.

One study documented only 20 Sierra Nevada localities and one Cascades Mountains locality where *R. draytonii* occurred between 1916 and 1975, extending from Tehama County southeast about 405 kilometers to Madera County. The elevation range of most of the historical localities was 200 to 900 meters (about 40 kilometers from lower to upper elevation), but three apparently extirpated populations that may have originated from deliberate translocations occurred at 1,500 to 1,536 meters elevation in Yosemite National Park. They surveyed directly or within 5 kilometers of 20 of the 21 historical Sierra Nevada/Cascades *R. draytonii* localities and found that at least one of these historical populations persists today, in large numbers. They also discovered or confirmed six new Sierra Nevada *R. draytonii* populations and individual frogs at three additional new sites, for a total of seven recent populations and three recent single-specimen occurrences extending from Butte County southeast about 275 kilometers to Mariposa County. Historically, *R. draytonii* in the Sierra Nevada probably bred in stream pools, which tend to be small with limited forage and thus may have constrained the historical size and number of Sierra Nevada *R. draytonii* populations. Since the 1850s, constructed ponds sometimes capable of supporting large *R. draytonii* populations have supplemented stream pool breeding habitat. Excluding the southernmost and Yosemite historical localities, the current range of Sierra Nevada *R. draytonii* differs little from the historical range, and further surveys may reveal additional surviving Sierra Nevada *R. draytonii* populations (Barry and Fellers 2013).