Appendix 6A
Environmental Setting Background Information

6A.1 Fish and Aquatic Resources Species Descriptions

6A.1.1 Delta Smelt

6A.1.1.1 Legal Status

Delta Smelt (*Hypomesus transpacificus*) was listed as a threatened species under the California Endangered Species Act (CESA) in 1993. An emergency petition was filed in February 2007 with the California Fish and Game Commission to elevate the status of Delta Smelt from threatened to endangered under CESA (The Bay Institute et al. 2007). On March 4, 2009, the California Fish and Game Commission elevated the status of Delta Smelt to endangered under CESA. A 12-month finding on a petition to reclassify the Delta Smelt as an endangered species under the federal Endangered Species Act (ESA) was completed on April 7, 2010. After reviewing all available scientific and commercial information, the U.S. Fish and Wildlife Service (USFWS) determined that reclassifying Delta Smelt from threatened to endangered was warranted but was precluded by other higher-priority listing actions (U.S. Fish and Wildlife Service 2010). The recommendation to reclassify Delta Smelt from threatened to endangered was confirmed by USFWS in November 2020, with USFWS noting that reclassifying the species to endangered status will not substantively increase protections for the Delta Smelt, but rather more accurately classify the species given its current status (85 *Federal Register* [FR] 73175).

6A.1.1.2 Life History and General Ecology

Delta Smelt are endemic to the San Francisco Estuary and Sacramento–San Joaquin Delta (Delta) where they occupy open-water habitats in Suisun Bay, Suisun Marsh, and the Delta, generally away from shore but also nearer to shore to facilitate migration or to remain within preferred habitats (Feyrer et al. 2013; Bennett and Burau 2015). On occasion, Delta Smelt distribution can extend 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 the Napa River and San Francisco Bay. Delta Smelt is primarily an annual species, completing its life cycle in one year. In captivity, Delta Smelt can survive to spawn at two years of age (Lindberg et al. 2013), but age-2 Delta Smelt are now rare in the wild (Bennett 2005; Damon et al. 2016).

Delta Smelt complete their entire life cycle within the low-salinity zone of the Upper San Francisco Estuary, in the tidal freshwater region of the Cache Slough Complex or move between the two regions of freshwater and low salinity (Bennett 2005; Sommer and Mejia 2013; Hobbs et al. 2019). Komoroske et al. (2016) found Delta Smelt can acclimate to salinities greater than 6 parts per thousand (ppt) in the laboratory, but observations of Delta Smelt presence in waters having salinities exceeding 6 ppt in the wild are comparatively rare (e.g., 92 percent of fish caught in the Fall Midwater Trawl [FMWT] survey are at salinity ≤6 ppt; Komoroske et al. 2016). This could be because the osmoregulatory costs at high salinities are too high to support growth and survival (Komoroske et al. 2016); however, a bioenergetic study by Hammock et al. (2017) on Delta Smelt did not find any significant effect of salinity for even the highest salinity tested (12 ppt). The discrepancy between field observations and laboratory observations may be evidence that Delta Smelt distribution in the wild is due to a factor or factors other than salinity (U.S. Fish and Wildlife Service 2019:122).

Although Delta Smelt are physiologically euryhaline (i.e., can tolerate salinities of 0.4–34.0 ppt), the cumulative costs associated with physiological adjustments required to achieve homeostasis across a large, fluctuating salinity gradient may be higher than the continual maintenance cost for homeostasis within the low-salinity zone (Komoroske et al. 2016:976).

Delta Smelt spawning occurs predominantly at night with several males attending a female that broadcasts her eggs onto bottom substrates (Bennett 2005; Lindberg et al. 2020; Tsai et al. 2021a, 2021b). Although preferred spawning substrates are unknown, spawning habits of the Delta Smelt's closest relative, Surf Smelt (*Hypomesus pretiosus*), in addition to experimental trials, suggest that sand or small pebbles may be the preferred substrate (Bennett 2005; Hirose and Kawaguchi 1998; Lindberg et al. 2020). Hatching success peaks at water temperatures of 15 degrees Celsius (°C) to 16 °C (59.0 degrees Fahrenheit [°F] to 60.8 °F), ceasing when water temperatures exceed 20 °C (68 °F) (Bennett 2005). Water temperatures suitable for spawning occur most frequently March through May, but ripe female Delta Smelt have been observed as early as January and larvae have been collected as late as July (Damon et al. 2016). Most spawning occurs at 9 °C to 18 °C (48.2 °F to 64.4 °F) (Damon et al. 2016), with the duration of temperatures within this window potentially affecting the number of times individual Delta Smelt females could spawn given a spawning frequency of around once per month (Damon et al. 2016). Damon et al. (2016) estimated the minimum size at maturity to be 55 millimeter (mm) fork length (FL) and during Spring Kodiak Trawl (SKT) sampling from 2003 to 2015 within the thermal spawning window found many fish greater than this minimum by February, most fish above the minimum by March, and all fish above the minimum by April. Delta Smelt appear to have one spawning season for each generation, which makes the timing and duration of the spawning season important every year. Achievement of spawning size by April would result in only one spawn per female, with subsequent spawning events after April being rare except in exceptional years when the thermal spawning window extends past May (Damon et al. 2016). Kurobe et al. (2016) found that eggs (oocytes) matured from February to April during their study from November 2011 through April 2012. Prior studies suggest that spawning location changes based on hydrological conditions (reviewed by Bennett 2005:13). However, a more recent study indicated most migrations from juvenile and subadult rearing locations to spawning areas occurs by January, spawning habitat locations are relatively constant within and between years, and no substantial further restructuring of the population at regional scales occurs after fish move to spawning locations (Polansky et al. 2018). The main spawning locations are in the lower Sacramento and San Joaquin rivers, and the north Delta, including the Cache Slough Complex and Sacramento Deep Water Ship Channel (Polansky et al. 2018).

Although adult Delta Smelt can spawn more than once, as noted above, most spawning is complete by the time water temperature reaches 18 °C (64.4 °F) (Damon et al. 2016). The egg stage averages about 10 days before the embryos hatch into larvae (Bennett 2005). The larval stage averages about 30 days. Metamorphosing post-larvae appear in monitoring surveys from April into July during most years (Bennett 2005). By July, most Delta Smelt have reached the juvenile life stage. Delta Smelt collected during the fall are considered subadults. Sampling for adults by the SKT survey and Enhanced Delta Smelt Monitoring (EDSM) program begins in December/January, which generally aligns with the time period at which maturity is reached (Kurobe et al. 2016). Delta Smelt disperse landward after the first significant precipitation event of the winter (i.e., First Flush), and stage in these areas until they attain sexual maturity (Grimaldo et al. 2009; Sommer et al. 2011; Polansky et al. 2018). Some adults exhibit very limited dispersal during the spawning season (Murphy and Hamilton 2013; Polansky et al. 2018).

In the wild, larval Delta Smelt are presumed to be surface-oriented, exhibiting greater dispersion during the night (Bennett et al. 2002). In laboratory experiments, newly hatched larval Delta Smelt can manipulate their position in tanks, but there is no evidence they can swim against prevailing currents. Juveniles vary their position in the water column with respect to tides, water quality, and bathymetry; presumably, these movements facilitate maintenance in favorable habitats (Feyrer et al. 2013). Adults appear to use tidal migration or move horizontally toward shore during spawning migrations to upstream habitats (Bennett and Burau 2015). Laboratory studies of Delta Smelt measuring 32–68 mm standard length (SL) reported mean critical swimming velocities of about 28 centimeters (cm) per second, generally comparable to other fishes of similar size (Swanson et al. 1998).

From March through June, larval Delta Smelt rely heavily on juvenile and adult life stages of the calanoid copepods *Eurytemora affinis* and *Pseudodiaptomus forbesi*, as well as cladocerans (Nobriga 2002; Hobbs et al. 2006; Slater and Baxter 2014) and the copepod *Sinocalanus doerrii*. Nobriga (2002) found Delta Smelt larvae preferred *E. affinis* and *P. forbesi*, consuming these prey species in greater proportion than available in the environment, consistent with the findings of Slater and Baxter (2014). Such selection was not noted for other zooplankton prey species. For example, Slater and Baxter (2014) found neutral selection for *S. doerrii* and neutral to negative (i.e., consumption in lower proportion than available in the environment) selection for the small cyclopoid copepod *Limnoithona tetraspina* and copepod nauplii, which were consumed only when extremely numerous in the environment. Regional differences in food use occurs; *E. affinis* and *P. forbesi* are major prey items downstream in the low-salinity zone and *S. doerrii* and cyclopoid copepods are major prey items upstream into the Cache Slough Complex. Juvenile Delta Smelt (June through September) rely extensively on calanoid copepods such as *E. affinis* and *P. forbesi*, especially in fresh water (salinity <1 ppt) and the Cache Slough Complex, but there is great variability among regions (Interagency Ecological Program Management Analysis and Synthesis Team 2015). Larger fish are also able to take advantage of mysids and amphipods (Moyle et al. 1992; Lott 1998; Feyrer et al. 2003; Slater et al. 2019), and Slater and Baxter's (2014) study during April–September found cladocerans in the diet from May onwards. The presence of several epibenthic species in diets indicates food sources for Delta Smelt are not solely connected to pelagic pathways.

6A.1.1.3 Distribution and Abundance

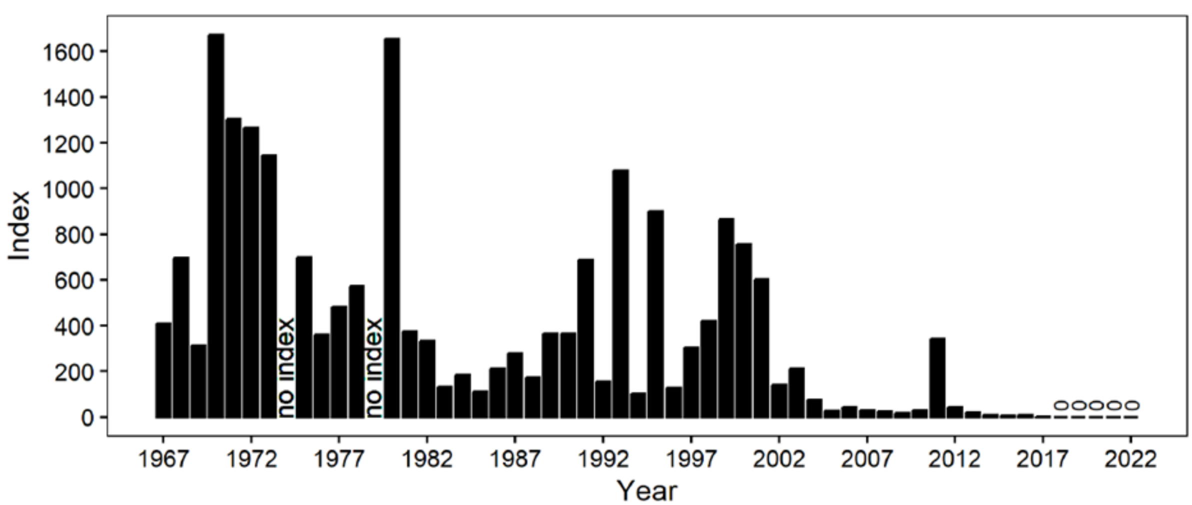
The California Department of Fish and Wildlife (CDFW) conducts four annual fish surveys from which it develops indices of Delta Smelt relative abundance. Each survey has variable capture efficiency (Mitchell et al. 2017). In each, there is high frequency of zero catches of Delta Smelt mostly due to the species' rarity (Latour 2016; Polansky et al. 2018), but also because surveys are carried out independent of other factors that affect catch, such as tide (Bennett and Burau 2015) and channel location (Feyrer et al. 2013), and because there may be highly localized and ephemeral aggregations, as suggested by infrequent large catches (Polansky et al. 2018). Detection probability decreases with increasing water clarity (Peterson and Barajas 2018) and relatively high numbers of Delta Smelt may occur in areas without long-term sampling stations, such as portions of the Cache Slough Complex (Murphy and Weiland 2019). Mahardja et al. (2017) found a high detection probability of Delta Smelt larvae/early juveniles by the 20-mm survey at the level of replication (three tows) at each site.

USFWS implemented a new smelt monitoring program in 2016, called the EDSM program. This new program is used to measure the abundance and distribution of all life stages of Delta Smelt using a generalized random tessellation stratified design. Delta Smelt population estimates are now derived from this survey.

The distribution of the Delta Smelt population varies with life stage, season, and environmental conditions (Bennett 2005; Sommer and Mejia 2013; Murphy and Hamilton 2013; Hobbs et al. 2019). Subadult and adult Delta Smelt typically make landward movements soon after first flush periods of initial winter precipitation and runoff, when turbidities elevate (Grimaldo et al. 2009). During extreme Wet years, some adults may move seaward into San Francisco Bay and the Napa River (see summary by Sommer and Mejia 2013). Larval Delta Smelt can be broadly distributed depending on hydrologic conditions during March and April. During Wet years, larval Delta Smelt are distributed farther seaward than in drier years (Sommer and Mejia 2013). Juvenile Delta Smelt distribution is generally greatest in the North Delta Arc (see Moyle et al. 2018:44), which extends from Cache Slough to Suisun Bay and Suisun Marsh (Merz et al. 2011; Murphy and Hamilton 2013).

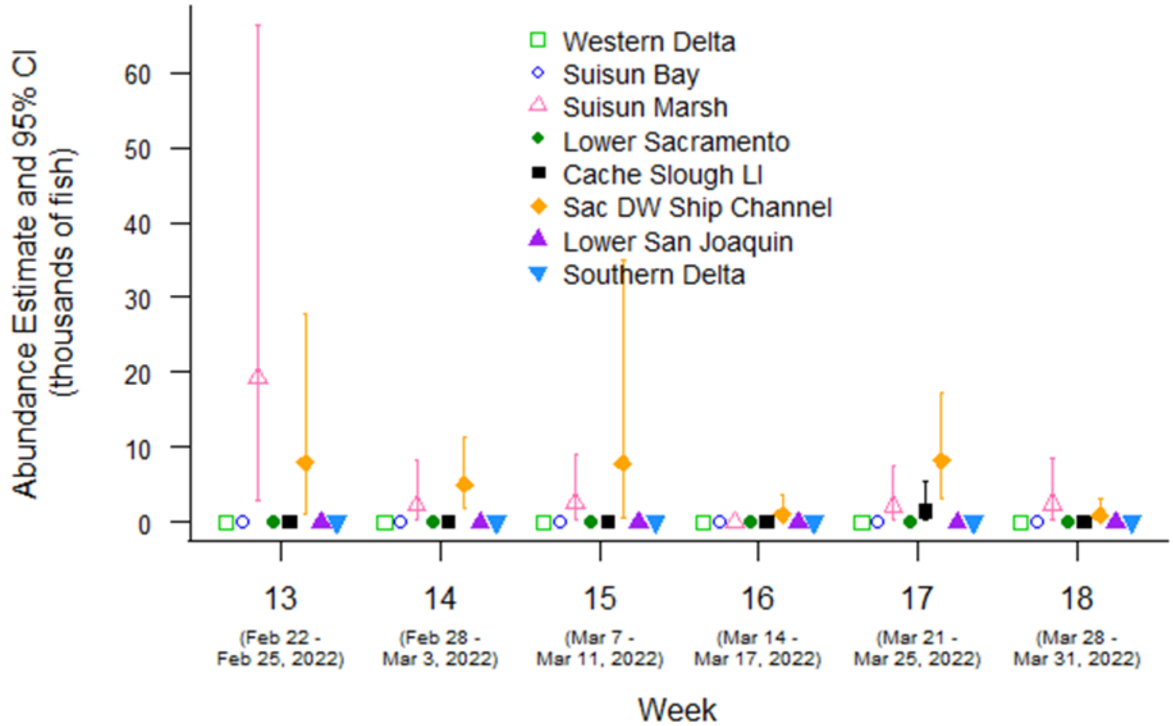
Trawl abundance indices indicate that the relative abundance of Delta Smelt has declined substantially since the 1980s. The observed decline in Delta Smelt abundance generally is consistent with declines of other pelagic species (e.g., Longfin Smelt [*Spirinchus thaleichthys*], Threadfin Shad [*Dorosoma pentenense*], and juvenile Striped Bass [*Morone saxatilis*]) in the Delta (Sommer et al. 2007a; Baxter et al. 2010; Stompe et al. 2020).

The CDFW FMWT Delta Smelt annual abundance index has been zero every year from 2018 through 2022 (Water Years [WY] 2019–2023), the lowest on record (Figure 6A-1). All CDFW relative abundance indices show a declining trend since the early 2000s. As previously noted, USFWS implemented a new smelt monitoring program in 2016, called the EDSM program. This new program is used to measure the abundance and distribution of all life stages of Delta Smelt using a generalized random tessellation stratified design. Delta Smelt population estimates are now derived from this survey, with abundance suggested to be several thousand fish or fewer in recent years (see Figure 6A-2 for 2022 as an example).



Source: White 2022.

Figure 6A-1. Time Series of the Fall Midwater Trawl Water Years 1967–2022 Abundance Index for Delta Smelt



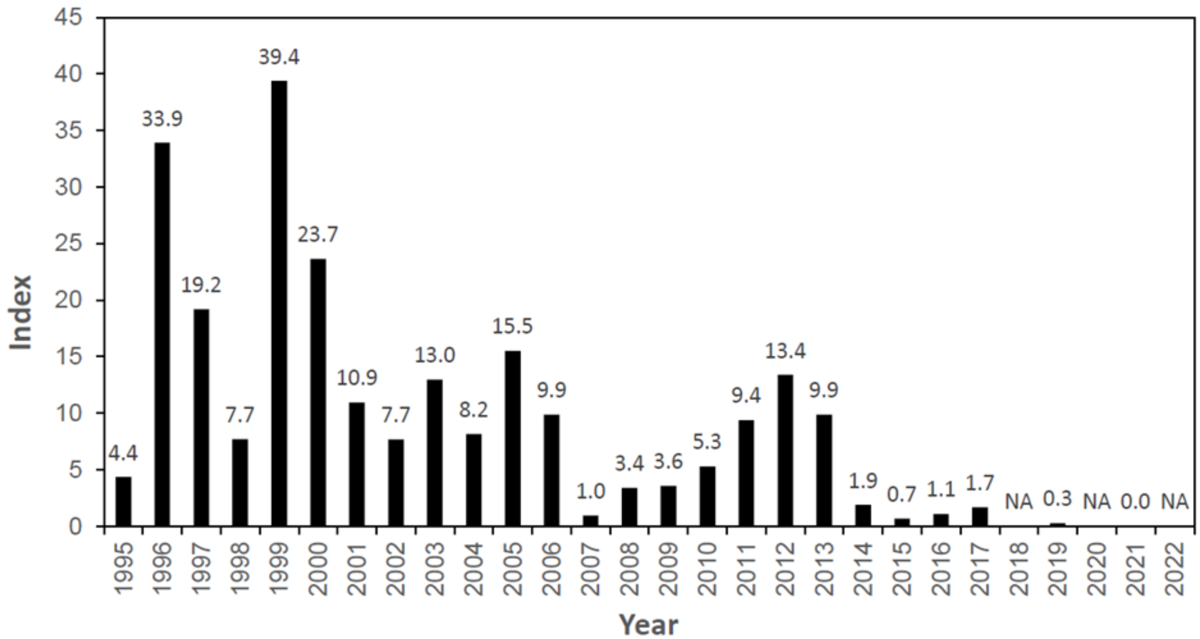
Source: U.S. Fish and Wildlife Service 2022a.

Figure 6A-2. Abundance Estimates for Delta Smelt from Enhanced Delta Smelt Monitoring Program Phase 1, Weeks 13–18, Water Year 2022

The continued low spawning stock of Delta Smelt relative to historical numbers suggests the population would continue to be vulnerable to stochastic events and continued human-caused alteration of the Delta. As described in detail by CDFW (2021:10), the Experimental Release of Delta Smelt Project proposes to annually release up to 120,000 adult equivalents of surplus cultured Delta Smelt each year into a portion of the current range of the species for a three-year period (2021–2024). For example, in WY 2023, nearly 44,000 marked Delta Smelt reared at the University of California, Davis Fish Conservation and Culture Laboratory were released into the Sacramento River at Rio Vista and the Sacramento Deep Water Ship Channel during late November and mid to late January (Columbia Basin Research, University of Washington 2023a). The purpose of the Experimental Release of Delta Smelt Project is to inform the feasibility and design of potential future supplementation efforts. The cultured Delta Smelt are propagated at the University of California Davis Fish Conservation and Culture Laboratory in Byron, California.

Considerable progress has been made on estimating absolute abundance of Delta Smelt, including adults (Polansky et al. 2019). These estimates are affected by factors such as fish behavior and local habitat features, such as turbidity influencing catchability (Polansky et al. 2019:721–722). However, turbidity may only have limited effects on catchability according to a recent simulation analysis (Tobias 2021). The continued low spawning stock of Delta Smelt relative to historical estimates suggest the population continues to be vulnerable to key threats (described below), especially when these stressors are occurring in consecutive years (e.g., drought) or across sequential life stages (e.g., high water temperatures).

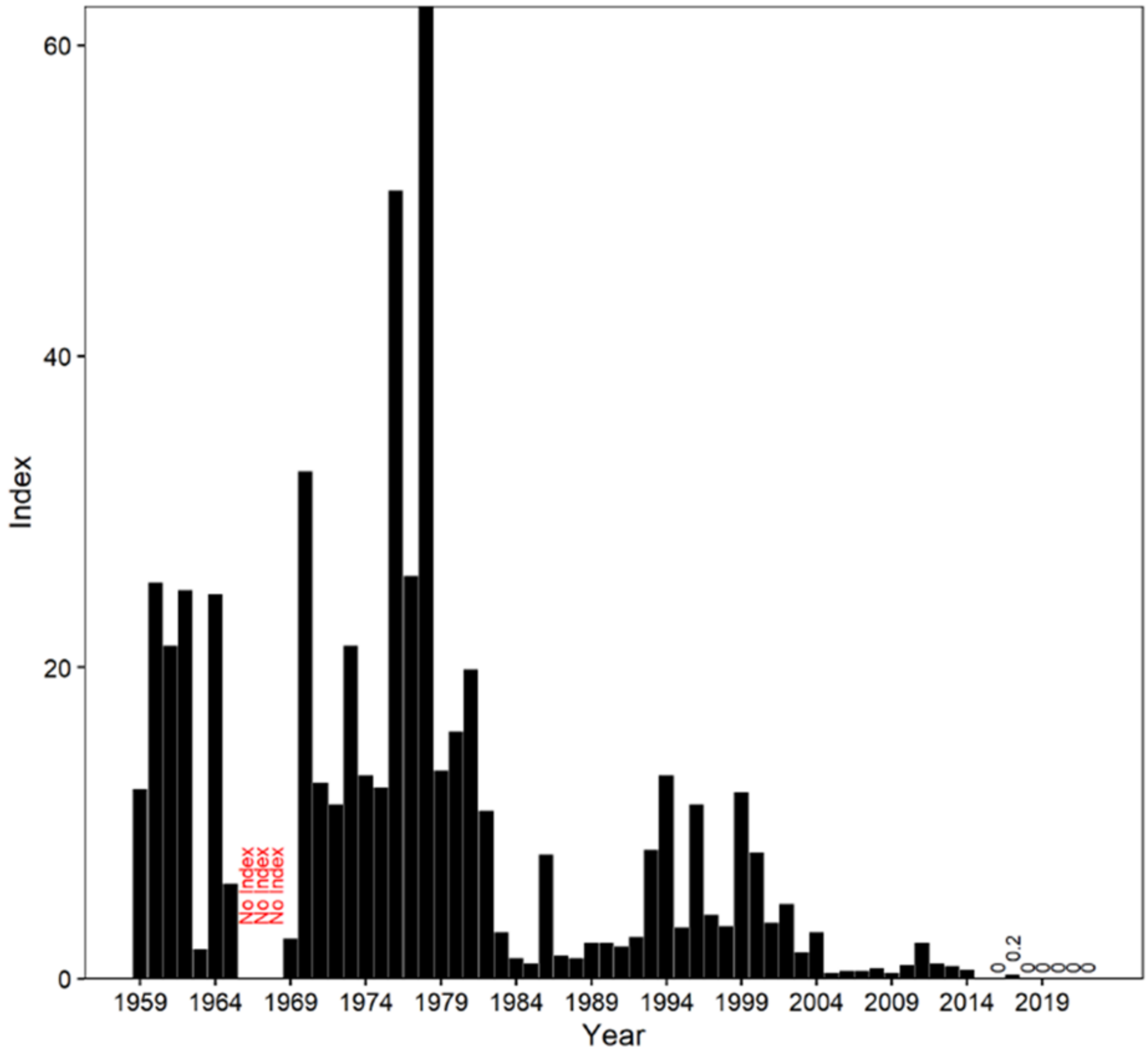
The Summer Townet Survey (STN) and FMWT abundance indices for Delta Smelt have documented long-term declines, while the newer 20-mm and SKT abundance index trends generally are consistent with the more recent trends shown by the older surveys (Figure 6A-3 and Figure 6A-4). During the period of record, Delta Smelt relative abundance has declined from peak levels observed during the 1970s. The STN and FMWT abundance indices declined rapidly during the early 1980s, increased somewhat during the 1990s, and then collapsed in the early 2000s. Since 2005, the STN and the FMWT have produced indices that reflect less year-to-year variation than their 20-mm analog, but overall, the trends in both sets of indices are similar. During the past decade, the index has continued to decrease and the most recent values for the indices (FMWT, STN, and 20-mm) were zero, or not calculable because of insufficient catch (Figure 2-1 and Figures 2-3a and 2-3b).



Source: Damon and Mora 2022.

Note: NA indicates years where index was not calculable because of insufficient catch.

Figure 6A-3. Time Series of the 20-mm Survey (Water Years 1995–2022) Abundance Index for Delta Smelt



Source: Malinich 2022.

Note: NA indicates years where index was not calculable because of insufficient catch.

Figure 6A-4. Time Series of the Summer Townet Survey (Water Years 1959–2022) Abundance Index for Delta Smelt

The general distribution of Delta Smelt is well understood due to its limited geographic distribution (Moyle et al. 1992; Bennett 2005; Hobbs et al. 2006, 2007). Potentially suitable Delta Smelt habitat in the Delta is geographically limited, generally characterized by high turbidity and tidally influenced low-salinity conditions. Suitable spawning and migration habitat are occupied seasonally. In most years, surveys do not sample Delta Smelt at the fringes of their geographic distribution. 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. This distribution represents a range of salinity from approximately 0 ppt to 20 ppt. However, most Delta Smelt observed in the extensively surveyed San Francisco Estuary have been collected from locations within the legal critical habitat delineation. In addition, all habitats known to be occupied year-round by Delta Smelt occur within the conditions defined in the critical habitat rule (59 FR 852). Each year, the distribution of Delta Smelt seasonally expands following first flush, when adults disperse in response to increased winter flows and associated increases in turbidity and decreases in water temperature. 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 (OMR) in the south Delta each year, but face a high risk of entrainment when they do (Grimaldo et al. 2009; Smith et al. 2020).

The frequency of occurrence (percentage of samples) of Delta Smelt by life stage and region from monitoring in the Delta, as assessed by Merz et al. (2011), is provided in Table 6A-1.

Table 6A-1. Average Annual Frequency (%) of Delta Smelt Occurrence by Life Stage, Interagency Ecological Program Monitoring Program, and Region

	Average Annual Frequency (%)										
Life Stage:	**Larvae (<15 mm)**	**Sub-Juvenile (≥15, <30 mm)**		**Juvenile (30–55 mm)**			**Subadult (>55 mm)**	**Mature Adults (>55 mm)**		**Pre-Spawning**	**Spawning[a]**
Monitoring Program:	**20-mm**	**20-mm**	**STN**	**20-mm**	**STN**	**FMWT**	**FMWT**	**BS**	**BMWT**	**SKT**	**SKT**
Years of Data Used:	**1995–2009**	**1995–2009**	**1995–2009**	**1995–2009**	**1995–2009**	**1995–2009**	**1995–2009**	**1995–2009**	**1995–2006**	**2002–2009**	**2002–2009**
Region/Time Period:	**Apr–Jun**	**Apr–Jul**	**Jun–Aug**	**May–Jul**	**Jun–Aug**	**Sep–Dec**	**Sep–Dec**	**Dec–May**	**Jan–May**	**Jan–Apr**	**Jan–May**
San Francisco Bay	NS	NS	NS	NS	NS	NS	NS	0.0	0.0	NS	NS
West San Pablo Bay	NS	NS	NS	NS	NS	0.2	0.0	0.0	1.2	NS	NS
East San Pablo Bay	0.0	1.0	0.0	2.8	3.6	0.7	0.6	NS	2.7	NS	NS
Lower Napa River	7.3	7.7	3.3	13.3	14.0	1.7	0.8	NS	NS	14.3	11.8
Upper Napa River	11.6	21.2	NS	12.0	NS	NS	NS	NS	NS	NS	NS
Carquinez Strait	5.7	9.3	1.1	24.4	33.7	1.9	3.3	NS	5.4	16.7	0.0
Suisun Bay (SW)	17.8	18.3	1.3	17.5	26.9	4.3	4.3	NS	4.3	23.3	5.6
Suisun Bay (NW)	2.2	8.9	1.1	21.7	34.8	7.3	10.0	NS	8.7	23.3	5.6
Suisun Bay (SE)	19.5	24.9	11.0	20.9	45.7	11.0	12.1	NS	6.5	28.3	6.9
Suisun Bay (NE)	17.8	19.2	33.6	29.7	66.7	20.3	29.3	NS	28.3	48.3	13.9
Grizzly Bay	16.3	27.6	17.9	42.9	72.8	15.0	19.6	NS	30.4	30.0	5.6
Suisun Marsh	21.4	33.6	14.2	18.5	19.2	22.8	27.2	NS	NS	62.0	23.1
Confluence	35.7	41.6	25.7	29.2	36.1	20.2	24.5	1.8	17.4	30.0	10.4
Lower Sacramento River	16.5	37.0	43.3	26.2	55.5	22.9	37.1	NS	18.8	54.4	17.8
Upper Sacramento River	10.8	8.2	1.3	0.0	0.0	2.7	8.0	5.8	16.7	21.7	15.3
Cache Slough and Ship Channel	17.2	47.3	NS	54.3	NS	9.8	26.7	NS	NS	33.9	21.1
Lower San Joaquin River	28.0	24.5	4.1	5.1	5.6	2.6	3.5	0.9	12.6	30.6	9.7
East Delta	14.6	8.8	0.0	1.2	0.0	0.0	0.0	1.6	NS	5.7	2.3
South Delta	18.4	10.8	0.0	1.4	0.3	0.0	0.0	0.3	NS	7.1	1.1
Upper San Joaquin River	NS	NS	NS	NS	NS	NS	NS	0.2	NS	NS	NS
Sacramento Valley	NS	NS	NS	NS	NS	NS	NS	0.2	NS	NS	NS

Source: Merz et al. 2011

20-mm = 20-millimeter Townet; BMWT = Bay Midwater Trawl; BS = Beach Seine; FMWT = Fall Midwater Trawl; KT = Kodiak Trawl; NS = indicates no survey conducted in the given life stage and region; NE = northeast; NW = northwest; SKT = Spring Kodiak Trawl; STN = Summer Townet; SE = southeast; SW = southwest.

[a] Gonadal stages of male and female Delta Smelt found in SKT database were classified by CDFW with descriptions of these reproduction stages available at: https://wildlife.ca.gov/Conservation/Delta/Spring-Kodiak-Trawl/Egg-Stages

Mature adults, pre-spawning: Reproductive stages: females 1–3; males 1–4.

Mature adults: spawning: Reproductive stages: females 4; males 5.

6A.1.1.4 Species Threats

Delta Smelt are believed to be limited by various stressors, including water temperature, water quality, prey availability, entrainment at water diversions, increasing frequency and duration of droughts, and contaminants (Sommer et al. 2007a; Miller et al. 2012; Wagner et al. 2011; Interagency Ecological Program Management Analysis and Synthesis Team 2015; Fong et al. 2016; Hamilton and Murphy 2020). Since 2010, several conceptual models (Interagency Ecological Program Management Analysis and Synthesis Team 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 Delta Smelt to identify and describe the reasons behind the population decline. Some of these models have recreated a trend observed in abundance indices, but each model has applied different methodology and predictive covariates. Collectively, these modeling efforts generally support water temperature, water clarity, and prey availability as key factors limiting Delta Smelt populations; water diversions and predation may also have significant impacts. The threats discussed below may be directly or indirectly affected by water operations.

All Delta Smelt life stages are vulnerable to entrainment at the south Delta export facilities. In general, Delta Smelt salvage increases when certain conditions occur, principally when adults move into the south Delta when turbidity exceeds 10 to 12 Nephelometric Turbidity Units (NTU) (Sommer et al. 2011; Bennett and Burau 2015), and with increasing net OMR flow reversal (i.e., more negative net OMR flows). Based on field and salvage data, Kimmerer (2008, 2011) calculated that from near 0 percent to 25 percent of the larval and juvenile population, and from 0 percent to nearly 40 percent of the adult population, can be entrained at the Central Valley Project (CVP) and State Water Project (SWP) facilities annually in years with periods of high exports. Methods to calculate proportional loss estimates have since been debated (Kimmerer 2011; Miller 2011) and work on entrainment estimation has continued (Smith 2019; Smith et al. 2020). Korman et al. (2021) provided a preliminary estimate of adult entrainment during 2002 of 35 percent, based on combined behavior-driven movement models and population dynamics models, which was more than double Kimmerer's (2008) estimate of less than 15 percent. Modeling efforts suggest that entrainment losses have the potential to adversely affect the Delta Smelt population (Kimmerer 2011; Rose et al. 2013a, 2013b; Kimmerer and Rose 2018), although the recent modeling study by Smith et al. (2021) reported that a hypothetical change from the recent OMR flow management strategy (i.e., tidally averaged flows no more negative than -5,000 cubic feet per second [cfs]) to a hypothetical management strategy of 0 cfs flow gave a relatively limited 3.6 percent increase in probability of population growth. Data on population distribution (Murphy and Hamilton 2013) suggest that entrainment is likely to be at the low end of the estimated range in most years. As a result of investigations into entrainment loss, entrainment risk has been limited by restrictions on export pumping since 2008 (U.S. Fish and Wildlife Service 2008:280–282, 2019:40–49).

Delta Smelt adults are most vulnerable to entrainment at the SWP and CVP pumps when they move upstream into the central/southern Delta. Delta Smelt larvae are most vulnerable, if occurring, in the southern and central Delta before and during movement downstream into the west Delta and Suisun Bay/Marsh. While some Delta Smelt live year-round in fresh water far from the CVP and SWP pumps, most rear in the low-salinity regions of the estuary, which lie a safe distance from the SWP and CVP pumps. The timing, direction, and geographic extent of adult spawning movements affect entrainment risk (Sweetnam 1999; Sommer et al. 2011). Prior to the 1990s, high salvage of adult and juvenile Delta Smelt occurred at high, intermediate, or low export levels. Since the 1990s, entrainment risk for fish moving into the central Delta and south Delta is highest when net Delta outflow is at intermediate levels (about 20,000 to 75,000 cfs), and OMR flow is more negative than -5,000 cfs (U.S. Fish and Wildlife Service 2008). When adults move upstream to the Sacramento River and into the Cache Slough Complex, or do not move upstream at all, entrainment risk is appreciably lower. During extreme Wet years, very few Delta Smelt (all life stages) are salvaged because the distribution shifts seaward away from the footprint of the SWP and CVP facilities and because there is relatively less hydrodynamic influence of the south Delta export facilities (i.e., more positive OMR flow; Grimaldo et al. 2009). Hierarchical modeling has recently been developed to characterize the potential for south Delta entrainment losses of vulnerable Delta Smelt life stages (Smith 2019; Smith et al. 2020). Smith (2019) estimated adult entrainment loss in the south Delta from 1994 through 2016 as ranging from 53 fish in 2014 to just over 119,000 fish in 2004. Smith et al. (2020) estimated post-larval entrainment loss in the south Delta during April–June 1995–2015 as ranging from less than 500 fish in 1995 to over 800,000 fish in 2002.

The Interagency Ecological Program Management Analysis and Synthesis Team Delta Smelt conceptual model report found statistically significant relationships of spring Delta outflow (represented by X2, which is the distance in kilometers [km] from the Golden Gate Bridge to the point where the salinity on the bottom is 2 ppt) and prior indices of parental stock (FMWT or SKT indices) as predictors of larval/early juvenile Delta Smelt 20-mm survey abundance indices for the post-Pelagic Organism Decline era (Sommer et al. 2007a; Interagency Ecological Program Management Analysis and Synthesis Team 2015:153–162). (This report stressed that results were preliminary and included for illustrative purposes only and that peer-reviewed publications of the analyses needed to be completed before could be used to draw any conclusions [Interagency Ecological Program Management Analysis and Synthesis Team 2015:152].) In contrast, more recent peer-reviewed results from statistical population dynamics modeling by Polansky et al. (2021) did not find a well-supported link between March–May outflow and Delta Smelt recruitment.

During late summer and fall, Delta outflow affects the location of the low-salinity zone within the upper estuary landscape. Higher Delta outflows (or low X2) expand the low-salinity zone, while lower outflows constrict the extent of the low-salinity zone (Feyrer et al. 2011; Bever et al. 2016). It has been hypothesized that summer and fall environmental conditions are better for Delta Smelt in Wet years, because of a more westerly X2 and an expanded low-salinity zone (Interagency Ecological Program Management Analysis and Synthesis Team 2015). The overlap of the low-salinity zone with Suisun Marsh/Bay results in a considerable increase in an index of habitat based on turbidity (Secchi depth) and salinity (conductivity) as calculated by Feyrer et al. (2011). However, others (Manly et al. 2015) have questioned the use of outflow and X2 location by Feyrer et al. (2011) as an indicator of Delta Smelt habitat because other factors may be influencing survival. Some analyses have shown no relationship of fall X2 (ICF International 2017) or the volume of the low-salinity zone (Polansky et al. 2021) with juvenile Delta Smelt abundance/survival, whereas Polansky et al. (2021) found some evidence for lower fall X2 being positively correlated with Delta Smelt recruitment in the following spring. Polansky et al. (2021) did not find statistical support for volume of the low-salinity zone to be related to juvenile Delta Smelt survival (generally similar to the finding by Kimmerer et al. 2013). Murphy and Weiland (2019) found that the low-salinity zone is not a reliable indicator of Delta Smelt habitat and reported that Delta Smelt can be found in the lower Sacramento River, east of the Delta in largely freshwater conditions, as well as in western regions of the Delta, such as Suisun Bay, where salinity levels typically are higher. Although both conditions bound the range of the species, X2 does not determine the location of other important resources such as food or predators and therefore is not, by itself, a reliable surrogate for Delta Smelt habitat.

Recent work suggested that summer/early fall Delta outflow provides a *P. forbesi* subsidy from the upper Delta to the western portion of the low-salinity zone (Kimmerer et al. 2018; Hamilton et al. 2020), resulting in low prey abundance in the low-salinity zone, where mortality rate is high because of clam grazing; without subsidy from the Delta, abundance of *P. forbesi* would be zero (Kimmerer et al. 2019). Kimmerer et al. (2018) did not find a statistically significant relationship between *P. forbesi* density in the Delta and Delta outflow, whereas Hamilton et al. (2020) found statistically significant decreases in mean total copepod biomass with increasing September/October flow during higher flow conditions at most monitoring locations they examined in the Delta region (discussed below). Detailed examination of a fall flow action in 2017 compared to 2011–2016 did not provide evidence for an increase in Delta Smelt prey with increased outflow resulting in X2 farther downstream in 2017 (Schultz et al. 2019; Flow Alteration Management Analysis and Synthesis Team 2020), whereas Lee et al. (2023) found support for higher abundance of *P. forbesi* in Suisun Bay and Suisun Marsh during years of higher September–November Delta outflow (2017 and 2019) relative to years with lower Delta outflow (2018 and 2020). These empirical observations have been supported by recent modeling analyses, while noting that achieving detectable net gains in *P. forbesi* density in the low-salinity zone may be difficult given the large amount of Delta outflow required (Hassrick et al. 2023). Variability in water temperature and turbidity are primarily driven by climate, but in general, Suisun Bay and Suisun Marsh tend to support more suitable water temperature and turbidity than the Delta (Nobriga et al. 2008), with Suisun Marsh having greater Delta Smelt prey resources than Suisun Bay, which has relatively low food resources as inferred from biological responses such as Delta Smelt growth (Hammock et al. 2015). The Delta Smelt Summer-Fall Habitat Action under the SWP/CVP long-term operations (e.g., California Department of Water Resources 2020) includes reoperation of the Suisun Marsh Salinity Control Gates in order to increase overlap of relatively food-rich areas in Suisun Marsh with low-salinity water for Delta Smelt.

Delta Smelt is a pelagic species, and its physical habitat generally is defined by water quality (Bennett 2005; Feyrer et al. 2007; Nobriga et al. 2008) with some association to bathymetric features (Feyrer et al. 2013) and velocity (Bever et al. 2016). Recent analyses indicate waterbody type and depth are also physical indicators of habitat quality, and seasonal prey density is an indicator of biological habitat quality (Hamilton and Murphy 2020), with higher foraging success (indicated by greater stomach fullness) closer to tidal wetlands (Hammock et al. 2019a). Feyrer et al. (2013) found juvenile Delta Smelt were relatively abundant throughout the water column during flood tides and that during ebb tides they occurred only in the lower half and sides of the water column, suggesting a manipulation of position in the water column to facilitate retention in favorable habitats. Mitchell et al. (2017) sampled subadult Delta Smelt during flood tides (to maximize catch) and found Delta Smelt to be more abundant in surface trawl tows than in oblique trawl tows covering the full water column, suggesting strong surface orientation, possibly due to visual feeding. (The authors noted that their results applied primarily to flood-tide sampling and further research is needed to determine whether similar catch patterns occur during ebb tides.) Bennett and Burau (2015) sampled Delta Smelt during the spawning migration and found that Delta Smelt were caught consistently at the shoal-channel interface during flood tides and near the shoreline during ebb tides in the turbid Sacramento River, with apparent selective tidal movements facilitating either maintenance of position or movement upriver on flood tides and minimizing advection down-estuary on ebb tides. After first flush and initial dispersal, adults appear to hold their position geographically (Polansky et al. 2018).

Multiple field and modeling studies have established the association between elevated turbidity and the presence and abundance of Delta Smelt. Sommer and Mejia (2013) and Nobriga et al. (2008) found that late larval and juvenile Delta Smelt are strongly associated with turbid water, a pattern that continues through fall (Feyrer et al. 2007). Long-term declines in turbidity and higher water temperatures may help explain the modern rarity of juvenile Delta Smelt in the south Delta in summer (Nobriga et al. 2008). Thomson et al. (2010) found decreases in turbidity significantly predicted Delta Smelt decline in abundance. Grimaldo et al. (2009) found that the presence of adults at the fish salvage facilities was linked, in part, with high turbidity associated with first flush events. Turbidity may also cue swimming behavior, at the small-scale (lateral and vertical movements in the water column) and the larger-scale (migration) movements (Bennett and Burau 2015), and a recent laboratory study found that higher turbidity is correlated with reduced physiological stress in juvenile Delta Smelt (Pasparakis et al. 2023). The decline in turbidity appears to be attributable to a decline in sediment supply from upstream, entrapment by invasive submerged aquatic vegetation, and a long-term decrease in wind speed (Hestir et al. 2016; Bever et al. 2018). In addition to occurrence, patterns of Delta Smelt survey catch in relation to turbidity in part may reflect differences in probability of capture, that is, greater ability to avoid capture in clearer water (Latour 2016; Peterson and Barajas 2018), although as previously noted, a recent simulation analysis suggested that the effects of turbidity on catchability may be limited (Tobias 2021).

Upper water temperature limits for juvenile Delta Smelt survival are based on laboratory studies and corroborated by field data. Based on the critical thermal maximum (CTmax), juvenile Delta Smelt acclimated to 17 °C (62.6 °F) could not tolerate water temperatures higher than 25.4 °C (77.7 °F) (Swanson et al. 2000). However, for juveniles acclimated to 11.9 °C, 15.7 °C, and 19.7 °C (53.4 °F, 60.3 °F, and 67.5 °F, respectively), consistently higher CTmax values were estimated—27.1 °C, 28.2 °C, and 28.9 °C (80.8 °F, 82.8 °F, and 84 °F, respectively) (Komoroske et al. 2014), which corresponded closely to the maximum water temperatures recorded in the STN and FMWT. Swanson et al. (2000) used wild-caught fish, while Komoroske et al. (2014) used hatchery-reared fish, which may have contributed to the differences in results. Sublethal temperature effects occur at lower temperature than lethal limits. For example, Smith and Nobriga (2023) estimated that juvenile Delta Smelt prey consumption begins to decline at 21.6 °C and Davis et al. (2019) found that swimming behavior is altered at 21 °C relative to fluctuating (17–21 °C) temperature. Based on the STN (Nobriga et al. 2008) and the 20-mm survey (Sommer and Mejia 2013), most juveniles were predicted to occur in field samples when water temperature was below 25 °C (77 °F). In a multivariate autoregressive modeling analysis with 16 independent variables, Mac Nally et al. (2010) found that high summer (June through September) water temperature had a negative effect on subadult abundance in the fall. Water temperature was also one of several factors affecting Delta Smelt life stage dynamics in the state-space model of Maunder and Deriso (2011) and in an individual-based Delta Smelt life cycle model (Rose et al. 2013a, 2013b).

Harmful algal blooms, in particular *Microcystis*, may have negative effects on Delta Smelt (Brooks et al. 2012). There is no routine quantitative monitoring program in place that specifically targets harmful algae. The STN and FMWT surveys have included qualitative, visual assessment of *Microcystis* since 2007. Available studies in the Delta suggest retention time and water temperature correlates with *Microcystis* bloom amplitude and once established, *Microcystis* is likely to be resistant to even very high flows as long as water quality, especially water temperature, remain favorable (Lehman et al. 2022). Despite increased understanding of the drivers of *Microcystis* blooms, uncertainties remain regarding their direct and indirect effects on Delta Smelt relative to other factors and actions that can be implemented to prevent them.

Delta Smelt prey species have been affected by changes in phytoplankton production and species composition, and the invasion of *Potamocorbula*. For example, there has been a decrease in mean zooplankton size (Winder and Jassby 2011) and calanoid copepod abundances, including a major step-decline in the abundance of the copepod *E. affinis*. This may be due to predation by *Potamocorbula* (Kimmerer et al. 1994) or to indirect effects of clam grazing on copepod food supply. Predation by *Potamocorbula* has also affected other zooplankton species (Kimmerer 2008; Winder and Jassby 2011). There have been timing shifts in key Delta Smelt zooplankton prey peak abundance to earlier in the year, which may have affected Delta Smelt (Merz et al. 2016).

The interaction of *Potamocorbula* grazing with ambient nutrient composition is thought to affect Delta Smelt prey availability. Diatoms (i.e., phytoplankton prey of Delta Smelt's zooplankton prey) preferentially take up ammonium over nitrate but grow more slowly using ammonium (Glibert et al. 2015). *Potamocorbula* consumption of diatoms reduces diatom growth rates and abundance, by limiting metabolization of ammonium to lower levels necessary for rapid diatom growth and greater diatom abundance. Monitoring is ongoing to assess how upgrades to the largest source of dissolved ammonium in the Delta (the Sacramento Regional Wastewater Treatment Plant) that were completed in 2023 affect diatom production in the Delta. A recent analysis concluded high ammonium loading is not a driver of low productivity in the Delta (Strong et al. 2021).

In addition to a long-term decline in calanoid copepods and mysids (Orsi and Mecum 1996) in the upper San Francisco Estuary, there have been numerous introductions of copepod species (Winder and Jassby 2011). *P. forbesi*, a calanoid copepod that was first observed in the estuary in the late 1980s, has replaced *E. affinis* as the most common Delta Smelt prey during the summer. It may have a competitive advantage over *E. affinis* because of its more selective feeding ability. Selective feeding may allow *P. forbesi* to utilize the remaining high-quality algae in the system while avoiding increasingly more prevalent low-quality and potentially toxic food items such as *Microcystis* (Mueller-Solger et al. 2006; Ger et al. 2010). After an initial rapid increase in abundance, *P. forbesi* declined in abundance from the early 1990s in the Suisun Bay and Suisun Marsh regions, but maintained its abundance, with some variability, in the central and south Delta (Winder and Jassby 2011).

The abundance of a more recent invader, the cyclopoid copepod *Limnoithona tetraspina*, significantly increased in the Suisun Bay region beginning in the mid-1990s. It is now the most abundant copepod species in Suisun Bay and confluence region of the estuary (Bouley and Kimmerer 2006; Winder and Jassby 2011). Gould and Kimmerer (2010) found that it grows slowly and has low fecundity. Based on these findings, they concluded that the population success of *L. tetraspina* must be due to low mortality and that this small copepod may be able to avoid the visual predation to which larger copepods are more susceptible. It has been hypothesized that *L. tetraspina* is an inferior food for pelagic fishes, including Delta Smelt, because of its small size, generally sedentary behavior, and ability to detect and avoid predators (Bouley and Kimmerer 2006; Gould and Kimmerer 2010). Nevertheless, this copepod has been found in the guts of Delta Smelt when *Limnoithona* spp. occurs at extremely high densities relative to other zooplankton (Slater and Baxter 2014). Experimental studies addressing this issue suggest that larval Delta Smelt consume and grow on *L. tetraspina*, but growth is slower than with *P. forbesi* (Rose et al. 2013a:1245). It remains unclear if consuming this small prey is energetically beneficial for Delta Smelt at all sizes or if there is a breakpoint above which larger Delta Smelt receive little benefit from such prey. *Acartiella sinensis*, a calanoid copepod species that invaded at the same time as *L. tetraspina*, also reached considerable densities in Suisun Bay and the western Delta after 2000 (Slaughter et al. 2016), although its suitability as food for Delta Smelt remains unclear.

Hamilton et al. (2020) conducted modeling of potential Sacramento and San Joaquin river flow management actions that suggested increasing flows in fall (September and October) of wetter years generally could have negative effects on copepod biomass, whereas increases in flows in the spring (April and May) of drier years could provide regional increases in biomass, particularly in the lower Sacramento and San Joaquin rivers. The latter result is consistent with earlier studies showing X2 to be negatively correlated with *E. affinis* density (Kimmerer 2002). In addition to *Potamocorbula* grazing, recent studies have suggested that south Delta exports also negatively affect phytoplankton and zooplankton productivity (Hammock et al. 2019b; Kimmerer et al. 2019). Tidal wetlands appear to confer substantial benefits to Delta Smelt foraging success, as observed stomach fullness increased with increasing adjacent tidal wetland area (Hammock et al. 2019a).

Delta Smelt declines are negatively associated with metrics assumed to reflect the abundance of predators in the estuary (Maunder and Deriso 2011; Miller et al. 2012; Hamilton and Murphy 2018), including Mississippi Silverside (*Menidia audens*), Largemouth Bass (*Micropterus salmoides*), and other centrarchids. These potential predators are of concern because of their increasing abundance (Bennett and Moyle 1996; Brown and Michniuk 2007; Thomson et al. 2010). Largemouth Bass abundance is inversely correlated to Delta Smelt abundance (Nobriga and Feyrer 2007; Thomson et al. 2010; Maunder and Deriso 2011), possibly due to Largemouth Bass predation on Delta Smelt or the very different responses of the two species to changing habitat within the Delta (Moyle and Bennett 2008). Largemouth Bass will readily eat Delta Smelt when the opportunity arises (Ferrari et al. 2014). However, little evidence supports Largemouth Bass as major consumers of Delta Smelt, due to low spatial co-occurrence (Nobriga et al. 2005; Baxter et al. 2010). Thus, the inverse correlations between these species may not be mechanistic. Rather, they may reflect adaptation to, and selection for, different environmental conditions (e.g., increased submerged aquatic vegetation providing greater habitat suitability for Largemouth Bass and lower habitat suitability for Delta Smelt) (Ferrari et al. 2014).

Moyle et al. (2016) suggested Mississippi Silverside is currently the most important predator of Delta Smelt early life stages, as reflected by recent studies of Delta Smelt DNA in silverside diets (Baerwald et al. 2012; Schreier et al. 2016). Two recent statistical examinations support silverside abundance negatively affecting Delta Smelt survival and abundance (Hamilton and Murphy 2018; Polansky et al. 2021). Silversides may also compete with Delta Smelt for prey and may have an advantage over Delta Smelt because they spawn repeatedly throughout late spring, summer, and fall (Bennett 2005). The closely related smelt species, Wakasagi (*Hypomesus nipponensis*), occurs in the Delta and has prompted concern because of its overlap with Delta Smelt in habitat use, phenology, diet, growth (Davis et al. 2022a) and its broader environmental tolerance than Delta Smelt (Swanson et al. 2000), which could lead it to outcompete Delta Smelt and hybridize with it. However, genetic analyses suggest relatively low levels of hybridization (Fisch et al. 2014).

During the period from 1963 through 1964, Stevens (1966) evaluated seasonal variation in the diets of juvenile Striped Bass throughout the Delta; only age 2 and age 3 Striped Bass contained more than trace amounts of Delta Smelt. The highest reported predation on Delta Smelt was 8 percent of the age-2 Striped Bass summer diet by volume. Thomas (1967) reported on spatial variation in Striped Bass diet composition, based on collections throughout the San Francisco Estuary and the Sacramento River above tidal influence. Delta Smelt accounted for 8 percent of the spring diet composition and about 16 percent of the summer diet composition in the Delta. Brandl et al. (2021) used genetic analysis and found 1.3 percent of Striped Bass had Delta Smelt in their guts, noting that this was higher than in previous reports (0–0.4 percent; Nobriga and Feyrer 2008), which could have been explained by factors such as the sensitivity of the genetic detection method or differences in season or location sampled. Although Delta Smelt are relatively rare in the stomachs of Striped Bass (Nobriga and Feyrer 2007; Nobriga et al. 2013), a recent examination suggested that Striped Bass are important because historical data suggest that declines in Delta Smelt before the current monitoring program began were driven by the invasion of Striped Bass into the estuary (Nobriga and Smith 2020).

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 (Knowles and Cayan 2002; Dettinger 2005), indicate additional challenges to maintaining a sustainable Delta Smelt population (Brown et al. 2013, 2016). A rebound in relative abundance during the very wet and cool conditions in 2011 indicated that Delta Smelt retained some population resilience (Interagency Ecological Program Management Analysis and Synthesis Team 2015). Examination of genetic effective population size during 2011–2014 found that Delta Smelt were not declining because of genetic factors and were not at immediate risk of losing genetic diversity (Finger et al. 2017). Since 2012, declines to record low population abundance indices have been broadly associated with the 2012–2016 drought, and wetter conditions in 2017 and 2019 did not produce a rebound in Delta Smelt numbers as seen in 2011. A more recent evaluation of effective population size has not been published since this further decline.

Central California's warm summers appear to cause energetic stress for Delta Smelt and warm springtime temperatures are assumed to compress the duration of their spawning season (Rose et al. 2013a; Moyle et al. 2016). Central California's climate is anticipated to get warmer (Cayan et al. 2009:6–12). Warmer estuary water temperatures likely present a significant conservation challenge for Delta Smelt (Brown et al. 2013, 2016). Mean annual water temperatures in the Delta are expected to increase steadily during the second half of this century (Cloern et al. 2011). Long periods of higher than normal water temperatures in July and August 2017 had a major negative effect on Delta Smelt in 2017 (Flow Alteration Management Analysis and Synthesis Team 2020:20). The Flow Alteration Management Analysis and Synthesis Team (2020:20) concluded that water temperature is likely a primary factor in the lack of response of the Delta Smelt population to the high flows in 2017.

6A.1.2 Longfin Smelt

6A.1.2.1 Legal Status

In December 2007, CDFW completed a preliminary review of the Longfin Smelt petition (California Department of Fish and Game 2007) and concluded that there was sufficient information to warrant further consideration by the California Fish and Game Commission to list the species. On February 7, 2008, the California Fish and Game Commission designated Longfin Smelt as a candidate for potential listing under CESA. On June 26, 2009, the California Fish and Game Commission determined that it was appropriate to list Longfin Smelt as threatened under CESA. Longfin Smelt is not listed under the ESA, but listing was found to be warranted for the Bay-Delta Distinct Population Segment (DPS) in April, 2012 (77 FR 19756) and a proposed listing rule was published on October 7, 2022 (87 FR 60957); the comment period for the listing was closed on December 6, 2022, and was reopened on February 27, 2023, with a public hearing held on March 14, 2023.

6A.1.2.2 Life History and General Ecology

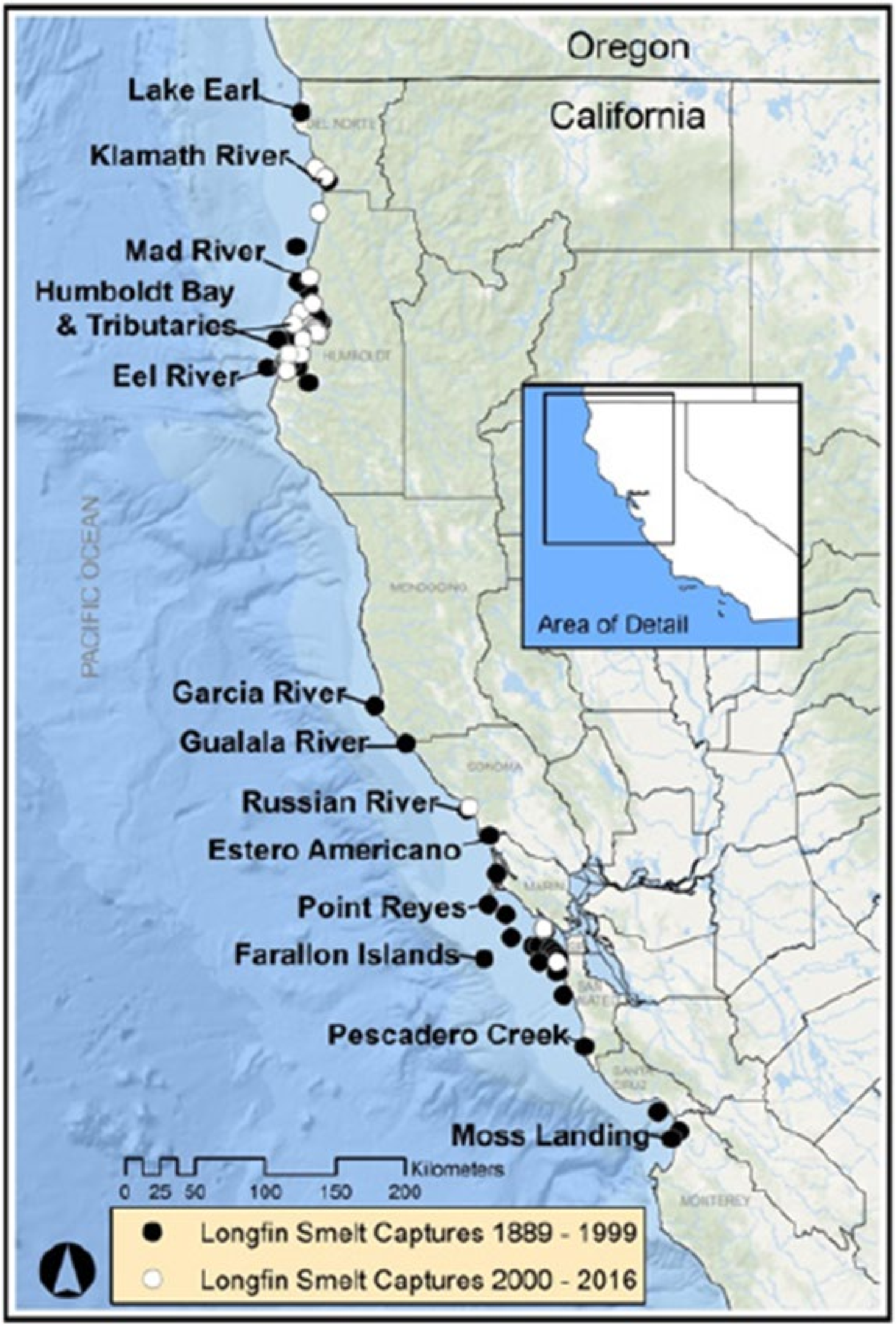
Longfin Smelt is a small, euryhaline, anadromous, and semelparous fish with a life cycle of approximately two to three years (Rosenfield 2010). Longfin Smelt reach 90 to 110 mm SL, with a maximum size of 120 to 150 mm SL (Moyle 2002; Rosenfield and Baxter 2007). Longfin Smelt belongs to the true Smelt family *Osmeridae* and is one of three species in the *Spirinchus* genus; the Night Smelt (*Spirinchus starksi*) also occurs in California and the Shishamo (*Spirinchus lanceolatus*) occurs in northern Japan (McAllister 1963:10, 15). Delta Smelt and Longfin Smelt hybrids have been observed in the Delta, although the offspring are not thought to be viable because Delta Smelt and Longfin Smelt are not closely related taxonomically or genetically (Fisch et al. 2013). Longfin Smelt reside and rear in San Francisco Bay and the nearshore ocean outside the Golden Gate (Garwood 2017). They spawn in tidal fresh water in the estuary's low-salinity zone where brackish and fresh waters meet (Grimaldo et al. 2017) and in fresh water in tributaries to San Francisco Bay (Lewis et al. 2020). Longfin Smelt can be distinguished from other California smelt by their long pectoral fins that reach or nearly reach the bases of the pelvic fins, their incomplete lateral line, weak or absent striations on the opercular bones, low number of scales in the lateral series, and long maxillary bones (which in adults extend just short of the posterior margin of the eye [Moyle 2002]). Populations occur along the Pacific Coast of North America from Hinchinbrook Island in Prince William Sound, Alaska, to the San Francisco Estuary (Lee et al. 1980) and have been detected as far south as Monterey Bay (Garwood 2017).

Longfin Smelt are periodically caught in the nearshore ocean, suggesting that some individuals disperse out into the Gulf of the Farallones to feed and then return to the estuary (Rosenfield and Baxter 2007). They have been documented in Humboldt Bay, the Eel River estuary, the Klamath River estuary, Russian River, and in smaller river estuaries from the central and northern coast of California, including Pescadero Creek, the Garcia River, Gualala River, and Mad River (Figure 6A-5) (Moyle 2002; Pinnix et al. 2004; Garwood 2017; Brennan et al. 2022). It is not known what portion of ocean-bound fish return to San Francisco Bay each year or to other coastal streams north and south of San Francisco Bay (Rosenfield and Baxter 2007; Nobriga and Rosenfield 2016).

Genetic isolation exists between the Longfin Smelt population segment in the San Francisco Estuary and the more northern breeding populations (Stanley et al. 1995; Israel and May 2010a); Due to the low likelihood of southward migration from more northern breeding populations as close as Humboldt Bay, USFWS determined that listing the San Francisco Estuary population as a DPS was warranted (U.S. Fish and Wildlife Service 2012). Longfin Smelt occur throughout the San Francisco Bay and the Delta and in coastal waters west of the Golden Gate Bridge. Within the San Francisco Estuary and Central Valley watershed, they have been observed as far north as the town of Colusa on the Sacramento River, as far east as Lathrop on the San Joaquin River, and as far south as Alviso and Coyote Sloughs in the southern San Francisco Bay as well as various tributaries in northern San Francisco Bay (Merz et al. 2013; Hobbs et al. 2015; Lewis et al. 2020).

As noted by USFWS (2022b:17–18), Longfin Smelt maturation begins in the fall with mature fish observed as late as May of the following year (Tempel et al. 2021). The species is sexually dimorphic; males darken in color and the base of their anal fin hardens and elongates, presumably for sweeping fine sediments from spawning sites (Wang 1986:6–10). Most Longfin Smelt that exhibit onset of maturation are > 90 mm FL (Baxter pers. comm. 2021 to USFWS 2022b:18), while fecundity increases exponentially as a function of female size, and ranges from about 1,900 eggs in a 73-mm female to over 16,000 eggs in a 132-mm female (California Department of Fish and Game 2009a:Figure 3, p. 11). USFWS (2022b:18) also noted that studies of fecundity for the Lake Washington and Harrison Lake populations yielded similar results, with fecundity tending to be a function of both size and feeding success (Dryfoos 1965:120; Chigbu and Sibley 1994:7–8).

In Lake Washington, Longfin Smelt spawn over sandy substrate (California Department of Fish and Game 2009a:11), but spawning substrates are unknown in the San Francisco Estuary. Longfin Smelt eggs are adhesive and demersal (Moyle 2002). Evidence from Grimaldo et al. (2017) suggests spawning habitats include open shallow water and tidal marshes. Longfin Smelt produce between 1,900 and 18,000 eggs, with greater fecundity in fish with greater lengths (California Department of Fish and Game 2009a). Incubation times for egg development range between 25 and 42 days (Rosenfield 2010). Evidence for individuals spawning multiple times in one season has not been investigated but given that Longfin Smelt have such a broad spawning window (five to six months), some females may undergo repeated spawning events. Newly hatched larvae have been observed in salinities up to 12 practical salinity units (psu) with peak observations occurring between 2 and 4 psu (Grimaldo et al. 2017). Early juveniles (20–40 mm SL) are found in salinities up to 30 psu, but most are found in salinities between 2 and 18 psu (MacWilliams et al. 2016). By late summer, juveniles can tolerate full seawater.



Source: Garwood 2017.

Note: Locations with black circles have not necessarily been sampled since 1999, so there is no implication regarding changes in occurrence over time intended by this figure.

Figure 6A-5. Locations of Longfin Smelt Captures, 1889–2016, Excluding the San Francisco Estuary and Delta

Longfin Smelt are anadromous and semelparous, moving from saline to brackish or freshwater for spawning from November to May (Grimaldo et al. 2017; Lewis et al. 2020). They usually live for two years, spawn, and then die, although some individuals may spawn as one-year-old or three-year-old fish before dying (Rosenfield 2010). Age-2 adults appear to move into spawning areas during the late fall and early winter (Rosenfield and Baxter 2007). Spawning occurs at temperatures that range from 5 °C to 15 °C (41 °F to 59 °F) (Grimaldo et al. 2017). Peak spawning takes place in January and February of most years when water temperatures are between 5 °C (41 °F) and 11 °C (51.8 °F), which is consistent with a laboratory study showing greater hatching and physiological performance at 9–12 °C relative to 15 °C (Yanagitsuru et al. 2021). CDFW Smelt Larval Survey (SLS) data show that spawning appears to be centered in brackish water (2 to 4 psu); however, special studies that cover regions seaward of the SLS extent found newly hatched larvae in salinities up to 12 psu and concentrations of larvae peak between 2 and 4 psu (Grimaldo et al. 2017, 2020). Hobbs et al. (2010) provide evidence that larvae with the greatest recruitment success to later life stages are those that hatched and reared in salinities around 2 ppt. A laboratory study found that larval growth and survival were significantly greater at 5–10 ppt than 0.4 ppt, indicating best performance in moderately brackish conditions (Yanagitsuru et al. 2022).

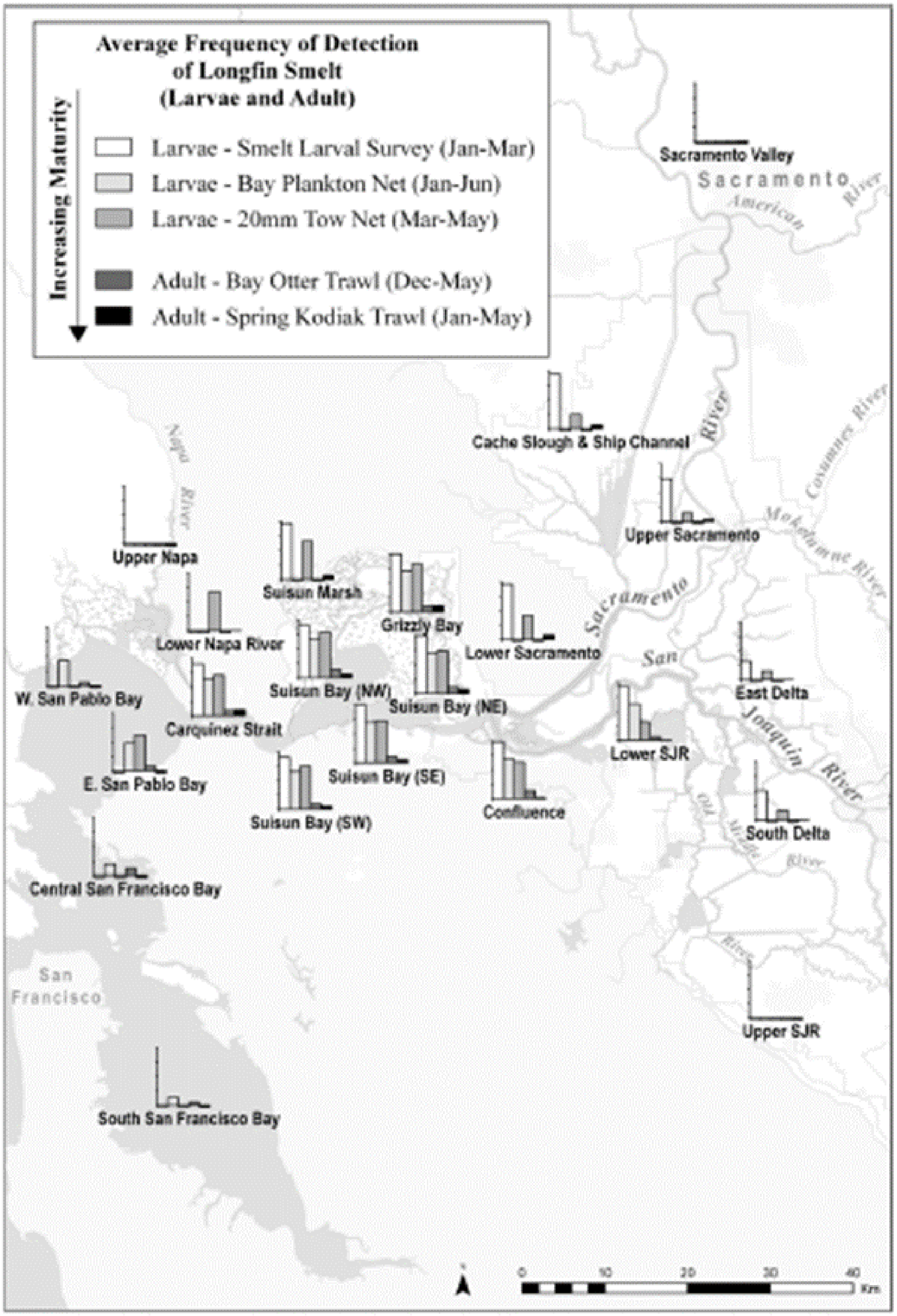
Newly hatched Longfin Smelt larvae appear to be surface-oriented and probably have little ability to control their position in the water column before they develop their air bladder (Bennett et al. 2002). Once their air bladder is developed (approximately 12 mm SL), they can control their position in the water column by undergoing reverse diel vertical migrations or tidal vertical migration, depending on flow conditions (Bennett et al. 2002). Bennett et al. (2002) suggested that the ability of Longfin Smelt to undergo tidal vertical migrations allows them to maintain their position on the axis of the estuary. During the first few months of their lives (approximately January through May), they primarily prey on calanoid copepods, such as *P. forbesi* and *E. affinis*, before switching to mysids as soon as they are large enough to feed on them (Slater 2008; Baxter et al. 2010; Jungbluth et al. 2021; Barros et al. 2022). Mysid density has been positively correlated to spring Delta outflow (negatively correlated to spring X2) (Mac Nally et al. 2010), although Kimmerer (2002) found a changing relationship to X2 for the mysid *Neomysis mercedis* (negative prior to 1987; positive following 1987).

6A.1.2.3 Distribution and Abundance

A general summary of the spatial/temporal distribution of larval and adult Longfin Smelt based on available survey data is provided in Figure 6A-6. The summary in this paragraph and the following paragraph was adapted from USFWS (2022b:21-22). The spatial distribution of Longfin Smelt larvae (< 20 mm length) within the Delta has not been fully resolved due to lack of adequate coverage by monitoring programs (Grimaldo et al. 2017:Figure 5, p. 1777; 2020:Figure 6, p. 10). The majority of larvae are affiliated with the estuary's major low-salinity zone generated by the mixing of freshwater outflow from the Delta with the brackish waters of the estuary (U.S. Fish and Wildlife Service 2022b:Section 2.3). However, larvae can also be found in tributaries when flows from those tributaries are high enough and water temperatures low enough to support egg survival and hatching (Lewis et al. 2019:3). The spatial distribution of larvae reflects the year-to-year variation in the geographic location of the low-salinity zone (Dege and Brown 2004:Figure 3, p. 57; Grimaldo et al. 2020:Figure 6, p. 10). Within the low-salinity zone and adjacent waters, larvae have been commonly collected in both littoral (nearshore) and pelagic (offshore) habitats. Upon hatching, larvae may swim toward the water surface, which would facilitate relatively rapid seaward transport (California Department of Fish and Game 2009a:8). However, it is not clear that such behavior would facilitate retention in the low-salinity zone, especially when Delta outflow is high (Kimmerer et al. 2014:Figure 5, p. 910). Using a 3D hydrodynamic modeling framework, Kimmerer et al. (2014:Figure 5, p. 910 and Figure 6, p. 911) applied the relatively modest swimming capabilities of copepods to show how well simple behaviors could help planktonic animals avoid being washed out to sea and keep them loosely associated within particular salinity ranges. Copepods are considerably smaller than larval fishes, and if they are able to influence their own location in the estuary, it follows that Longfin Smelt larvae may possess this capacity as well (Bennett et al. 2002:1502). The recent findings of larval densities in tidal marsh channels and other edge habitats in densities comparable to offshore waters provides another potential low-salinity zone retention mechanism since tidal currents are slower over shallow shoals and associated marsh channels (Bever et al. 2016:Figure 8b, p. 15).

Aggregated survey data have been used to show that juveniles (>20 mm in length) have been detected at one time or another throughout the estuary and into some tributaries to the Delta above tidal influence (Merz et al. 2013:Figure 2, p. 132). However, the spatial distribution of juveniles shows a distinct seaward migration as water temperatures warm in the late spring and early summer (Rosenfield and Baxter 2007:1590; Tobias and Baxter 2023). Juveniles have been collected most frequently from deep water habitats as opposed to shoals (Rosenfield and Baxter 2007:1586). In Lake Washington, age-0 and age-1 Longfin Smelt favor deep water during daylight and move closer to the surface at night (Quinn et al. 2012:342), likely moving in relation to mysid shrimp, which is their major source of food (Chigbu et al. 1998:180). It is possible that the Bay-Delta DPS may exhibit this movement behavior as well, but this has not been evaluated for post-larval fish. Selection for deep water and a general shift to marine habitat were hypothesized to be behavioral responses to seasonally increasing water temperatures (Tobias and Baxter 2023). Phillis et al. (2021) utilized boosted regression trees and concluded that the strongest predictors of juvenile Longfin Smelt catch in the 20-mm Survey were bottom salinity, Secchi depth, Julian Day, water temperature, surface salinity, and the seven-day average position of X2. The same study predicted larval habitat availability during March through July under low and high spawner abundance in Dry, moderate, and Wet years (Phillis et al. 2021). These authors also predicted that, in Dry years, habitat distributions shifted to Suisun Bay and north San Pablo Bay. In moderate flow years, their analysis predicted that higher freshwater flows resulted in lower salinity into areas of San Pablo Bay, and habitat suitability was predicted to increase in the South San Francisco Bay. In Wet years, they predicted high suitability habitat is available in Suisun Bay, San Pablo Bay, and some of the south San Francisco Bay. Based on otter trawl survey data, juvenile Longfin Smelt rapidly adapt to and inhabit increased salinities because about half the juveniles captured by the larval net came from the salinity range 8 to 24 ppt (Baxter et al. 1999:189–190), well seaward of X2. This increase in salinity distribution represents both seasonal increases in upper estuary salinity as outflow declines and downstream movement of some individuals (Baxter et al. 1999:191). By their first summer of life, juveniles inhabit salinities up to and including marine water (i.e., 32–33 psu; Baxter et al. 1999:191; Rosenfield and Baxter 2007:1590; Kimmerer et al. 2009:385). By May of most years, young-of-the-year (YOY) begin to reach 40 mm FL (Rosenfield and Baxter 2007:1581). At this size, and regardless of outflow, these approximately 40 mm YOY are typically distributed throughout the estuary (Baxter et al. 1999:189; Merz et al. 2013:136–139). They are found from low salinity (and occasionally fresh water) on the upstream end of the Bay-Delta DPS' range, to marine conditions on the downstream end.

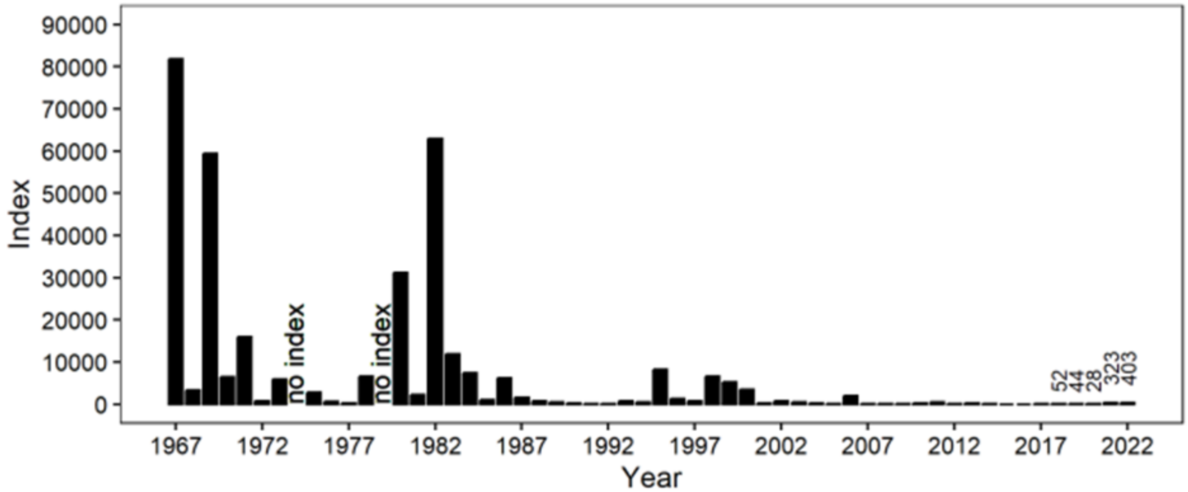
Abundance indices for Longfin Smelt in the Delta have declined over time. For example, there was an approximate 30-fold reduction in the FMWT index since the early 1980s (Figure 6A-7); although these indices do not sample large portions of the area occupied by the species in the Delta (Rosenfield and Baxter 2007; Sommer et al. 2007a; Kimmerer et al. 2009), and an index of 2-year-olds based on the San Francisco Bay Study midwater and otter trawl decreased from a mean of 1,931 from 1980 through 1986 (prior to the *Potamocorbula* invasion) to a mean of 918 from 1987 through 2002, with a further decline following the onset of the Pelagic Organism Decline to a mean of 422 from 2003 through 2013 (Nobriga and Rosenfield 2016). The rate of decline suggested by abundance indices has been particularly steep, especially since the onset of the Pelagic Organism Decline (Sommer et al. 2007a; Thomson et al. 2010), although a recent analysis of an integrated dataset featuring eight different surveys suggests that the original decline dates back to the early to mid-1980s (Stompe et al. 2020). Although the population has declined, prior studies have shown that correlations between winter-spring flow and Longfin Smelt abundance indices have been maintained, suggesting that flow or hydrologic conditions may be strong drivers of population abundance (Kimmerer et al. 2009; Maunder et al. 2015; Nobriga and Rosenfield 2016), although specific mechanisms are unknown. The intercept of such statistical relationships between Delta outflow and abundance indices has decreased over time, possibly because of declining food supply related to *Potamocorbula* (Kimmerer et al. 2009).



Source: Merz et al. 2013.

Note: To calculate the annual frequency of Longfin Smelt detection in a region, the percentage of sampling events where Longfin Smelt were observed is divided by the total number of sampling events for the region. In this graphic, where no column/bar is shown in the bar graph for a region, the average annual frequency of detection for the given Longfin Smelt life stage (s) was zero. Where the column is below the x-axis, a survey did not sample in that region (e.g., the Smelt Larval Survey, which does not include stations west of Carquinez Strait).

Figure 6A-6. Average Annual Frequency of Longfin Smelt Detection (%) for Larval and Adult Life Stages by Region and Interagency Ecological Program Survey Type



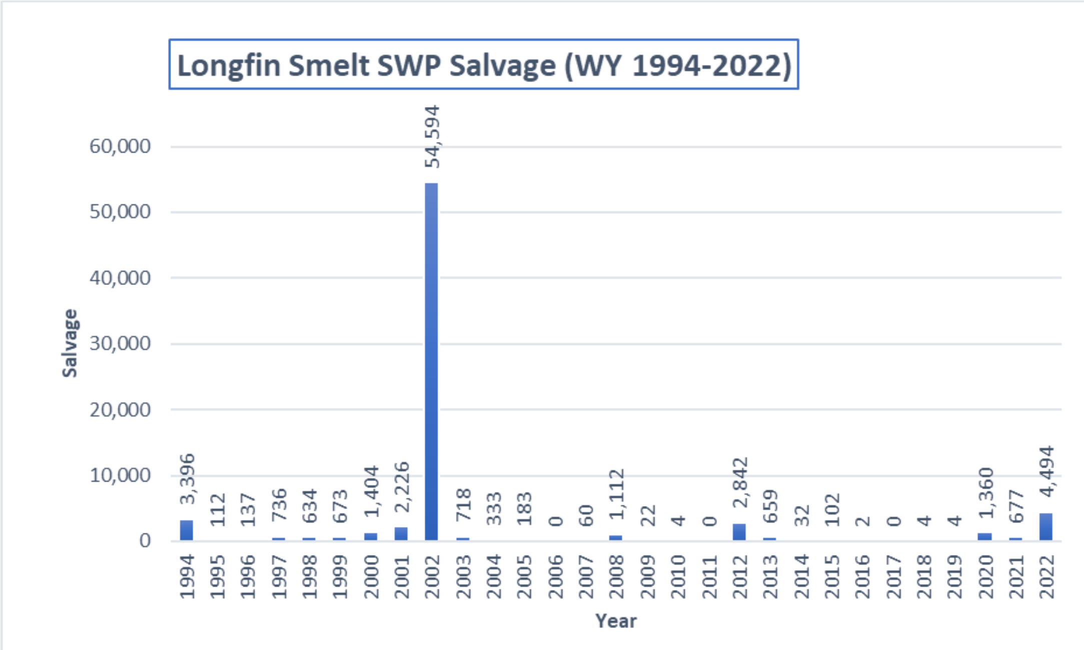
Source: White 2022

Figure 6A-7. Longfin Smelt Fall Midwater Trawl Abundance Index, 1967–2022

6A.1.2.4 Species Threats

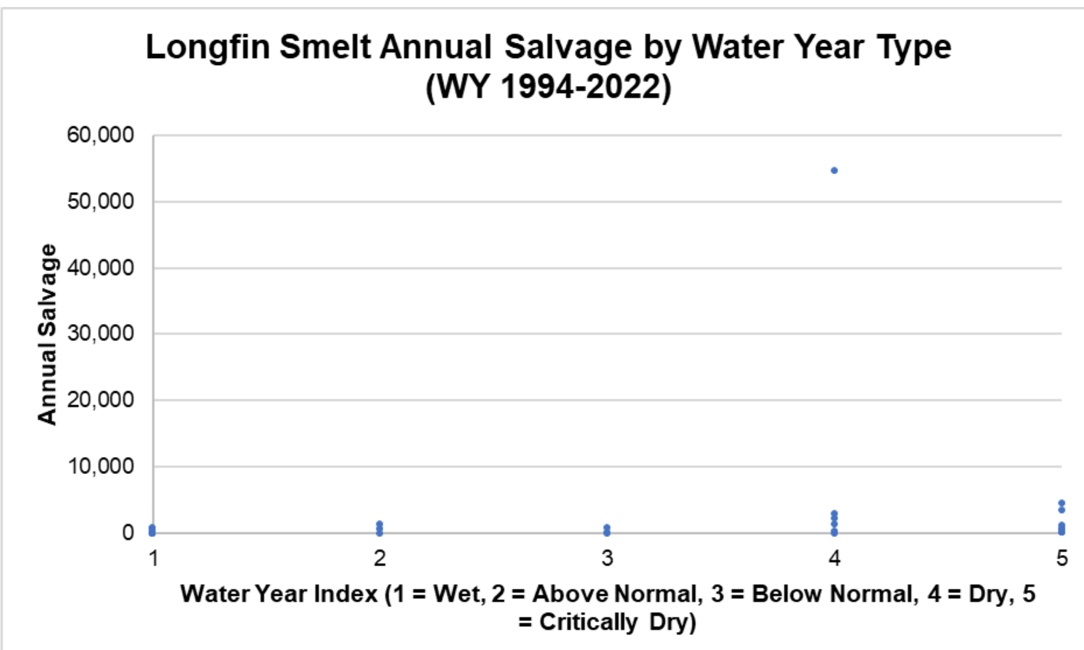
There are several threats to Longfin Smelt. The USFWS (2012) determination that listing is warranted for the Bay-Delta DPS concluded that reductions in freshwater flow and introduced species are threats, and that ammonium may be a threat. The recent federal listing proposed rule examined threats facing the Bay-Delta DPS of the Longfin Smelt as including include habitat alteration and changes to hydrology associated with reduced and altered freshwater flows and resulting increases in saline habitat conditions; increased water temperatures; reduced food resource availability; predation; entrainment from freshwater diversion facilities; and contaminants (87 FR 60957). The discussion below also describes other threats that have been noted (e.g., California Department of Fish and Game 2009a), but not all have been concluded to be of significance to the species (e.g., entrainment; U.S. Fish and Wildlife Service 2012).

Longfin Smelt are vulnerable to entrainment at the south Delta export facilities. The annual number of Longfin Smelt salvaged generally has been low since the 1980s, except in some years (1988, 2002), as illustrated for the SWP salvage facility (Figure 6A-8; California Department of Water Resources 2019:2-15). In general, Longfin Smelt entrainment risk increases with reverse OMR flow (Grimaldo et al. 2009), and salvage can be higher in drier years compared to wetter years (as illustrated for the SWP salvage facility; see Figure 6A-9; California Department of Water Resources 2019:2-15), probably as a result of the landward shift in distribution in drier years. Figure 6A-10 shows the distribution of larval and juvenile Longfin Smelt salinity tolerance in water years of varying runoff. The data presented do not report catch of Longfin Smelt smaller than 40 mm FL and the methodology does not efficiently collect Longfin Smelt larger than 11–12 mm FL. This leaves a potential gap in which fish 12–39 mm FL are not accounted for.



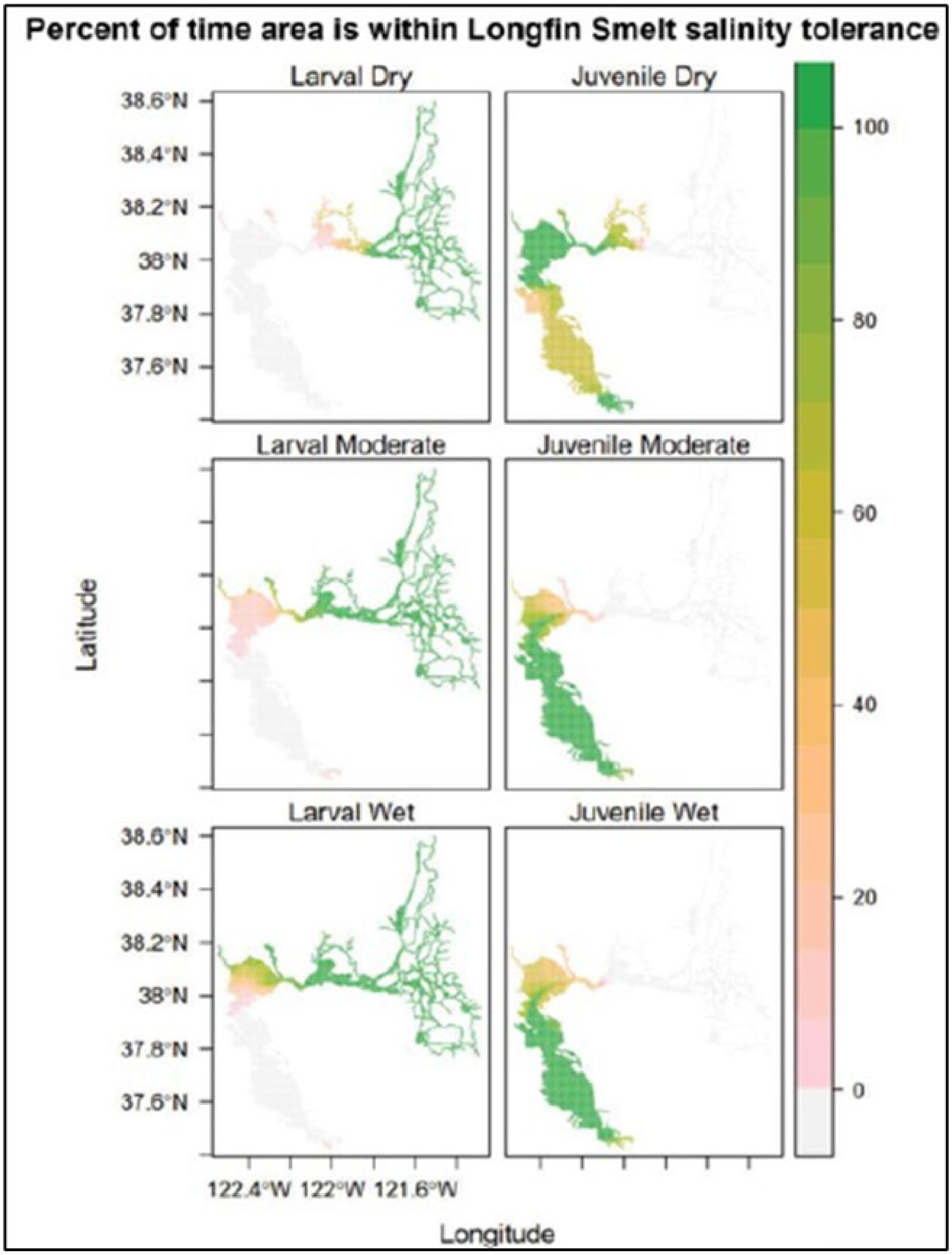
Source: https://apps.wildlife.ca.gov/Salvage/Chart/AcrefeetSalvage?Adipose=All&SampMethod=1&orgCode=25&orgDes=Longfin%20Smelt&endDate=09%2F30%2F2022%2000%3A00%3A00&StartDate=10%2F01%2F1980%2000%3A00%3A00&ShowValue=False

Figure 6A-8. Salvage at the State Water Project John E. Skinner Delta Fish Protective Facility, 1994–2022



Source: https://apps.wildlife.ca.gov/Salvage/Chart/AcrefeetSalvage?Adipose=All&SampMethod=1&orgCode=25&orgDes=Longfin%20Smelt&endDate=09%2F30%2F2022%2000%3A00%3A00&StartDate=10%2F01%2F1980%2000%3A00%3A00&ShowValue=FalseCalifornia Department of Water Resources 2019:2-15

Figure 6A-9. Salvage at the State Water Project John E. Skinner Delta Fish Protective Facility by Water Year Type, 1994–2022



Source: California Department of Water Resources 2019:2-16.

Note: The larval maps span January 1–March 31 and the juvenile maps span April 1–August 31. Salinities are within the tolerable range for Longfin Smelt based on 10th- and 90th-percentile salinities for catches in the Smelt Larval Survey (larval) and the Bay Study (juvenile). The three water years are 2014 (labeled as "dry"; Sacramento Valley runoff = 4.29 million acre-feet [maf] [October–March] and 7.46 maf [total water year]), 2011 (labeled as "moderate"; Sacramento Valley runoff = 12.68 maf [October–March] and 25.21 maf [total water year]), and 2006 (labeled as "wet"; Sacramento Valley runoff = 18.06 maf [October–March] and 32.09 maf [total water year]). The color scale is the percentage of days in the evaluated range that met the salinity tolerance criteria (green = 100%; grey = 0% days in salinity tolerance range). Note that "tolerance" is not taken to mean physiological tolerance, but as described above, the 10th–90th percentile salinity of Longfin Smelt catches.

Figure 6A-10. Distribution of Larval and Juvenile Longfin Smelt Salinity Tolerance in 2014 (Labeled "Dry"), 2011 (Labeled "Moderate"), and 2006 (Labeled "Wet") Water Years

Larval Longfin Smelt are also susceptible to entrainment at the south Delta export facilities; however, because the salvage facilities generally do not sample fish smaller than 20 mm SL, direct larval entrainment is difficult to measure (California Department of Fish and Game 2009a). Larval entrainment at the SWP is likely higher during drier periods compared to wetter periods, but overall larval entrainment risk is likely low because most Longfin Smelt hatch downstream of the upper Delta (Grimaldo et al. 2017). Overall, the effect of entrainment on the Longfin Smelt population has not been found to be important (Maunder et al. 2015), perhaps because a small fraction of the population is estimated to be entrained on an annual basis (California Department of Water Resources 2019:4-48, 4-55; Kimmerer and Gross 2022). Kimmerer and Gross (2022) examined available 2009–2020 survey data for all Longfin Smelt life stages and noted that vulnerability to south Delta entrainment is greatest in early larvae, but that larval losses to entrainment averaging 1.5 percent of the population were too low to measurably influence population dynamics. Consistent with this, Gross et al. (2022) used hydrodynamic and particle-tracking models to estimate that proportional larval entrainment was practically zero in the extreme Wet year of 2017 and approximately 2 percent of the population in the moderately Dry year of 2013. Application of the same methods gave estimates of just under 1 percent larval entrainment in 2021 and 1.3 percent larval entrainment in 2022 (Resource Management Associates 2023).

Longfin Smelt abundance indices have been positively correlated with winter-spring Delta outflow, negatively correlated with winter-spring X2 (Jassby et al. 1995; Kimmerer 2002; Kimmerer et al. 2009; Baxter et al. 2010; Mac Nally et al. 2010; Thomson et al. 2010; Mount et al. 2013; Nobriga and Rosenfield 2016), or positively correlated with general indicators of hydrological conditions (e.g., watershed runoff) (Maunder et al. 2015). Numerous mechanisms have been proposed for this relationship, including lower entrainment losses, advection to suitable habitat, reduced predation due to elevated turbidity, increased retention in favorable habitats, and access to marsh habitats that are unsuitable during drier periods.

The effect of entrainment appears to be unimportant (Maunder et al. 2015) or at least has diminished in recent decades, since Longfin Smelt population-level entrainment losses are low (see discussion above). Vertical retention via estuarine circulation is still hypothesized to be an important mechanism that retains age-0 Longfin Smelt in high-quality habitats during higher flows (Kimmerer et al. 2009). Horizontal retention in large, shallow bays is now hypothesized to be an important feature that enhances Longfin Smelt survival and abundance during higher flows based on new data that targeted larval and juvenile Longfin Smelt in shallow and marsh habitats (Grimaldo et al. 2020).

Kimmerer et al. (2009) concluded that habitat volume, as defined by salinity and water clarity, may be partly responsible for the Longfin Smelt abundance relationship with Delta outflow (X2), but that other mechanisms, such as outflow-driven retention, are more important. With respect to habitat availability, although freshwater flow affects dynamic habitat availability, recent investigations by Grimaldo et al. (2017, 2020) of stationary habitat found that larval Longfin Smelt were relatively abundant in tidal marsh and shallow open waters of the low-salinity zone. This work suggests that stationary shallow habitat also provides key rearing habitat for larval Longfin Smelt, a situation that increased when San Pablo Bay and the south San Francisco Bay became freshwater to low-salinity habitat during Wet years.

Adults use tidal marshes for spawning (Lewis et al. 2020). Larval Longfin Smelt use marsh and shoal habitats as rearing habitat (Grimaldo et al. 2017, 2020) and juveniles are mostly found in deeper channels, often exhibiting diel movements, presumably to reduce predation risk (Bennett et al. 2002).

The salinity distribution in the San Francisco Estuary is not solely dependent on Delta outflow. For example, MacWilliams et al. (2016) showed that salinity in San Francisco Bay was influenced by tributaries as well (e.g., in south San Francisco Bay). Figure 6A-10 shows the availability of habitat for larval and juvenile Longfin Smelt based on salinity tolerance in water years of varying hydrology. Habitat suitability is represented by the percentage of time when a specific location is within the salinity range where 80 percent of larval and juvenile Longfin Smelt were observed in the CDFW SLS and Bay Study surveys, respectively. These surveys do include the full range occupied by the species and therefore limit the scope of inference regarding distribution.

Turbidity levels have declined in the Delta (Cloern et al. 2011). Although Delta Smelt has often been the focus for potential effects of turbidity reduction, some of the same mechanisms appear to be as important for Longfin Smelt (Mahardja et al. 2017). For example, young juvenile Longfin Smelt distribution in spring is negatively associated with water clarity (Kimmerer et al. 2009), and trends in abundance are also negatively associated with water clarity in fall (Thomson et al. 2010). Greater water clarity could somewhat reflect changes in catchability during surveys (fish are better able to avoid trawls when water is clearer) (Latour 2016; Peterson and Barajas 2018; Tobias 2021).

Longfin Smelt have experienced a significant decline in food resources in recent decades (Sommer 2007). A decrease in foraging efficiency and/or the availability of suitable prey for various life stages may result in reduced growth, survival, and reproductive success. This may contribute to an observed lower population abundance and a downward shift in the flow-abundance index relationship, particularly after the introduction of the invasive clam *Potamocorbula amurensis* (Feyrer et al. 2003; Nobriga and Rosenfield 2016). Other factors possibly affecting food resources include ammonium, which was found to be negatively associated with abundance indices in the population dynamics model of Maunder et al. (2015).

Nonnative predators, such as Mississippi Silverside and Striped Bass, have been identified as a potential threat to Longfin Smelt populations (Sommer 2007; Rosenfield 2010), with potentially large predation losses even if the predation rate is low (California Department of Fish and Game 2009a). A composite index of predatory fish density in central San Francisco Bay and San Pablo Bay was found to be negatively associated with trends in Longfin Smelt abundance in population dynamics modeling by Maunder et al. (2015). Competition also occurs with species such as age-0 Striped Bass or American Shad (*Alosa sapidissima*) (Feyrer et al. 2003), although the effect of competition on the Longfin Smelt population is unknown.

Water temperature tends to limit the upstream distribution of Longfin Smelt in the warmer months[1] (Baxter et al. 2010) and spring (April–June) water temperature has been negatively correlated with survival (Maunder et al. 2015). By analogy with Delta Smelt (Brown et al. 2013, 2016), climate change could result in detrimental effects on Longfin Smelt ecology related to factors such as maturation and spawning season length and timing, as well as reduction in habitat extent; potential negative physiological effects of climate change have been demonstrated (Jeffries et al. 2016).

Available information suggests that contaminants may have affected pelagic species in the Delta (Fong et al. 2016), although information specific to Longfin Smelt is very limited. Mauduit et al. (2023) found that experimental exposure of larval Longfin Smelt to environmentally relevant concentrations of the pyrethroid insecticide bifenthrin did not significantly affect cardiac function but bifenthrin altered larval behavior and resulted in smaller hatchlings with reduced yolk sac volumes, suggesting a possible contribution to the observed population decline.

[1] For example, 75 percent of juveniles are found at temperatures of 19.3 °C or less (Davis et al. 2022b).

6A.1.3 Winter-Run Chinook Salmon Sacramento River Evolutionarily Significant Unit

6A.1.3.1 Legal Status

On May 16, 1989, the California Fish and Game Commission listed the Sacramento River winter-run Chinook Salmon (*Oncorhynchus tshawytscha*) Evolutionarily Significant Unit (ESU) as endangered under CESA due to persistent long-term declines. The National Marine Fisheries Service (NMFS), under an emergency interim rule, listed the Sacramento River winter-run Chinook Salmon ESU as threatened under the ESA in August 1989 (54 FR 32085). In 1994, NMFS reclassified the ESU as endangered due to several factors, including the continued decline and increased variability of run size including expected weak returns due to small year classes in 1991 and 1993 and other 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, though efforts to reintroduce the run in Battle Creek have had success in recent years with at least 700 subadults and adults returning in 2020 as a result of juvenile releases undertaken in 2018 and 2019 (U.S. Fish and Wildlife Service 2020). NMFS reaffirmed the ESU's listing 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 artificial propagation program.

6A.1.3.2 Life History and General Ecology

Adults enter fresh water 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 (RBDD) from January through May, peaking in mid-March (National Marine Fisheries Service 2009, 2014a). Adults prefer to hold in deep cold pools until they are sexually mature and ready to spawn in spring or summer. Winter-run Chinook Salmon primarily spawn 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 spawning substrates. Depending on ambient water temperature, embryos hatch within 40 to 60 days and alevin (yolk-sac fry) remain within gravels for an additional four to six weeks. As their yolk-sacs become depleted, fry begin to emerge from gravels 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 RBDD occurs as early as mid-July and extends as late as the end of March during Dry water years (Vogel and Marine 1991; National Marine Fisheries Service 1997), although primary migration ends in December (Poytress and Carrillo 2010, 2011, 2012). A large pulse of juveniles has 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) and may be present in the Delta from November through April. Ocean entry begins as early as November and continues through May (Fisher 1994; Myers et al. 1998, both cited in National Marine Fisheries Service 2014a). Winter-run Chinook Salmon then, for the most part, spend three years in the ocean before returning to natal locations as spawning adults. Further discussion of species life stage timing is provided below in Section 6A.1.3.3, “Distribution and Abundance.”

During juvenile rearing and downstream movement, Chinook Salmon 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 because these areas can be much more productive than the main channel and provide predation refugia (Maslin et al. 1997; Sommer et al. 2001). However, side channels with narrow inverts and nearshore areas with broad flat areas including low-gradient floodplains can also strand and isolate juveniles when high flows subside quickly (National Marine Fisheries Service 1997). The greater availability of prey and favorable rearing conditions in floodplains increase juvenile growth rates compared with conditions in the mainstem, and this can lead to improved survival rates during their migration through the Delta and later in the marine environment (Sommer et al. 2001). However, newer research has not found that the Yolo Bypass, a large floodplain, consistently provides better survival conditions for Chinook Salmon than the mainstem Sacramento River (Sommer et al. 2005; Takata et al. 2017; Johnston et al. 2018; Pope et al. 2021).

Winter-run Chinook Salmon spawn during the summer months when air temperatures usually approach their warmest. As a result, winter-run Chinook Salmon require stream reaches with coldwater sources to protect incubating eggs from the warm ambient conditions. Suitable water temperatures for adult winter-run Chinook Salmon migrating upstream to spawning grounds range from 13.9 °C to 19.4 °C (57 °F to 67 °F) (National Marine Fisheries Service 1997). However, winter-run Chinook Salmon are immature when upstream migration begins and need to hold in suitable habitat for several months prior to spawning. The maximum suitable water temperature reported for holding is 15.0 °C to 15.6 °C (59 °F to 60 °F) (National Marine Fisheries Service 1997).

Adult Chinook Salmon reportedly require water deeper than 0.8 foot and water velocities less than 8 feet per second (ft/sec) for successful upstream migration (Thompson 1972). Chinook Salmon generally hold in pools with deep, cool, well-oxygenated water. Holding pools for adults have been characterized as having moderate water velocities ranging from 0.5 ft/sec to 1.3 ft/sec (National Marine Fisheries Service 2014a:13).

Chinook Salmon spawn in clean, loose gravel in swift, relatively shallow riffles, or along the margins of deeper river reaches where suitable water temperatures, depths, and velocities favor redd construction and oxygenation of incubating eggs. Winter-run Chinook Salmon are adapted for spawning and rearing in the clear, spring-fed rivers of the upper Sacramento River Basin, where summer water temperatures are typically 10.0 °C to 15.0 °C (50 °F to 59 °F). Chinook Salmon require clean loose gravel from 0.75 to 4.0 inches in diameter for successful spawning (National Marine Fisheries Service 1997). Moyle (2002) reported that water velocity preferences (i.e., suitability greater than 0.5 ft/sec) for Chinook Salmon spawning range from 0.98 ft/sec to 2.6 ft/sec (0.3 to 0.8 meter per second) at a depth of a few centimeters to several meters, whereas USFWS (2003) reported that winter-run Chinook Salmon prefer water velocities ranging from 1.54 ft/sec to 4.10 ft/sec (0.47 to 1.25 meters per second).

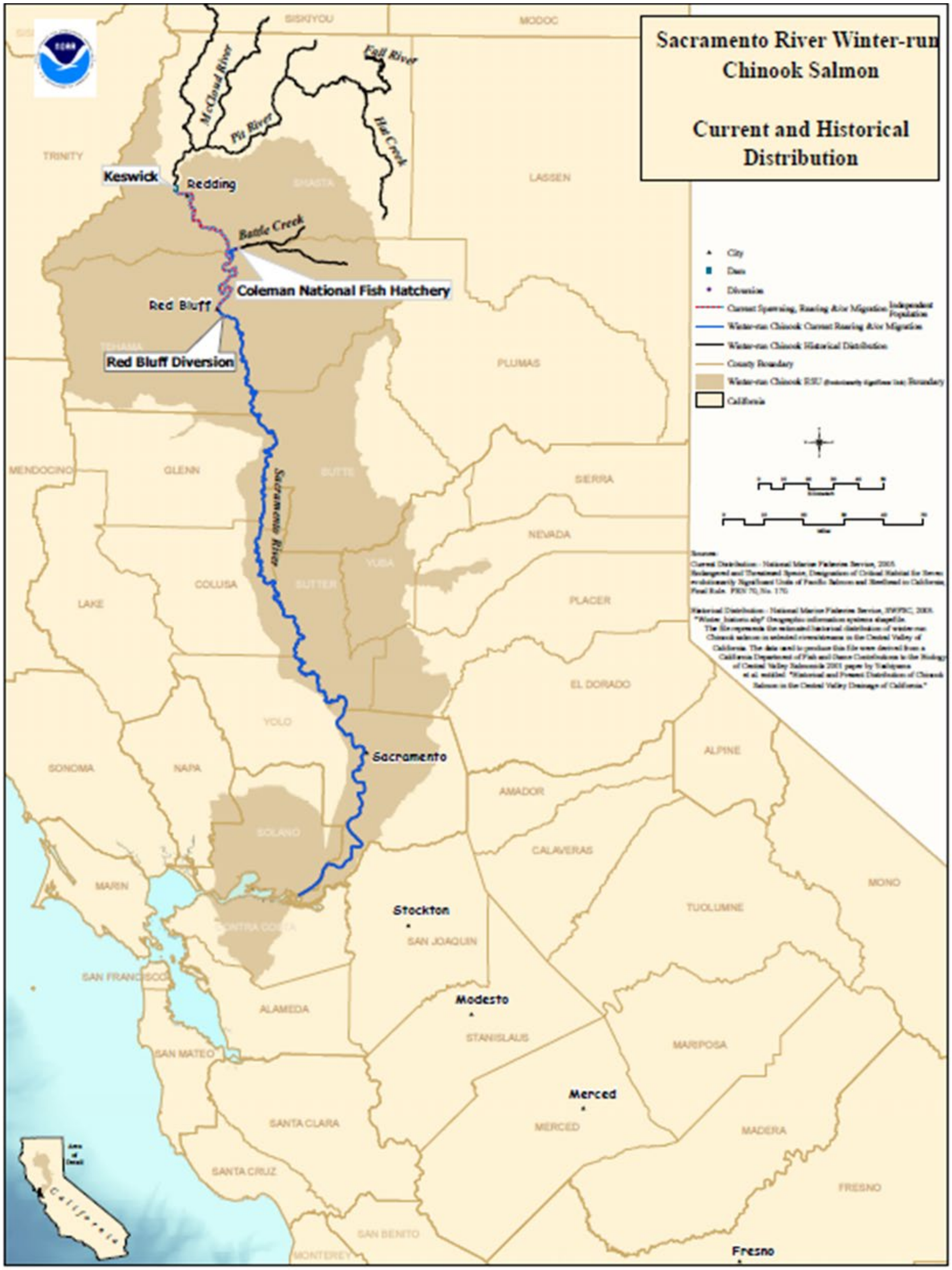
Physical habitat requirements for embryo incubation are the same as the requirements discussed above for spawning. However, it is also important that flow regimes remain relatively constant or at least not decrease significantly during the embryo incubation life stage to maintain sufficient flow of oxygen across egg membranes for successful incubation.

Chinook Salmon fry swim or are displaced downstream after emerging from gravels (Healey 1991). Fry seek streamside habitats containing beneficial aspects such as riparian vegetation and associated substrates that provide aquatic and terrestrial invertebrates for food, predator avoidance cover, and slower water velocities for resting (National Marine Fisheries Service 1996). Juveniles move into deeper water with higher current velocities as they grow larger, but still seek shelter and velocity refugia to minimize energy expenditures (Healey 1991). Within the Delta, juvenile Chinook Salmon are present in water up to 10 ppt salinity (Hendrix et al. 2014) and forage in shallow areas (Williams 2012). Cladocerans, copepods, amphipods, larvae of diptera, small arachnids, and ants are common prey items (Kjelson et al. 1982; MacFarlane and Norton 2002; Sommer et al. 2001).

6A.1.3.3 Distribution and Abundance

Areas where winter-run Chinook Salmon historically spawned are now inaccessible due to Keswick and Shasta dams. Streams in which populations were known to historically exist were fed by cool, constant springs that provided the flows and low water 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 historically would have precluded spawning. This suggests that the area below Shasta and Keswick dams was likely utilized by juveniles as rearing and migration habitat only. The life-history timing, 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, Hat Creek, and Battle Creek (Yoshiyama et al. 1996, 2001; Lindley et al. 2004; National Marine Fisheries Service 2014a). Flow and water temperature requirements were met in these waterbodies (Yoshiyama et al. 2001).

Winter-run Chinook Salmon currently are found in the mainstem Sacramento River downstream of Keswick Dam. This population is maintained through coldwater releases from Shasta Dam that create spawning and rearing habitat in the reach between Redding and the RBDD. The construction of the Anderson-Cottonwood Irrigation District Diversion Dam in 1916 created a partial passage barrier, as did the RBDD in 1962 (until the RBDD gates were permanently locked in the open position in 2013). Since completion of Shasta Dam in 1945, primary spawning and rearing habitats have been confined to the coldwater areas between Keswick Dam and the RBDD (Figure 6A-11).



Source: National Marine Fisheries Service 2014a:12.

Figure 6A-11. Current and Historical Sacramento River Winter-Run Chinook Salmon Distribution

Relative distribution, abundance, and migration timing in the Delta and Sacramento River are presented in Tables 6A-2, 6A-3, 6A-4a, and 6A-4b.

Table 6A-2. Generalized Temporal Occurrence of Winter-Run Chinook Salmon by Life Stage in the Delta

	Month											
Life Stage	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Adult	M	H	H	H	M	M	N	N	N	N	M	M
Juvenile	L	M	H	M	N	N	N	N	N	L	L	M
Salvaged	M	H	H	L	L	L	N	N	N	N	N	L

Relative Abundance: H=High (blue), M=Medium (green), L=Low (yellow), N=None.

Source: National Marine Fisheries Service 2019:68. Note: Table reflects monitoring based on length-at-date classification of juvenile winter-run Chinook Salmon.

Table 6A-3. Generalized Temporal Occurrence of Winter-Run Chinook Salmon Adults in the Sacramento River

	Month											
Location	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Sacramento River basin	M	M	M	M	M	M	M	N	N	N	M	M
Upper Sacramento River spawning	N	N	N	N	L	H	H	M	N	N	N	N

Relative Abundance: H=High (blue), M=Medium (green), L=Low (yellow), N=None.

Source: National Marine Fisheries Service 2019:67.

Table 6A-4a. Frequency of Occurrence (Percent) of Adipose Fin-Unclipped Winter-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	68.1% (586)	57.8% (555)	64.1% (641)	39.9% (581)	4.3% (607)	0.6% (665)	57.7% (742)	95.1% (715)	99.3% (670)	99.4% (710)	99.1% (692)	93.5% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	22.8% (298)	17.4% (270)	8.5% (307)	1.6% (313)	0% (278)	0% (111)	0% (72)	1.6% (62)	7.8% (204)	15.1% (325)	19.3% (337)	34.7% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	27.6% (413)	24.1% (386)	12.3% (423)	2.8% (393)	0% (349)	0% (130)	Not Sampled	11.8% (17)	20.3% (148)	22.4% (344)	24.6% (345)	36.9% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	9.2% (2,784)	6.8% (2,149)	1.8% (2,220)	0.1% (2,060)	0% (2,204)	0% (2,107)	0% (2,043)	0% (2,090)	0.2% (2,086)	1.1% (3,316)	3.8% (3,480)	12.5% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	3.7% (3,402)	6.4% (3,273)	5% (3,524)	2.5% (3,502)	0% (2,908)	0% (2,316)	0% (2,700)	0% (2,637)	0% (2,591)	0.6% (2,664)	2.6% (2,631)	6.1% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	2.5% (4,225)	5.9% (3,257)	23.5% (3,445)	12.6% (4,738)	0.3% (6,348)	0% (3,539)	0% (2,441)	0% (2,264)	0% (2,290)	0% (2,704)	0% (2,612)	1.3% (3,718)
Salvage	1/1/1993-8/10/2023	Days	29.8% (955)	38.4% (874)	56.7% (954)	18.6% (930)	1.1% (960)	0% (930)	0% (960)	0% (940)	0% (900)	0% (929)	0% (900)	16% (930)

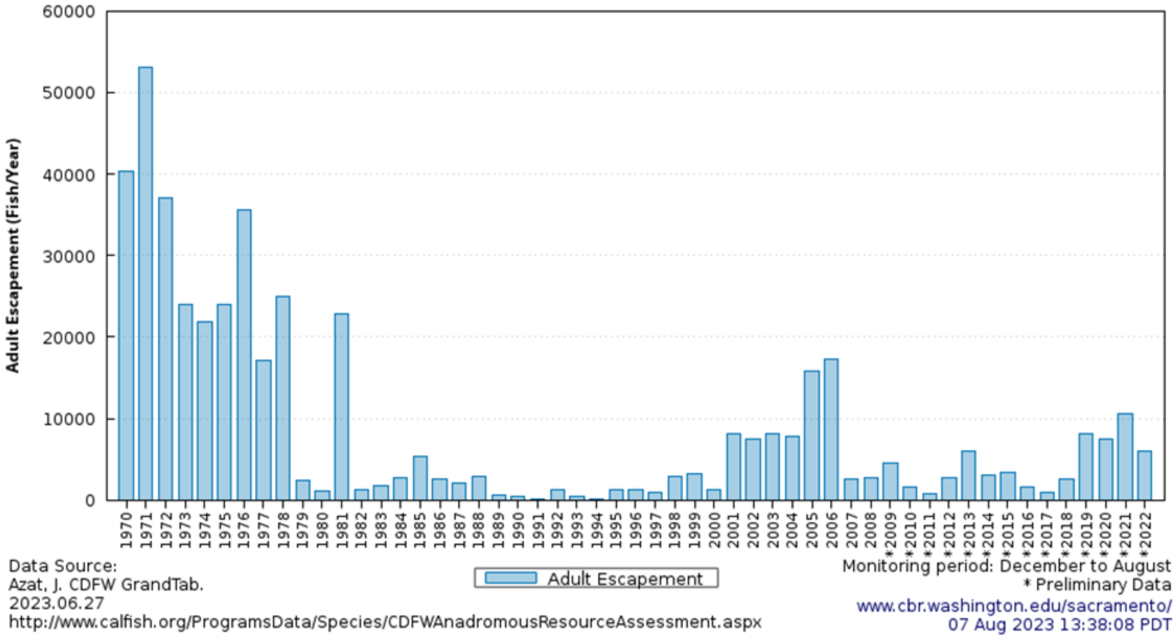
Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one winter-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Table 6A-4b. Frequency of Occurrence (Percent) of Adipose Fin-Clipped Winter-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	66.4% (586)	60.5% (555)	43.1% (641)	12.4% (581)	0.3% (607)	0.2% (665)	0% (742)	0% (715)	0% (670)	0% (710)	1.9% (692)	37.6% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	5.7% (298)	20.4% (270)	8.8% (307)	0% (313)	0% (278)	0% (111)	0% (72)	0% (62)	0.5% (204)	0% (325)	0% (337)	6.6% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	14.5% (413)	21% (386)	12.8% (423)	0.8% (393)	0% (349)	0% (130)	Not Sampled	0% (17)	0% (148)	0% (344)	0% (345)	3.5% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	0.4% (2,784)	1% (2,149)	0.5% (2,220)	0% (2,060)	0% (2,204)	0% (2,107)	0% (2,043)	0% (2,090)	0% (2,086)	0% (3,316)	0% (3,480)	0.4% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	0.6% (3,402)	2.8% (3,273)	2.9% (3,524)	0.7% (3,502)	0% (2,908)	0% (2,316)	0% (2,700)	0% (2,637)	0% (2,591)	0% (2,664)	0% (2,631)	0.5% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	1.5% (4,225)	2.6% (3,257)	8.6% (3,445)	2% (4,738)	0% (6,348)	0% (3,539)	0% (2,441)	0% (2,264)	0% (2,290)	0% (2,704)	0% (2,612)	0.8% (3,718)
Salvage	1/1/1993-8/10/2023	Days	46.8% (955)	42.2% (874)	33.3% (954)	8.7% (930)	1% (960)	0% (930)	0% (960)	0% (940)	0% (900)	0% (929)	0% (900)	15.2% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one winter-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Annual winter-run Chinook Salmon escapement was estimated by counts in traps at the top of fish ladders at the RBDD and more recently estimated using carcass counts. Escapement has declined from the 1960s and 1970s. Run size in 1969 was approximately 120,000 fish, while run size averaged 600 fish from 1990 to 1997 (Moyle 2002). Escapement subsequently increased after RBDD operations were modified, and water temperature control shutters were installed on Shasta Dam, but has declined since 2006 (U.S. Bureau of Reclamation 2008; National Marine Fisheries Service 2016a). Winter-run Chinook Salmon adult escapement data for the Sacramento River Basin from 1970 to 2022 are provided in Figure 6A-12. Escapement in brood year 2022 was ~5,900 fish, lower than in the previous few years.



Source: Columbia Basin Research, University of Washington 2023b. Note: Includes in-river and hatchery fish; data from 2009 to 2022 are preliminary (indicated by asterisks).

Figure 6A-12. Winter-Run Chinook Salmon Adult Annual Escapement in the Central Valley, 1970–2022

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 adults reared in non-natal habitats as juveniles. The lower reaches of the Sacramento River, Delta, and San Francisco Bay serve as migration corridors for the downstream migration of juveniles and the upstream migration of adults.

Adult Sacramento River winter-run Chinook Salmon enter the San Francisco Bay in November to begin their spawning migration and continue upstream from December through July to the extent of anadromy at the base of Keswick Dam (Figure 6A-11). Winter-run Chinook Salmon spawn in the upper mainstem Sacramento River from mid-April through August, peaking in June and July. All known winter-run Chinook Salmon production currently occurs either in the mainstem Sacramento River or the Livingston Stone National Fish Hatchery (California Department of Fish and Game 2004), although a nascent reintroduction effort in Battle Creek led to the return of 942 adults in 2020, 167 adults in 2021, and 127 adults in 2022 (Azat 2023). Current spawning is confined to the mainstem of the Sacramento River above RBDD (River Mile [RM] 243) and below Keswick Dam (RM 302) (National Marine Fisheries Service 2014a). Until recent years, salmon passage was not possible above the Coleman National Fish Hatchery barrier weir located on Battle Creek.

6A.1.3.4 Species Threats

Construction of Keswick and Shasta dams for agricultural, municipal, and industrial water supply eliminated access to approximately 200 river miles of historical holding and spawning habitat above Keswick Dam (Yoshiyama et al. 1996). The Shasta Dam Fish Passage Evaluation is being undertaken to assess the feasibility of reintroducing anadromous fish upstream of Shasta Dam (Plumb et al. 2019). Rearing habitat quantity and quality has been reduced in the upper mainstem Sacramento River as a result of channel modification and levee construction (Lindley et al. 2009). Without access to historical coldwater spawning tributaries above Shasta Dam, persistence of the winter-run Chinook Salmon Sacramento River ESU is dependent on maintaining adequate coldwater pool in Shasta Reservoir to maintain suitable water temperatures downstream of Shasta Dam for winter-run Chinook Salmon egg incubation, fry emergence, and juvenile rearing life stages, especially in Critically Dry years and extended droughts. Warmwater releases during Critically Dry years contributed to low egg-to-fry survival rates. As part of a coordinated drought response, measures taken to preserve Shasta Reservoir's coldwater pool included relaxing Wilkins Slough navigational flow requirements, relaxing D-1641 Delta water quality requirements, and delaying Settlement Contractor depletions into the fall. Egg-to-fry survival rates were low in brood years 2021 (2.6 percent) and 2022 (2.2 percent), likely the result of low thiamine levels despite more proactive water temperature management (Marcinkevage 2023:3, 6). Approximately 215,000 juvenile winter-run Chinook Salmon from brood year 2022 were estimated to have passed RBDD, compared to approximately 570,000 from brood year 2021 and approximately 2.1 million from brood year 2020 (Marcinkevage 2023:3).

Much of the historical floodplain habitat has been developed or converted, which has decreased shallow water habitat with high residence time needed for food production (Jeffres et al. 2008; Katz et al. 2017; Ahearn et al. 2006). Juveniles have access to floodplain habitat in the Yolo Bypass only during mid- to high-water years, and the quantity of floodplain available for rearing during drought years is currently limited. The Yolo Bypass Salmonid Habitat Restoration and Fish Passage Implementation Plan, Long-Term Operation of the CVP and SWP BiOp Reasonable and Prudent Alternative Actions I.6.1 and I.7 include notching the Fremont Weir, which will provide access to floodplain habitat for juvenile salmon over a longer period (California Department of Water Resources and U.S. Bureau of Reclamation 2012). Shoreline armoring and development have reduced access to floodplain rearing habitat for rearing juveniles in the Sacramento River and Delta (Boughton and Pike 2013). Floodplain availability has the potential to increase valuable prey resources and resilience in Chinook Salmon (Goertler et al. 2018a, 2018b). Recent studies suggest Chinook Salmon migration survival through the Yolo Bypass is comparable to that in the Sacramento River (Johnston et al. 2018; Pope et al. 2018); entry into the bypass over Fremont Weir may vary considerably even when river flow into the bypass is substantial, possibly as a function of fish cross-channel position in the Sacramento River (Pope et al. 2018); and travel time in low-flow years is more variable in the bypass than in the river (Johnston et al. 2018).

Juvenile migration corridors are affected by reverse OMR flows that are exacerbated by south Delta export facility operations at the CVP and SWP pumping plants (discussed further in Section 6A.1.4, "Spring-Run Chinook Salmon Central Valley Evolutionarily Significant Unit"). Bidirectional flow in the Sacramento River at Georgiana Slough associated with lower Sacramento River inflow to the

Delta can cause juvenile Chinook Salmon to enter into the interior Delta in greater numbers than with unidirectional flow at high Sacramento River inflow. Entrainment into the interior Delta results in greater travel times and lower survival (Perry et al. 2013, 2018; see additional discussion in Section 6A.1.4). Perry et al. (2013, 2018) and other studies have typically used hatchery-origin juvenile late-fall-run Chinook Salmon large enough to bear acoustic tags, and the general movement patterns are assumed to be representative of other races including wild-origin winter-run juveniles, although this is uncertain and is being investigated further with Juvenile Salmon Acoustic Telemetry System (JSATS) tags that allow smaller fish to be tagged; initial results suggest similar patterns exist for these JSATS-tagged fish (Hance et al. 2022). The movement of juvenile Chinook Salmon into Georgiana Slough reflects the combination of their river cross-sectional distribution and the splitting of water remaining in the Sacramento River and water entering Georgiana Slough, as represented by the critical streakline (Hance et al. 2020). Modeling suggests south Delta exports have little influence on the proportion of Sacramento River flow entering Georgiana Slough (Cavallo et al. 2015).

Stressors thought to be of very high importance by NMFS (2014a:27) to winter-run Chinook Salmon include blockage of historical staging and spawning habitat by Shasta and Keswick dams; flow fluctuations, water pollution, and water temperature impacts in the upper Sacramento River during embryo incubation; loss of juvenile rearing habitat in the form of natural river morphology and function; lost riparian and instream cover; predation during juvenile rearing and outmigration; ocean harvest; and south Delta entrainment. A very recent potential threat identified for winter-run Chinook Salmon is thiamine deficiency complex, possibly the result of the oceanic diet of adults transferring negative effects to juveniles (National Marine Fisheries Service 2020; Mantua et al. 2021).[2] Recent water temperature modeling shows higher sensitivity to increases in water temperature because it leads to exponential increases in oxygen demand with a rise in water temperature during the final weeks of egg-embryo maturation before the alevin stage (Martin et al. 2017, 2020; Anderson et al. 2022). Individual-based modeling of winter-run Chinook Salmon in the upper Sacramento River (Keswick Dam to RBDD) suggested superimposition (i.e., a female salmon making a redd on top of an existing redd) and predation of juveniles are leading causes of mortality of eggs and juveniles, respectively. Modeling suggested that turbidity reduces predation of migrating juveniles (Dudley 2018). Water release scenarios which can cause turbid conditions could be used to assess increased turbidity and determine if that would increase juvenile survival (Dudley 2018). Further individual-based modeling suggested flow is not clearly linked to stranding risk on an annual basis but daily analysis suggested that stranding risk increases as flows decrease and there is a limited positive effect of flow on final outmigrant count; the proportion of eggs being superimposed increases with increasing flow (flow increases velocity, increasing spawner energy expenditure and reducing the time spent guarding the redd, allowing other spawners to make redds on top of the existing redds); and temperature has the largest effect on final juvenile outmigrant count, with decreasing number of outmigrants with increasing temperature (Dudley 2019). The studies of Dudley (2018, 2019) are based on modeling and field-based surveys of factors such as superimposition to validate the modeling results have apparently not been conducted. Martin et al. (2017) found that Chinook Salmon embryo temperature tolerance increases with increasing water velocity (flow). Michel (2019) found a statistically significant positive correlation between

[2] For example, thiamine concentrations in egg samples from 30 winter-run Chinook Salmon females spawned at Livingston Stone National Fish Hatchery in 2021 showed 83 percent of females with thiamine low enough where some fry mortality would be expected (Meyers 2022:6). Any thiamine deficiency impacts manifested in egg viability or early fry stages will lead to reduced juvenile production and number of downstream migrants compared to what would have been observed absent thiamine deficiency impacts (Meyers 2022:6).

Sacramento River flow and hatchery-origin winter-run Chinook Salmon smolt to adult return ratio, which was higher than the correlation with indices of marine productivity. Increased hatchery production of winter-run Chinook Salmon in response to drought conditions in 2014–2015 led to a greater proportion of hatchery-origin in-river spawners, above 80 percent in 2017 and 2018, and remaining at around 40 percent in 2019 and 2020 (U.S. Fish and Wildlife Service 2021:7). This in part contributed to the elevated risk of extinction from low risk at the time of the 2010 five-year species status evaluation to moderate risk at the time of the most recent (2015) evaluation (National Marine Fisheries Service 2016a:34).

Juvenile mortality in the Delta from predation by piscivorous nonnative fishes and conditions that increase risk of mortality of salmonids have been at the forefront of special studies (e.g., Demetras et al. 2016) and reviews (Grossman 2016; Lehman et al. 2019). Special studies are also underway to describe factors such as rearing in Delta bays and marshes and identifying variation in quality of rearing habitat.

Climate experts predict physical changes to ocean, river, and stream environments along the U.S. West Coast that include warmer atmospheric temperatures, diminished snowpack 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 changes in water temperature, hydrology, and ocean conditions are generally expected to have substantial effects on Chinook Salmon populations in the future (National Marine Fisheries Service 2014b; Lindley et al. 2009). Winter-run Chinook Salmon is particularly at risk from global warming because the run relies on the coldwater pool in Shasta Reservoir to maintain spawning conditions in the mainstem Sacramento River. Drought years are predicted to occur with greater frequency in the Sacramento Valley with climate change (Purkey et al. 2008). Increased water temperature associated with lower flows favors nonnative competitors and predators that are adapted to warm water because predation rates increase in response to elevated metabolic rates of predators (Petersen and Kitchell 2001). Increasing the frequency of Dry years also reduces turbidity because sediment loads are not mobilized and transported downstream, although recent studies suggest climate change may increase the frequency of high-flow events and therefore increase sediment transport downstream (Stern et al. 2020). Juvenile salmon are thought to use turbid water to avoid detection by predators (Gregory and Levings 1998). Increased prevalence of submerged aquatic vegetation in the Delta reduces water flow and therefore also reduces turbidity, which has the effect of creating cover for predators and making passing salmon easier for predators to detect (Hestir et al. 2016). Finally, climate change is projected to increase the variability of ocean conditions, such as the North Pacific Gyre Oscillation, the Pacific Decadal Oscillation, and El Niño Southern Oscillations (Di Lorenzo et al. 2010). Anomalies, such as the warm water blob in the North Pacific, disrupt upwelling processes, which drive plankton production in the California Current (Leising et al. 2015). Juvenile salmon distribution is associated with oceanic plankton distribution, and mismatches in space and time that reduce access to marine prey aggregations are thought to influence early marine survival of Central Valley Chinook Salmon populations (Hassrick et al. 2016). Recent studies highlight the importance of forage availability, upwelling, and thermal fronts on juvenile Chinook Salmon feeding in the ocean (Sabal et al. 2020).

6A.1.4 Spring-Run Chinook Salmon Central Valley Evolutionarily Significant Unit

6A.1.4.1 Legal Status

Spring-run Chinook Salmon, which were historically the most abundant run in the Central Valley, were either extirpated or severely diminished from most rivers by mining or dam construction (Williams 2006). Spring-run Chinook Salmon remnant populations now occur in Antelope, Battle, Big Chico, Butte, Clear, Cottonwood, Deer, and Mill creeks, the Feather River, and the Yuba River (National Marine Fisheries Service 2016a). Due to the small number of these remnant populations and the significant hybridization with fall-run Chinook Salmon that has occurred in the mainstem of the Sacramento (Moffett 1949) and Feather rivers (Lindley et al. 2004), spring-run Chinook Salmon were listed as threatened under CESA in 1999. Native spring-run Chinook Salmon have been extirpated from the San Joaquin River watershed, which represented a large portion of their historical range (see below for discussion regarding reintroduced spring-run Chinook Salmon in the San Joaquin River). The spring-run Chinook Salmon Central Valley ESU was listed as threatened under the ESA in 1999 because of the reduced range and small size of remaining 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 Salmon Program (National Marine Fisheries Service 2016a).

6A.1.4.2 Life History and General Ecology

Life history and habitat requirements are largely the same as those described for winter-run Chinook Salmon, with life history differences primarily in the duration and time of year that spring-run Chinook Salmon occupy freshwater habitat. Adult spring-run Chinook Salmon enter fresh water as immature fish between mid-February and July and remain in deep cold pools near spawning areas until they are sexually mature and ready to spawn in late summer and early fall, depending on water temperatures (California Department of Fish and Game 1998; National Marine Fisheries Service 2009).

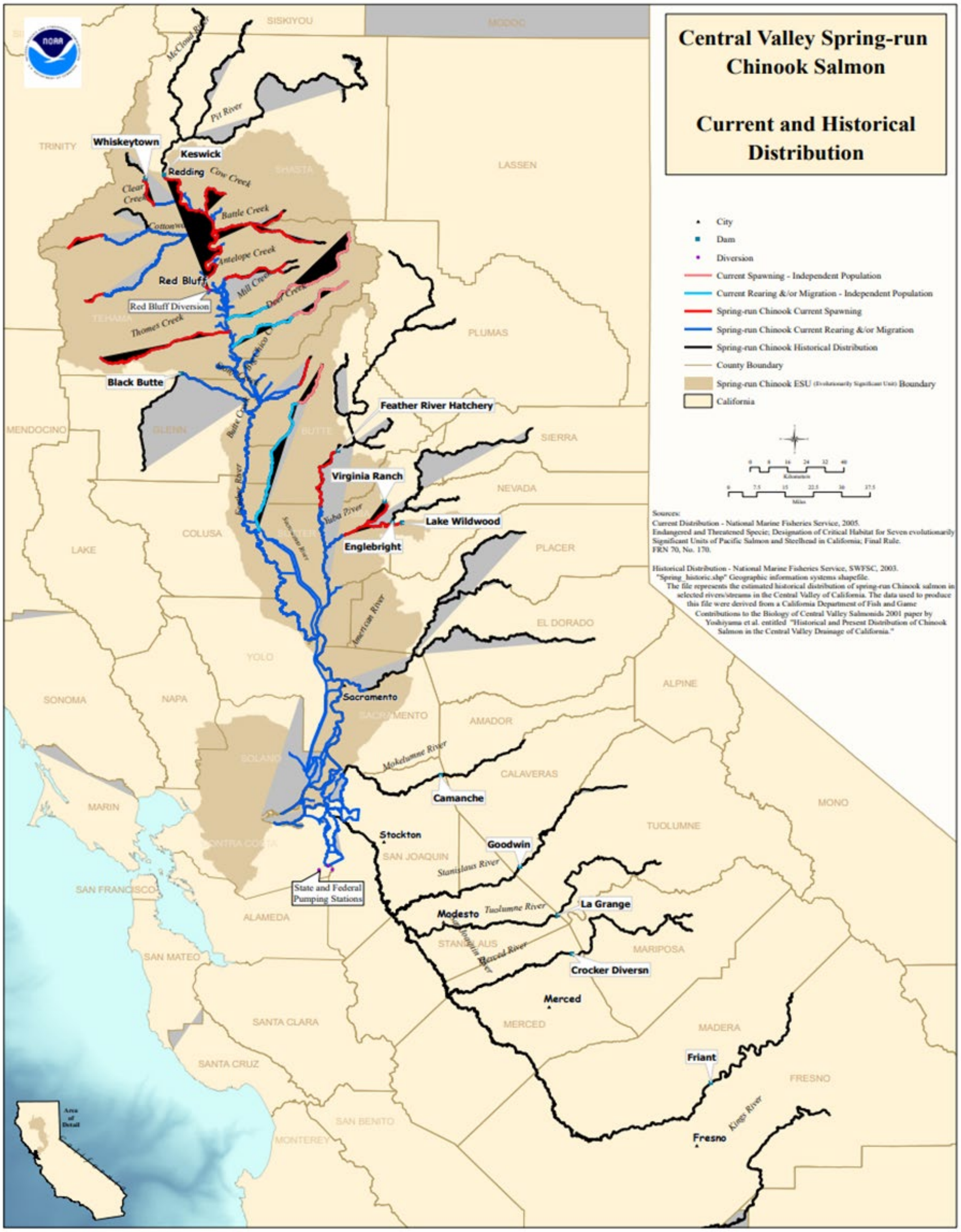
Spawning occurs in gravel substrate in relatively fast-moving, moderately shallow tail-outs of pools or tops of riffles or along banks with relatively high water and velocities that promote higher oxygen levels and eliminate fines in the substrate. Fry emerge from gravels from November to March (Moyle 2002) and can have highly variable emigration timing based on various environmental factors (National Marine Fisheries Service 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 gravels, whereas others over-summer and emigrate as yearlings with the onset of intense fall storms (California Department of Fish and Game 1998). The emigration period for spring-run Chinook Salmon can extend from November to early May, with up to 69 percent of the YOY fish outmigrating through the lower Sacramento River and Delta during this period (California Department of Fish and Game 1998). 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 YOY juveniles (National Marine Fisheries Service 2009). Further discussion of life stage timing is provided in Section 6A.1.4.3, “Distribution and Abundance.”

During rearing and downstream movements, juveniles prefer stream margin habitats with sufficient depths and velocities to provide suitable cover and foraging opportunities. Off-channel areas and floodplains can provide important rearing habitat. The greater availability of prey and favorable rearing conditions in floodplains increase juvenile growth rates compared with conditions in the mainstem Sacramento River. Increased juvenile growth can lead to improved survival rates during migration through the Delta and later in the marine environment (Sommer et al. 2001).

6A.1.4.3 Distribution and Abundance

Spring-run Chinook Salmon historically were the dominant salmon run in the Central Valley; the Central Valley drainage is estimated to have supported annual runs of spring-run Chinook Salmon as large as 600,000 fish between the late 1880s and 1940s (California Department of Fish and Game 1998). Following construction of major dams, annual runs were estimated to be no more than 26,000 fish in the 1950s and 1960s (Azat 2023; Yoshiyama et al. 1998). Dams on the Sacramento River blocked upstream passage of spring-run Chinook Salmon to their historic spawning habitat and confined them to a much smaller area of the watershed (Figure 6A-13). Historically, 18 or 19 independent populations of spring-run Chinook Salmon existed, whereas today only four populations (Battle Creek, Butte Creek, Deer Creek, and Mill Creek) are considered independent of contributions from other populations, with other populations (Antelope Creek, Big Chico Creek, Clear Creek, and Cottonwood Creek) low in number of fish and dependent on other populations (Johnson et al. 2023). Recent surveys have documented very few spring-run Chinook Salmon in the Stanislaus, Tuolumne, and Merced rivers. Nearly 50,000 adults were counted in the San Joaquin River (Fry 1961) before the construction of Friant Dam (completed in 1942). The San Joaquin River watershed populations were essentially extirpated by the 1940s, with only small remnants of the run persisting through the 1950s in the Merced River (Hallock and Van Woert 1959; Yoshiyama et al. 1998). In 2013, NMFS designated the Central Valley spring-run Chinook Salmon reintroduced to the San Joaquin River as an experimental non-essential population in accordance with Section 10(j) of the ESA (78 FR 79622) and Final Rule for Designation of Experimental Populations Under Section 10(j) of the ESA (81 FR 33416). These designations were finalized in 2016 and the implementation process began soon after; by March 2017, the first reintroduction began (California Department of Water Resources 2019). At the end of May 2019, 23 adult spring-run Chinook Salmon returned to the San Joaquin River for the first time in more than 65 years (California Department of Water Resources 2019). Subsequently, adult returns were 57 fish in 2020 (National Marine Fisheries Service 2021), 93 fish in 2021 (National Marine Fisheries Service 2022), and 11 fish in 2022 (National Marine Fisheries Service 2023). Phenotypically spring-running Chinook Salmon have been observed in the Tuolumne and Stanislaus rivers of the San Joaquin River Basin in the last decade and may represent strays from the FRFH (fall- or spring-run) or spring-run Chinook Salmon produced in the Sacramento River Basin (National Marine Fisheries Service 2019:7).



Source: National Marine Fisheries Service 2019:80.

Figure 6A-13. Current and Historical Central Valley Spring-Run Chinook Salmon Distribution

Relative distribution, abundance, and migration timing in the Delta and Sacramento River are presented in Tables 6A-5, 6A-6, 6A-7a, and 6A-7b.

Table 6A-5. Generalized Temporal Occurrence of Spring-Run Chinook Salmon Adults in the Sacramento River

	Month																							
Location	**Jan**		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sep**		**Oct**		**Nov**		**Dec**	
Sacramento River Basin	N	N	N	N	M	M	M	M	H	H	H	H	M	M	M	M	M	M	L	N	N	N	N	N
Sacramento River Mainstem	N	L	L	L	M	M	M	M	M	M	M	M	M	M	L	L	N	N	N	N	N	N	N	N
Adult Holding	N	N	L	L	M	M	H	H	H	H	H	H	H	H	H	M	M	L	L	N	N	N	N	N
Adult Spawning	N	N	N	N	N	N	N	N	N	N	N	N	N	N	L	M	H	H	M	L	N	N	N	N

Source: National Marine Fisheries Service 2019:83.

Relative Abundance: H=High (blue), M=Medium (green), L=Low (yellow), N=None

Table 6A-6. Generalized Temporal Occurrence of Spring-Run Chinook Salmon by Life Stage in the Delta

	Month											
Life Stage	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Adult	M	H	H	H	M	M	N	N	N	N	N	N
Juvenile	L	L	L	H	M	N	N	N	N	N	N	L
Salvaged	L	L	M	H	M	N	N	N	N	N	N	N

Source: National Marine Fisheries Service 2019:84.

Relative Abundance: H=High (blue), M=Medium (green), L=Low (yellow), N=None

* The data in this category reflects juveniles entrained into the salvage facilities. Note: Table reflects monitoring based on length-at-date classification of juvenile spring-run Chinook Salmon. Yearling spring-run Chinook Salmon rear in their natal streams through the first summer following their birth. Downstream emigration generally occurs the following fall and winter. Most young-of-the-year spring-run Chinook Salmon emigrate during the first spring after their hatch.

Table 6A-7a. Frequency of Occurrence (Percent) of Adipose Fin-Unclipped Spring-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	57% (586)	53.2% (555)	78.2% (641)	97.2% (581)	74.1% (607)	10.1% (665)	1.6% (742)	0.3% (715)	0% (670)	41.1% (710)	79.2% (692)	91.2% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	29.9% (298)	30.7% (270)	38.1% (307)	59.4% (313)	10.4% (278)	0% (111)	0% (72)	0% (62)	0.5% (204)	1.2% (325)	7.4% (337)	30.9% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	30% (413)	30.6% (386)	45.4% (423)	66.4% (393)	11.5% (349)	0% (130)	Not Sampled	0% (17)	0% (148)	3.8% (344)	6.7% (345)	26.2% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	19.5% (2,784)	23.6% (2,149)	16.9% (2,220)	9% (2,060)	0.9% (2,204)	0% (2,107)	0% (2,043)	0% (2,090)	0% (2,086)	0% (3,316)	0.9% (3,480)	13.9% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	4.7% (3,402)	9.7% (3,273)	17.2% (3,524)	44.9% (3,502)	9.7% (2,908)	0% (2,316)	0% (2,700)	0% (2,637)	0% (2,591)	0% (2,664)	0.1% (2,631)	4.6% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	0% (4,225)	0.2% (3,257)	13.6% (3,445)	77.5% (4,738)	27.3% (6,348)	0.6% (3,539)	0% (2,441)	0% (2,264)	0.1% (2,290)	0% (2,704)	0% (2,612)	0% (3,718)
Salvage	1/1/1993-8/10/2023	Days	1.4% (955)	5.1% (874)	48.3% (954)	89.8% (930)	69.3% (960)	17.2% (930)	0.3% (960)	0% (940)	0.1% (900)	0% (929)	0% (900)	0% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one spring-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Table 6A-7b. Frequency of Occurrence (Percent) of Adipose Fin-Clipped Spring-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	1.2% (586)	15.5% (555)	26.1% (641)	45.8% (581)	18.8% (607)	0.5% (665)	0% (742)	0.1% (715)	0% (670)	0% (710)	0% (692)	0% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	0% (298)	4.4% (270)	10.4% (307)	29.7% (313)	2.2% (278)	0% (111)	0% (72)	0% (62)	0% (204)	0% (325)	0% (337)	0.6% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	0% (413)	2.8% (386)	5.2% (423)	23.7% (393)	1.7% (349)	0% (130)	Not Sampled	0% (17)	0% (148)	0% (344)	0% (345)	0% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	0% (2,784)	1% (2,149)	2.1% (2,220)	2.5% (2,060)	0.6% (2,204)	0% (2,107)	0% (2,043)	0% (2,090)	0% (2,086)	0% (3,316)	0% (3,480)	0% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	0% (3,402)	1.1% (3,273)	4.9% (3,524)	16.9% (3,502)	7.9% (2,908)	0% (2,316)	0% (2,700)	0% (2,637)	0% (2,591)	0% (2,664)	0% (2,631)	0% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	0% (4,225)	0.1% (3,257)	2.1% (3,445)	21.4% (4,738)	15% (6,348)	1.6% (3,539)	0% (2,441)	0% (2,264)	0% (2,290)	0% (2,704)	0% (2,612)	0% (3,718)
Salvage	1/1/1993-8/10/2023	Days	0% (955)	2.1% (874)	16.7% (954)	35.4% (930)	32.8% (960)	5.5% (930)	0.3% (960)	0% (940)	0% (900)	0% (929)	0% (900)	0% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one spring-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

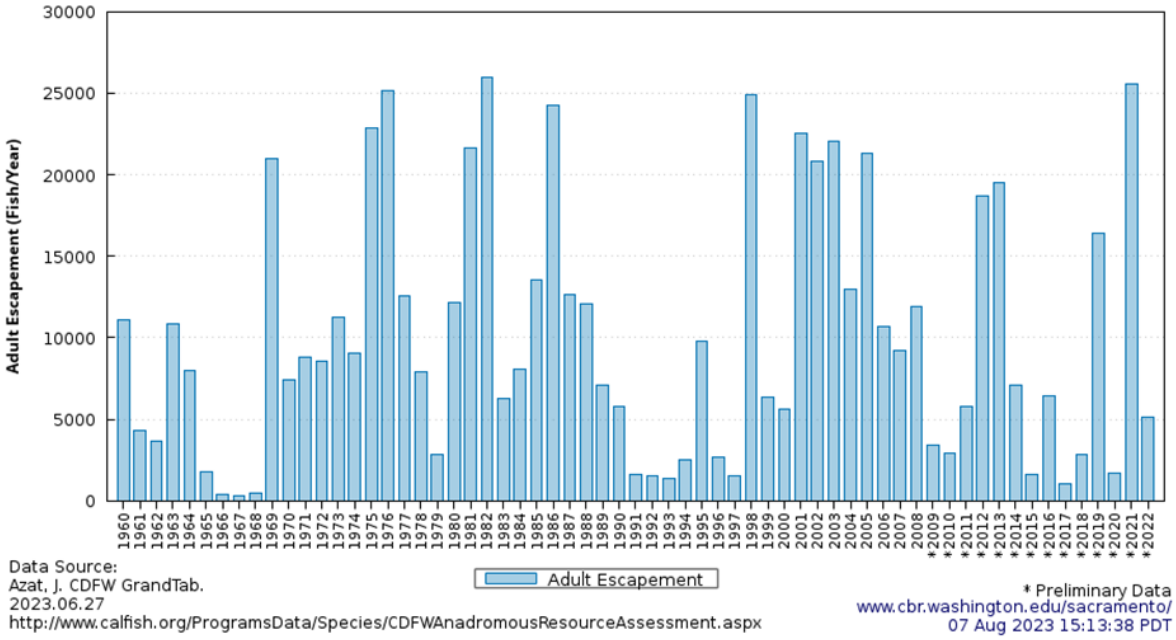
Spring-run Chinook Salmon populations historically occupied the headwaters of all major river systems in the Central Valley up to any natural barrier, such as an impassable waterfall (Yoshiyama et al. 1998; U.S. Bureau of Reclamation 2008). The Sacramento River was used by adults as a migratory corridor to spawning areas in upstream tributaries and headwater streams (California Department of Fish and Game 1998). The most complete historical record of spring-run Chinook Salmon migration timing and spawning is contained in reports to the U.S. Fish Commissioners of Baird Hatchery operations on the McCloud River (California Department of Fish and Game 1998). Spring-run Chinook Salmon 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 Chinook Salmon spawning and the onset of fall-run Chinook Salmon spawning at Baird Hatchery from 1888 through 1901 was 32 days.

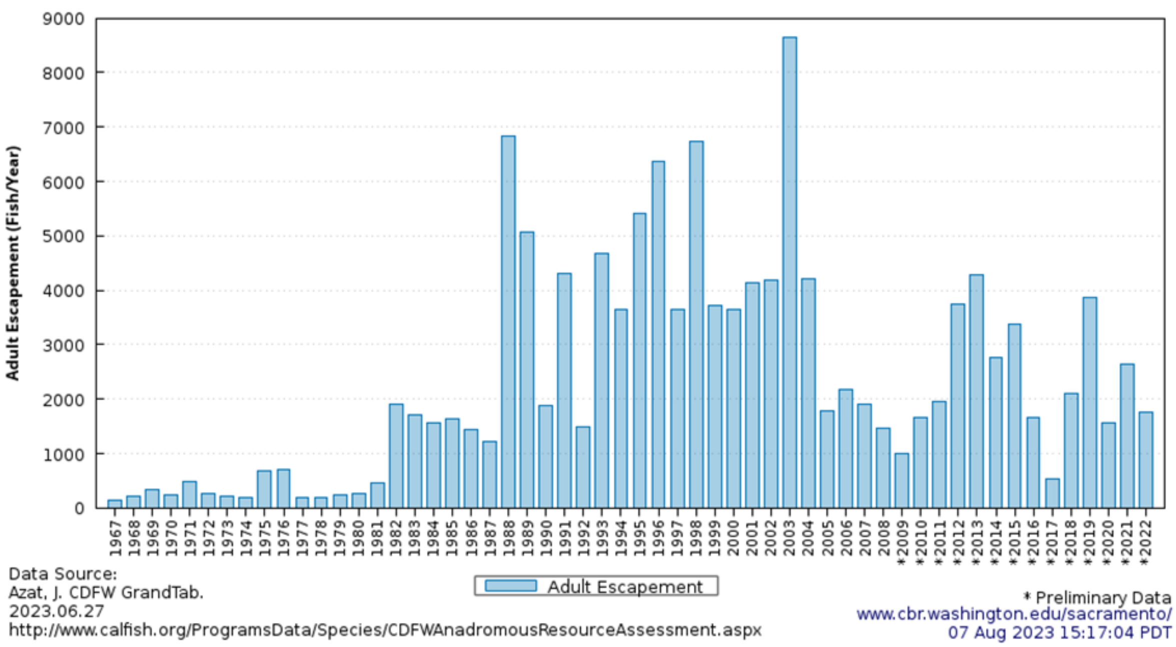
The spring-run Chinook Salmon Central Valley ESU has displayed broad fluctuations in adult abundance. During 1970 through 2022, estimates in the Sacramento River and its tributaries (including the FRFH) have ranged from 1,591 fish in 2017 to 30,697 in 2003.[3]

Independent populations of spring-run Chinook Salmon in the Central Valley include Mill, Battle, Deer, Butte, and Clear creeks, as well as the FRFH (Nelson et al. 2022). Butte Creek spring-run Chinook Salmon make up the largest portion of the independent populations, e.g., based on the three-year sum of annual run sizes for 2017–2019, a total population size of 17,740 fish out of 26,088 fish (68 percent; Nelson et al. 2022). During 2018–2022, spring-run Chinook Salmon escapement estimates (excluding in-river spawners in the Yuba and Feather rivers) in the surveyed tributaries to the Sacramento River ranged from 3,294 fish in 2020 to 28,238 fish in 2021 (Azat 2023).

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 (California Department of Fish and Game 1998). Annual runs were estimated to be no more than 26,000 fish in the 1950s and 1960s (Yoshiyama et al. 1998; Azat 2023) after the construction of most dams. Since 1970, spring-run Chinook Salmon in-river escapement estimates (excluding in-river spawners in the Yuba and Feather rivers, which are considered of hatchery origin) have been highly variable, ranging from 25,890 in 1976 to a low of 1,059 in 2017, with an estimate of 28,238 in 2021 (Figure 6A-14). Escapement to hatcheries increased in the 1980s and has remained higher than in the 1960s–1970s.

[3] The Sacramento River tributaries do not include the lower Yuba and Feather rivers because CDFW's GrandTab does not distinguish between fall-run and spring-run Chinook Salmon in-river spawners.





Source: Columbia Basin Research, University of Washington 2023b

Note: Axis scales differ between upper and lower panels; data from 2009 to 2022 are preliminary (indicated by asterisks).

Figure 6A-14. Spring-Run Chinook Salmon Adult (upper) In-River (excluding Yuba and Feather rivers), 1960–2022, and (lower) Hatchery Annual Escapement in the Central Valley, 1967–2022

6A.1.4.4 Species Threats

Threats and limiting factors for spring-run Chinook Salmon generally include loss of historical spawning habitat, degradation of remaining habitat, genetic threats from the FRFH program, and climate change. Accessible habitat for spring-run Chinook Salmon has been negatively affected by inadequate flows and increased water temperatures due to dam and water diversion operations on streams throughout the Sacramento River Basin (see Section 6A.1.3, “Winter-Run Chinook Salmon Sacramento River Evolutionarily Significant Unit”). Among the main stressors noted for tributaries with spring-run Chinook Salmon are agricultural diversions, diversion dams, and weirs impeding or blocking access on Deer, Mill, Antelope, and Butte creeks; warm water temperatures in Antelope, Butte, and Big Chico creeks during the adult immigration and holding period; and Englebright Dam and Oroville Dam blocking access to upstream habitat in the Feather River and Yuba River (National Marine Fisheries Service 2014a:44–45).

The construction of Whiskeytown Dam, gold mining, and significant gravel mining in the Clear Creek watershed have diminished the availability and recruitment of suitable spawning gravels. The presence of dams on the Sacramento River and its tributaries has blocked upstream passage to historically available spawning habitat and confined spring-run Chinook Salmon 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. The habitat that remains has been negatively affected by inadequate flows, lack of spawning gravels, and increased water temperatures from dam and water diversion operations on streams throughout the Sacramento River Basin, including on Deer, Mill, and Antelope creeks.

Degradation and simplification of aquatic habitat in the Central Valley has reduced the resiliency of spring-run Chinook Salmon to respond to additional stressors such as an extended drought and poor ocean conditions. Loss of life history diversity limits a species’ ability to deal with environmental change, such as timing of ocean productivity, and leads to increased vulnerability through a weakened portfolio effect (Carlson and Satterthwaite 2011). The lost ability to support the yearling life history in most historical spring-run streams may pose a particular threat to persistence because, for brood-years produced during drought, the yearling ecotype disproportionately survives to adulthood relative other ecotypes (Cordoleani et al. 2021). Levee construction and maintenance projects have simplified riverine habitat and have disconnected rivers from the floodplain which decreases juvenile rearing habitat and subsequent survival to adulthood (National Marine Fisheries Service 2016b).

Spring-run Chinook Salmon juvenile migration survival and routing has been statistically correlated with flow, particularly at junctions where fish can route into the interior Delta and become entrained by the export facilities in the south Delta, as shown for acoustically tagged late-fall-run Chinook Salmon juveniles (e.g., Perry et al. 2018). Within the Delta, warming and stable hydrology has favored the expansion of introduced predators, which function as a source of indirect mortality by entrainment toward the export facilities. Increased exports can influence the direction and velocity of flow in the south Delta, with high exports causing stronger reversal in flows nearer the export facilities. When Sacramento River Basin-origin fish route into the interior Delta via locations such as Georgiana Slough or the Delta Cross Channel (DCC) and enter the south Delta, entrainment from reverse flows in OMR may result in longer travel time and indirect mortality (i.e., predation) and direct mortality through loss at the export facilities, as suggested by studies of movement pathways of radio-tagged juveniles (see summary by Vogel 2011:103–105).

As discussed for winter-run Chinook Salmon, juvenile spring-run Chinook Salmon have access to floodplain habitat in the Yolo Bypass only during mid- to high water years and the quantity of floodplain available for rearing during drought years is currently limited, but notching of Fremont Weir due to the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Implementation Plan will provide access to floodplain habitat for juveniles over a longer period (U.S. Bureau of Reclamation 2012).

Operation of the FRFH may pose threats to spring-run Chinook Salmon stock genetic integrity (National Marine Fisheries Service 1998). A large portion of spring-run Chinook Salmon are of hatchery origin, and naturally spawning populations may be interbreeding with fall-run/late-fall-run and spring-run Chinook Salmon hatchery fish. The problem has been heightened by the continued production of spring-run Chinook Salmon from FRFH, especially considering reports suggesting a high degree of introgression between spring- and fall-/late-fall-run broodstock in the hatcheries. Hatchery broodstock management has attempted to segregate the two runs. Despite these efforts, substantial hybridization has occurred, resulting in substantial genetic introgression (Clemento et al. 2014; Meek et al. 2016). The California Department of Water Resources (DWR) and CDFW are implementing further actions to reduce this issue, including fish ladder operations to separate runs, use of genetics and coded-wire-tag retrieval to cull fall-run eggs during spring-run production; and a draft Hatchery and Genetics Management Plan.

Studies in the lower Feather River have shown that there is a relatively high rate of infection of juvenile Chinook Salmon with the myxozoan parasite *Ceratonova shasta*, which could affect spring-run Chinook Salmon. Adult Chinook Salmon carcasses from the Feather River low-flow channel (LFC) produce billions of myxospores annually that move downriver over the winter, which are then consumed by the alternative polychaete worm host *Manayunkia occidentalis*, from which waterborne actinospores capable of infecting juvenile Chinook Salmon are released (Foott et al. 2023). A five-year study demonstrated that the prevalence of infection is considerably greater in the Feather River high-flow channel (HFC) than the LFC and that there is an infectious zone beginning at the outlet of the Thermalito Afterbay Outlet, with initial infection of fry and detection of the actinospore stage in river water beginning in late January or early February, with overt, lethal disease occurring in March (Foott et al. 2023). The HFC itself appears to be the primary source of actinospore production, as opposed to the Afterbay (Foott et al. 2023). Foott et al. (2023) observed that ≥ 85 percent of the natural fry population has migrated past their HFC trapping location when disease severity increases in March, with disease progression occurring if exposure within the HFC is five days or more. Additional surveys are required to elucidate whether disease progresses in fry chronically exposed to low actinospore concentrations during downstream river migration in January and February (Foott et al. 2023). Despite high flows in 2017 and relatively low spore concentrations in 2018, most fish sampled in 2018 were assessed to be diseased. Foott et al. (2023) noted that after their first study year (2015) the infectious zone has since enlarged to include reaches of the Feather River below the Yuba River confluence. Foott et al. (2023) noted that the high flows of 2017, followed by increasing levels of infection in 2018 and 2019, suggest that some portion of the alternative host polychaete population is associated with stable habitat such as the lee side of boulders and riprap and can recolonize within a year.

Climate change poses a further threat to spring-run Chinook Salmon due to increasingly high water temperatures and changes to ocean conditions. Spring-run Chinook Salmon may be particularly vulnerable to these changes because adults over-summer in freshwater streams before spawning in autumn. Spring-run Chinook Salmon spawn primarily in the tributaries to the Sacramento River, and those tributaries without coldwater 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 (National Marine Fisheries Service 2016a). Juveniles may rear in their natal stream for one to two summers prior to emigrating and would be susceptible to warming water temperatures.

6A.1.5 Fall-Run/Late-Fall-Run Chinook Salmon Central Valley Evolutionarily Significant Unit

Central Valley fall-run and late-fall-run Chinook Salmon pass through the Delta as adults migrating upstream and juveniles outmigrating downstream. Adult fall-run and late-fall-run Chinook Salmon migrating through the Delta must navigate the many channels and avoid direct sources of mortality and minimize exposure to sources of nonlethal stress. Additionally, outmigrating juveniles are subject to predation and entrainment in the project export facilities and smaller diversions.

Adult fall-run Chinook Salmon migrate through the Delta and into Central Valley rivers from June through December. Adult late-fall-run Chinook Salmon migrate through the Delta and into the Sacramento River from October through April. Adult Central Valley fall-run and late-fall-run Chinook Salmon migrating into the Sacramento River and its tributaries primarily use the western and northern portions of the Delta, whereas adults entering the San Joaquin River system to spawn use the western, central, and southern Delta as a migration pathway.

Most fall-run Chinook Salmon fry rear in fresh water from December through June, with outmigration as smolts occurring primarily from January through June. In general, fall-run Chinook Salmon fry abundance in the Delta increases following high winter flows. Smolts that arrive in the estuary after rearing upstream migrate quickly through the Delta and Suisun and San Pablo Bays. Late-fall-run fry rear in fresh water from April through the following April and outmigrate as smolts from October through February (Snider and Titus 2000). Juvenile Chinook Salmon were found to spend about 40 days migrating through the Delta to the mouth of San Francisco Bay (MacFarlane and Norton 2002).

Results of mark-recapture studies conducted using juvenile Chinook Salmon released into both the Sacramento and San Joaquin rivers have shown high mortality during passage downstream through the rivers and Delta (Brandes and McLain 2001:62; Newman and Rice 2002; Buchanan et al. 2013). Juvenile salmon migrating from the San Joaquin River generally experience greater mortality than fish outmigrating from the Sacramento River. In years when spring flows are reduced and water temperatures are increased, mortality is typically higher in both rivers. As noted previously in the account for winter-run Chinook Salmon, flow-survival relationships have been demonstrated for juvenile Chinook Salmon in the Delta (Perry et al. 2018). Closing the DCC gates and installation of the Head of Old River Barrier to reduce the movement of juvenile salmon into the south Delta from the Sacramento and San Joaquin rivers, respectively, may contribute to improved survival of outmigrating juvenile Chinook Salmon from these watersheds.

Although not directly comparable to these previous coded wire tag studies in the San Joaquin River, Buchanan et al. (2013) found that survival of acoustically tagged hatchery-origin (Feather River) juvenile Chinook Salmon was either not statistically different between routes (2009) or was higher through the south Delta via the Old River route than via the San Joaquin River (2010). Additionally, most fish in the Old River that survived to the end of the Delta had been salvaged from the federal water export facility on the Old River and trucked around the remainder of the Delta (Buchanan et al. 2013; San Joaquin River Group Authority 2013). Buchanan et al. (2013) indicated that the differences in their results compared to past coded wire tag studies may reflect that an alternative nonphysical barrier was being used during their investigation to examine its ability to keep fish out of the Old River instead of the Head of Old River barrier, which is a physical barrier that reduces not only the number of fish, but also the majority of flows, from entering the Old River. Nonphysical barriers may deprive smolts routed to the San Joaquin River of the increased flows needed for improved survival and may have created habitat for increased predation at the site (Buchanan et al. 2013). However, as noted in the spring-run Chinook Salmon account, Buchanan and Skalski (2019) did not find a statistically significant relationship between juvenile fall-run Chinook Salmon survival from the Head of Old River to Chipps Island and Head of Old River barrier presence; Buchanan et al. (2021) did, however, find evidence for a positive Head of Old River barrier effect on juvenile steelhead survival. Further discussion of through-Delta survival issues is provided below.

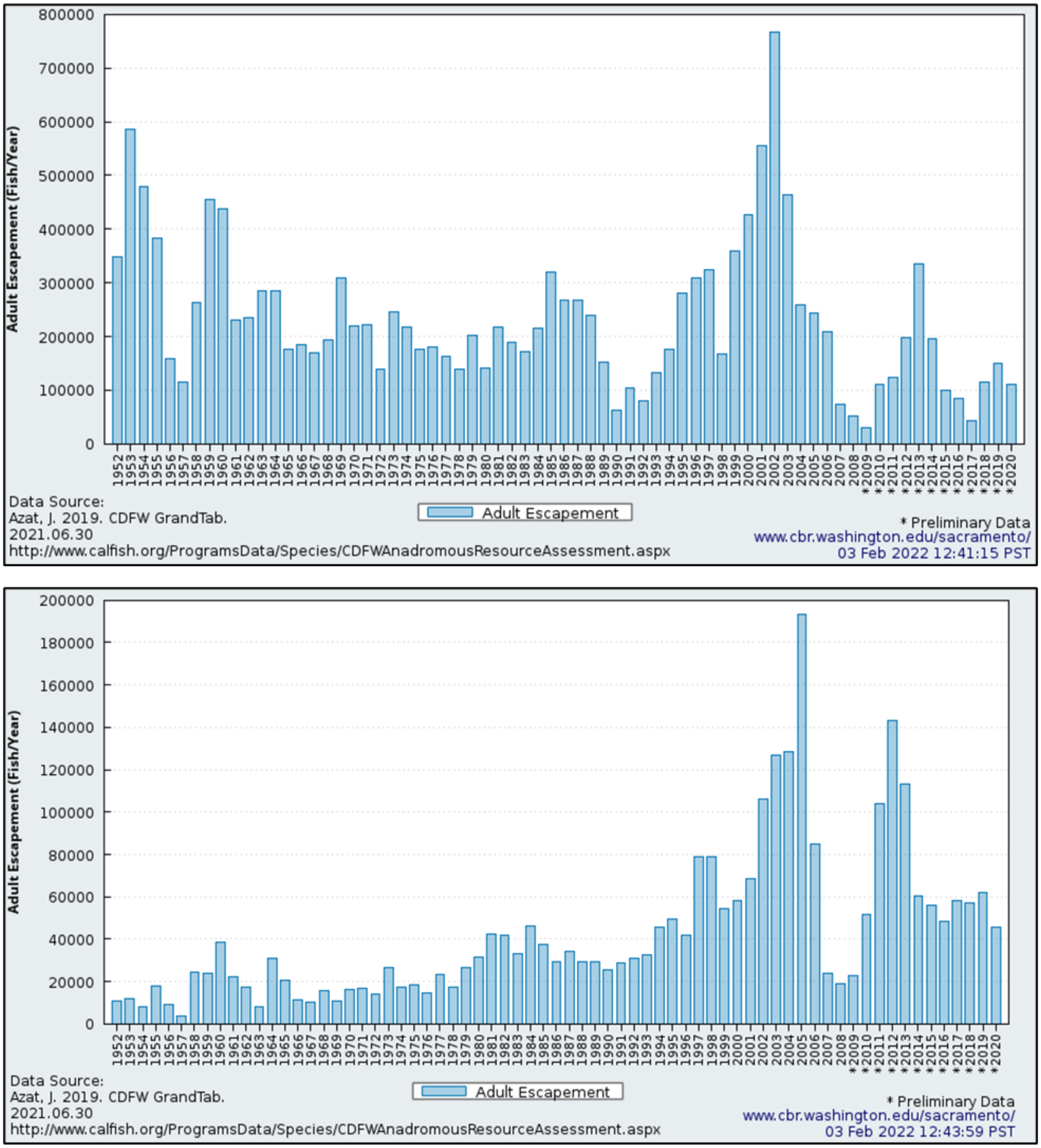
Juvenile fall-run and late-fall-run Chinook Salmon migrating through the Delta toward the Pacific Ocean use the Delta, Suisun Marsh, and the Yolo Bypass for rearing to varying degrees, depending on their life stage (fry versus juvenile), size, river flows, and time of year. Movement of juvenile Chinook Salmon in the estuarine environment is driven by the interaction between tidally influenced saltwater intrusion through San Francisco Bay and freshwater outflow from the Sacramento and San Joaquin rivers (Healey 1991).

In the Delta, tidal and floodplain habitat areas provide important rearing habitat for foraging juvenile salmonids, including fall-run Chinook Salmon. Studies have shown that juvenile salmon may spend two to three months rearing in these habitat areas, and habitat losses resulting from land reclamation and levee construction are considered to be major stressors (Williams 2010).

The fall-run Chinook Salmon has an ocean-maturing type of life history adapted for spawning in lowland reaches of big rivers, including the mainstem Sacramento River. The late-fall-run Chinook Salmon has a stream-maturing type of life history (Moyle 2002). Similar to spring-run, adult late-fall-run Chinook Salmon typically hold in the river for one to three months before spawning, while fall-run Chinook Salmon generally spawn shortly after entering fresh water. Fall-run Chinook Salmon migrate upstream past RBDD on the Sacramento River between July and December, typically spawning in upstream reaches from October through March. Late-fall-run Chinook Salmon migrate upstream past RBDD from August to March and spawn from January to April (National Marine Fisheries Service 2009; Tehama-Colusa Canal Authority 2008). The majority of young fall-run Chinook Salmon migrate to the ocean during the first few months following emergence, although some may remain in fresh water and migrate as yearlings. Late-fall-run Chinook Salmon juveniles typically enter the ocean after 7 to 13 months of rearing in fresh water, at 150- to 170-mm FL, considerably larger and older than fall-run Chinook Salmon (Moyle 2002). The primary spawning area used by fall-run and late-fall-run Chinook Salmon in the Sacramento River is the area from Keswick Dam downstream to RBDD. Spawning densities for all of the Chinook Salmon runs are highest in this reach, but fall-run Chinook Salmon generally spawn farther downstream in the reach than the other Chinook Salmon runs (Gard 2003:2).

Factors affecting fall-run and late-fall-run Chinook Salmon are generally similar to those discussed above for winter-run and spring-run Chinook Salmon. Recent life cycle modeling for fall-run suggested that among the processes examined, the most influential factors were temperature experienced during egg incubation, freshwater flow during juvenile outmigration, and environmentally mediated predation during early marine residence (Friedman et al. 2019). Michel (2019) found a statistically significant positive correlation between Sacramento River flow and hatchery-origin fall-run and late-fall-run Chinook Salmon smolt to adult return ratio, which was higher than the correlation with indices of marine productivity.

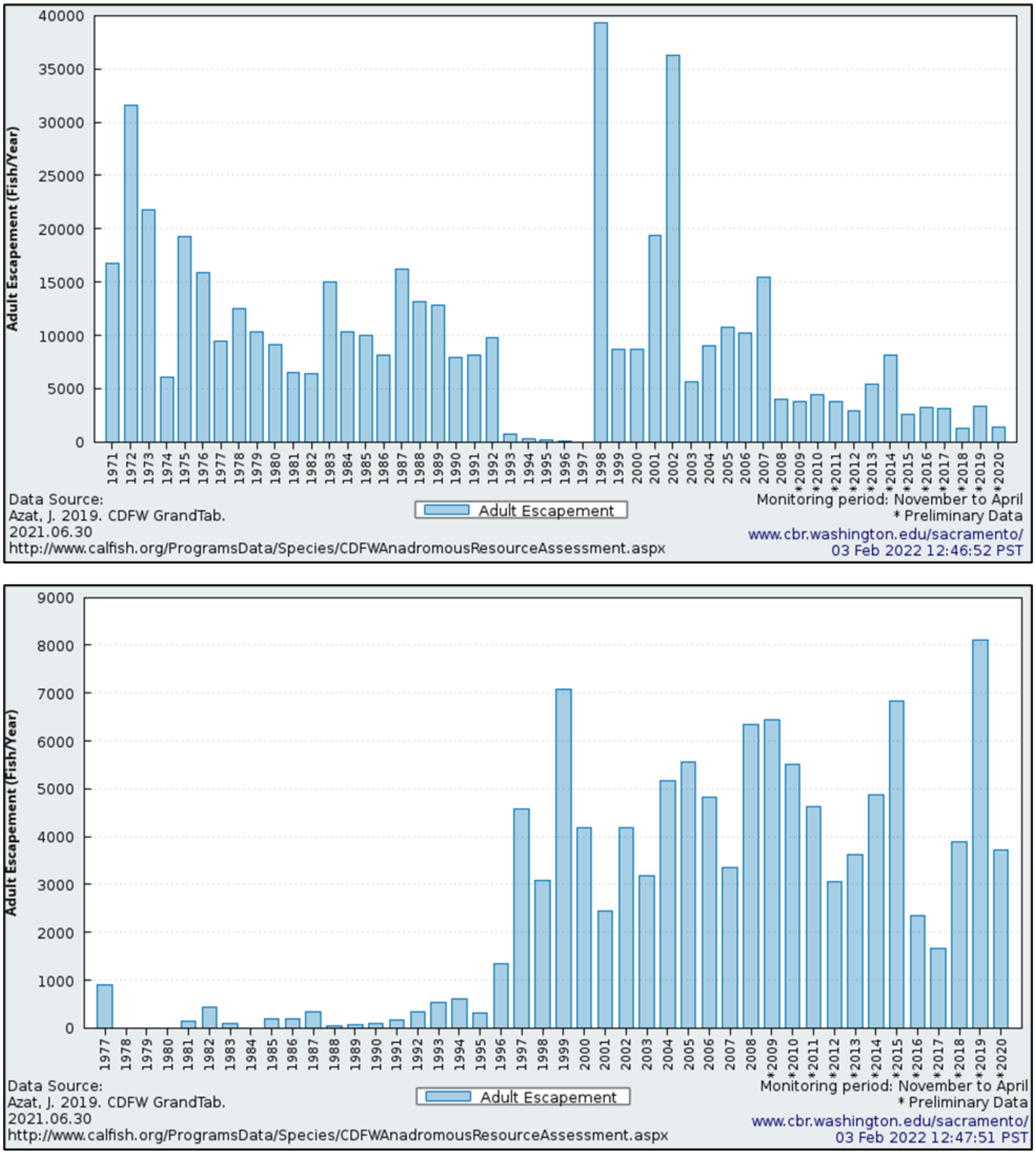
Annual fall-run and late-fall-run Chinook Salmon escapement to the Sacramento River and its tributaries has generally been lower in the last decade than historically, following peaks in the late 1990s to early 2000s (Columbia Basin Research, University of Washington 2022) (Figures 6A-15 and 6A-16). Hatchery fall-run escapement was relatively consistent at approximately 50,000–60,000 fish during 2014–2018 (Figure 6A-15), with hatchery escapement of late-fall run in recent years estimated to be greater than in-river numbers (Figure 6A-16). Studies have suggested that hatchery-produced Chinook Salmon may contribute large (approximately 90 percent) proportions of Chinook Salmon to the mixed-stock fishery along the central California coast (Barnett-Johnson et al. 2007). Sturrock et al. (2019) found that transport distance of hatchery-origin juvenile Chinook Salmon to release sites had increased over time (particularly during droughts) and was strongly associated with straying rate (averaging 0–9 percent vs. 7–89 percent for salmon released on site vs. in the bay upstream of Golden Gate Bridge, respectively), increasing the effects of hatchery releases on natural spawners. The authors suggested that decreasing variation in release location and timing could reduce spatiotemporal buffering, narrowing ocean arrival timings and increasing risk of mismatch with peak prey production. The percentage of hatchery-origin fish released downstream of the Delta has been variable over time. For example, from the mid-1980s to 2012, the proportion of hatchery fall-run Chinook Salmon juveniles released downstream of the Delta by state and federal hatcheries varied from around 20 to 60 percent (Huber and Carlson 2015). Similarly, from 2013 to 2017, the percentage of juvenile fall-run and spring-run Chinook Salmon released by state Central Valley hatcheries downstream of the Delta varied between 24 percent (2016) and 60 percent (2013) (California Department of Fish and Wildlife 2018).



Source: Columbia Basin Research, University of Washington 2022.

Note: Vertical axis scale differs between upper and lower panel.

Figure 6A-15. Fall-Run Chinook Salmon Adult In-River (Upper) and Hatchery (Lower) Annual Escapement in the Central Valley, 1952–2020



Source: Columbia Basin Research, University of Washington 2022.

Note: Vertical axis scale differs between upper and lower panel.

Figure 6A-16. Late Fall–Run Chinook Salmon Adult In-River (Upper) and Hatchery (Lower) Annual Escapement in the Central Valley

Beginning in 1995, restoration actions in Clear Creek have had a clear effect on fall-run Chinook Salmon populations. The combined actions have contributed to a near fourfold increase in escapement of fall-run Chinook Salmon to Clear Creek (population estimates average 1,749 from 1967 to 1991 and 7,333 from 1992 to 2017) (Clear Creek Technical Team 2018). Based on carcass surveys and juvenile outmigration trapping, fall-run Chinook Salmon typically spawn in Clear Creek from late September through early December, and peak outmigration of juveniles occurs in January and February (Earley et al. 2013:1, 11–12).

For late-fall-run Chinook Salmon generally (not just those in Clear Creek), peak spawning time is typically from October to November, but can continue through December and into January. Juveniles typically emerge from the gravel in December through March and rear in fresh water for one to seven months (Moyle et al. 2015:543–552). Fall- and late-fall-run Chinook Salmon generally outmigrate as age-0 fish, although some (8.3 percent in 2011) late-fall-run Chinook Salmon juveniles outmigrate in their second year. Outmigration is generally from April to June for late-fall-run and November to May for fall-run Chinook Salmon. Peak outmigration for fall-run in Clear Creek was in late December and the late-fall run peak outmigration was from mid-April to mid-May (Schraml et al. 2018).

Feather River fall-run Chinook Salmon are partially hybridized with Feather River spring-run Chinook Salmon, but they largely maintain separate fall and spring upstream adult migrations. Fall-run adults return to the Feather River as sexually mature fish and spawn from September into December. The fall-run spawning period begins after the spring-run spawning period, but the spawning periods overlap considerably, leading to superimposition of spring-run redds by subsequently spawning fall-run adults. For this reason, a separation weir has been proposed to physically separate Central Valley spring-run and fall-run Chinook Salmon in the river (National Marine Fisheries Service 2016c). Suitable water temperatures for Chinook Salmon spawning in the Feather River are 42 °F to 58 °F (5.5 °C to 14.4 °C). Incubation may extend through March with suitable incubation temperatures between 48 °F and 58 °F (8.9 °C to 14.4 °C) (California Department of Water Resources 2007:4.4-29). Studies have confirmed that juvenile rearing and probably some adult spawning are associated with secondary channels within the Feather River LFC. The lower velocities, smaller substrate size, and greater amount of cover (compared to the main river channel) likely make these side channels more suitable for juvenile Chinook Salmon rearing. Currently, this type of habitat comprises less than 1 percent of the available habitat in the LFC (California Department of Water Resources 2007:4.4-16). Juvenile Chinook Salmon in the Feather River have been reported to outmigrate as YOY (Seesholtz et al. 2004) and most appear to migrate out of the Feather River within days of emergence (National Marine Fisheries Service 2016b). Juvenile outmigration from the Feather River is generally from mid-November through June, with peak outmigration occurring from January through March (National Marine Fisheries Service 2016b).

The American River historically supported fall-run and perhaps late-fall-run Chinook Salmon (Williams 2001). Both natural-origin and hatchery-produced Chinook Salmon spawn in the lower American River. An analysis by Palmer-Zwahlen et al. (2018) found that constant fractional marking results from 2013 show that approximately 86 percent of the fall-run Chinook Salmon spawners returning to Nimbus Hatchery were hatchery-origin. Further, 71 percent of fall-run Chinook Salmon recorded at the Hatchery Weir and 65 percent of carcasses were identified as hatchery fish. Adult fall-run Chinook Salmon enter the lower American River from about mid-September through January, with peak migration from approximately mid-October through December (Williams 2001). Spawning in the American River occurs from about mid-October through early February, with peak spawning from mid-October through December. Chinook Salmon spawning occurs within an 18-mile stretch from Paradise Beach to Nimbus Dam; however, most spawning occurs in the uppermost 3 miles (4.8 km) (California Department of Fish and Game 2012a:95). Chinook Salmon egg and alevin incubation occurs in the lower American River from about mid-October through April. This period varies widely from year to year, although most incubation occurs from about mid-October through January. Chinook Salmon juveniles rear in the American River from about January to May (Snider and Titus 1995, 2002). Most Chinook Salmon outmigrate from the lower American River as fry between December and July; outmigration peaks February to March (Snider and Titus 2002; Pacific States Marine Fisheries Commission 2014).

In the Stanislaus River, data collected by private fishery consultants, nonprofit organizations, and CDFW demonstrate that the majority of fall-run Chinook Salmon adults migrate upstream from late September through December with peak migration from late October through early November. Most Chinook Salmon spawning occurs between Riverbank (RM 33) and Goodwin Dam (RM 58.4) (U.S. Bureau of Reclamation 2012:6). Based on redd surveys conducted by FISHBIO, peak spawning typically occurs in November with roughly 7 percent of spawning occurring prior to November 1, and 2 percent prior to October 15. The few redds created during late September and early October are typically in the reach just below Goodwin Dam. By late October, the amount of spawning in downstream locations increases as water temperatures decrease, and the median redd location is typically around Knights Ferry (State Water Resources Control Board 2015). In 2010, over 20 percent of the fall-run Chinook Salmon observed passing the Stanislaus River weir had adipose fin clips, indicating the presence of a coded wire tag in their snout. Since there is no hatchery on the Stanislaus River and no hatchery releases into this tributary have occurred since 2006, it is apparent that straying from other rivers is occurring (FISHBIO Environmental 2010). Subsequent surveys have also found a high proportion of hatchery-origin fish on the Stanislaus River spawning grounds (e.g., 2012: 83 percent [Palmer-Zwahlen and Kormos 2013:11]; 2014: 66 percent [Palmer-Zwahlen et al. 2018:8]). Rotary screw trap data indicate that about 99 percent of salmon juveniles migrate out of the Stanislaus River from January through May (Stanislaus River Fish Group 2004). Fry migration generally occurs from January through March, followed by smolt migration from April through May (U.S. Bureau of Reclamation 2012:8). Watry et al. (2012) found that in both 2010 and 2011, peak passage during the pre-smolt period generally corresponded with flow pulses. Zeug et al. (2014) examined 14 years of rotary screw trap data on the lower Stanislaus River and found a strong positive response in survival, the proportion of pre-smolt migrants and the size of smolts when cumulative flow and flow variance were greater. From the data, they concluded that periods of high discharge in combination with high discharge variance are important for successful outmigration as well as migrant size and the maintenance of diverse migration strategies.

Mesick (2001) surmised that when water exports are high relative to San Joaquin River flows, little, if any, San Joaquin River water reaches San Francisco Bay, where it may be needed to help attract the salmon back to the Stanislaus River. During mid-October from 1987 through 1989, when export rates exceeded 400 percent of Vernalis flows, Mesick (2001) found that straying rates ranged between 11 and 17 percent. In contrast, straying rates were estimated to be less than 3 percent when Delta export rates were less than about 300 percent of San Joaquin River flow at Vernalis during mid-October. Marston et al. (2012) provided statistical relationships between straying rate and flow and exports, concluding that the results indicate that flow is the primary factor but that empirical data indicate that little if any pulse flow leaves the Delta when south Delta exports are elevated; they hypothesized that exports in combination with pulse flows may explain straying. Peterson et al. (2017) studied environmental factors and management actions influencing upstream migration patterns of adult fall-run Chinook in the Stanislaus River. They found that the Head of Old

River rock barrier had positive and consistent influences on daily counts in the years it was installed and that managed pulse flows resulted in immediate increases in daily passages, but the response was brief and represented a small portion of the total run.

One of the limiting factors for juvenile fall-run Chinook Salmon from the Stanislaus River and elsewhere in the San Joaquin River Basin appears to be the high rates of mortality for juveniles migrating through dredged channels in the Delta, particularly the Stockton Deep Water Ship Channel (Newcomb and Pierce 2010). Pickard et al. (1982) reported that the survival of juvenile fish in the ship channel is highest during flood flows or when a barrier is placed at the Head of Old River that more than doubles the flow in the ship channel. As noted in the account for spring-run Chinook Salmon, the Collaborative Adaptive Management Team Salmonid Scoping Team work suggests that high correlations between inflows and exports make it difficult to evaluate their effects on salmon survival independently using statistical methods. The Stanislaus River Fish Group (SRFG) (2004) noted that escapement is also directly correlated with springtime flows, when each brood migrates downstream as smolts. However, the cause of the mortality in the ship channel has not been studied. Buchanan and Skalski (2019) found through-Delta survival of acoustically tagged fall-run Chinook Salmon smolts was positively associated with Old River flow in the strongly tidal interior Delta but not with higher San Joaquin River flow either entering the Delta from upstream or in the Delta near the riverine/tidal interface. Survival in the upstream, more riverine region of the Delta was positively associated with San Joaquin River flow measured at the riverine/tidal interface and average net flow in the interior Delta, which the authors suggested provided evidence of different mechanisms driving survival in the upstream versus downstream reaches of the Delta (Buchanan and Skalski 2019). A large portion of juvenile fall-run Chinook Salmon surviving through the Delta from the San Joaquin River Basin move through the CVP salvage facility (Buchanan et al. 2018).

Dredging for gravel and gold, regulated flows, and the diking of floodplains for agriculture have substantially limited the availability of spawning and rearing habitat for fall-run Chinook Salmon in the Stanislaus River (U.S. Bureau of Reclamation 2019). Reclamation has conducted spawning gravel augmentation to improve spawning and rearing habitats in the Stanislaus River reach between Goodwin Dam and Knights Ferry most years since 1999. The dredged areas also contain an abundance of large predatory fish, although SRFG concluded that there is uncertainty about whether predation is a substantial source of mortality for juvenile salmon. SRFG (2004) also concluded that water diversions for urban and agricultural use in all three San Joaquin River tributaries, which reduce flows and potentially result in unsuitably warm water temperatures during spring and fall, affecting fall-run Chinook Salmon juvenile rearing and adult and juvenile migration in the lower San Joaquin River and Delta.

General life stage timing for Central Valley fall-run and late-fall-run Chinook salmon is summarized in Tables 6A-8, 6A-9, 6A-10a, 6A-10b, 6A-10c, 6A-10d, and 6A-10e.

Table 6A-8. Generalized Temporal Occurrence of Central Valley Fall-Run Chinook Salmon Adults

	Month																							
Location	**Jan**		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sep**		**Oct**		**Nov**		**Dec**	
Adults																								
Delta	N	N	N	N	N	N	N	N	N	N	M	M	H	H	H	H	H	H	H	H	H	H	M	M
Sacramento River Basin	N	N	N	N	N	N	N	N	N	N	N	N	L	L	L	L	M	M	H	H	H	H	H	H
San Joaquin River	N	N	N	N	N	N	N	N	N	N	N	N	N	N	L	L	M	M	H	H	H	H	M	M

Sources: Vogel and Marine 1991; Yoshiyama et al. 1998; Martin et al. 2001; Moyle 2002.

Relative Abundance: H = high (blue); M = medium (green); L = low (yellow); N = none.

Table 6A-9. Generalized Temporal Occurrence of Central Valley Late-Fall-Run Chinook Salmon Adults

	Month																							
Location	**Jan**		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sep**		**Oct**		**Nov**		**Dec**	
Delta	H	H	H	H	H	H	M	M	M	M	N	N	N	N	N	N	N	N	M	M	M	M	M	M
Sacramento River Basin	H	H	H	H	H	H	M	M	N	N	N	N	N	N	N	N	N	N	M	M	M	M	M	M

Sources: Vogel and Marine 1991; Yoshiyama et al. 1998; Martin et al. 2001; Moyle 2002.

Relative Abundance: H = high (blue); M = medium (green); L = low (yellow); N = none.

Table 6A-10a. Frequency of Occurrence (Percent) of Adipose Fin-Unclipped Fall-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	99.8% (586)	99.8% (555)	99.4% (641)	99.5% (581)	100% (607)	100% (665)	98.9% (742)	92.7% (715)	64.2% (670)	35.1% (710)	17.8% (692)	99.3% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	67.8% (298)	75.6% (270)	55.4% (307)	65.5% (313)	46.8% (278)	13.5% (111)	2.8% (72)	3.2% (62)	0% (204)	0% (325)	0.9% (337)	33.4% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	58.1% (413)	64.8% (386)	58.4% (423)	56% (393)	40.7% (349)	23.1% (130)	Not Sampled	0% (17)	0.7% (148)	0% (344)	0% (345)	30.7% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	45.8% (2,784)	59.7% (2,149)	54.5% (2,220)	36.8% (2,060)	19.1% (2,204)	4.4% (2,107)	0.2% (2,043)	0% (2,090)	0% (2,086)	0.1% (3,316)	0% (3,480)	18.8% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	32.3% (3,402)	48.6% (3,273)	39.6% (3,524)	54% (3,502)	58.3% (2,908)	16% (2,316)	4.1% (2,700)	0.9% (2,637)	0.2% (2,591)	0% (2,664)	0% (2,631)	8.3% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	0% (4,225)	1% (3,257)	3.4% (3,445)	60.2% (4,738)	89.9% (6,348)	52.9% (3,539)	8.7% (2,441)	1.9% (2,264)	0.9% (2,290)	0.7% (2,704)	0.3% (2,612)	0.1% (3,718)
Salvage	1/1/1993-8/10/2023	Days	20% (955)	39.2% (874)	39.6% (954)	65.1% (930)	83.6% (960)	54.2% (930)	11.1% (960)	2.4% (940)	2.8% (900)	3.1% (929)	3.8% (900)	4.1% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one fall-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Table 6A-10b. Frequency of Occurrence (Percent) of Adipose Fin-Clipped Fall-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	2% (586)	1.6% (555)	20.7% (641)	59.6% (581)	70.5% (607)	17.7% (665)	2.8% (742)	0.1% (715)	0.3% (670)	0.4% (710)	7.2% (692)	15.5% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	0% (298)	0% (270)	6.5% (307)	32.6% (313)	15.8% (278)	0% (111)	1.4% (72)	0% (62)	0% (204)	0% (325)	0% (337)	0.3% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	0.2% (413)	0.8% (386)	6.4% (423)	40.2% (393)	20.6% (349)	0% (130)	Not Sampled	0% (17)	0% (148)	0% (344)	0% (345)	3% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	0% (2,784)	3.2% (2,149)	5.5% (2,220)	5.2% (2,060)	4.3% (2,204)	0.9% (2,107)	0% (2,043)	0% (2,090)	0% (2,086)	0.1% (3,316)	0% (3,480)	0.1% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	0% (3,402)	2.2% (3,273)	2.1% (3,524)	27.5% (3,502)	22.1% (2,908)	1.9% (2,316)	0% (2,700)	0% (2,637)	0% (2,591)	0% (2,664)	0% (2,631)	0% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	0.1% (4,225)	0.2% (3,257)	2.9% (3,445)	43.8% (4,738)	48% (6,348)	8.6% (3,539)	0.2% (2,441)	0% (2,264)	0% (2,290)	3.5% (2,704)	0.8% (2,612)	0.3% (3,718)
Salvage	1/1/1993-8/10/2023	Days	3% (955)	0% (874)	2.2% (954)	14.3% (930)	39% (960)	16.8% (930)	0% (960)	0% (940)	1% (900)	2.2% (929)	4.2% (900)	10.8% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one fall-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Table 6A-10c. Frequency of Occurrence (Percent) of Adipose Fin-Unclipped Late-Fall-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	20.1% (586)	2.7% (555)	2.5% (641)	76.9% (581)	66.9% (607)	62.3% (665)	75.1% (742)	76.2% (715)	80.7% (670)	91% (710)	90.5% (692)	72.6% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	2% (298)	0.7% (270)	0% (307)	6.1% (313)	1.8% (278)	0% (111)	0% (72)	0% (62)	0% (204)	1.8% (325)	3.9% (337)	7.8% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	5.1% (413)	0% (386)	0% (423)	3.6% (393)	1.1% (349)	0% (130)	Not Sampled	0% (17)	0% (148)	2.3% (344)	4.3% (345)	13% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	0.4% (2,784)	0.2% (2,149)	0% (2,220)	4.6% (2,060)	2.3% (2,204)	0.4% (2,107)	0.2% (2,043)	0.1% (2,090)	0% (2,086)	0.1% (3,316)	0.6% (3,480)	2.5% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	0.3% (3,402)	0% (3,273)	0% (3,524)	0.2% (3,502)	0.3% (2,908)	0% (2,316)	0.3% (2,700)	0.2% (2,637)	0.3% (2,591)	0.3% (2,664)	0.6% (2,631)	1.8% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	1.9% (4,225)	0.5% (3,257)	0% (3,445)	0% (4,738)	0% (6,348)	0% (3,539)	0.1% (2,441)	0.1% (2,264)	0.7% (2,290)	0.4% (2,704)	1.7% (2,612)	7% (3,718)
Salvage	1/1/1993-8/10/2023	Days	8.5% (955)	2.3% (874)	0.2% (954)	0.3% (930)	0% (960)	0.2% (930)	0% (960)	0.1% (940)	0.2% (900)	0.8% (929)	2.3% (900)	15.1% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one late fall-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Table 6A-10d. Frequency of Occurrence (Percent) of Adipose Fin-Clipped Late-Fall-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in Sacramento River and Delta Sampling Programs

Location	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Sacramento River RST at Red Bluff	7/18/1994-7/31/2023	Days	53.2% (586)	7.7% (555)	0.5% (641)	0.3% (581)	0.3% (607)	1.1% (665)	0% (742)	0% (715)	0.1% (670)	0.3% (710)	20.4% (692)	56.6% (558)
Sacramento River RST at Tisdale	7/6/2010-12/18/2022	Days	11.7% (298)	3% (270)	0% (307)	0% (313)	0% (278)	0% (111)	0% (72)	0% (62)	0.5% (204)	0% (325)	0.9% (337)	24.1% (320)
Sacramento River RST at Knights Landing	10/2/2006-10/22/2022	Days	20.8% (413)	2.8% (386)	0% (423)	0.5% (393)	0% (349)	0% (130)	Not Sampled	0% (17)	0% (148)	0.3% (344)	0.9% (345)	20.4% (401)
Delta and Sacramento River Beach Seines	1/3/2000-7/29/2022	Seine Sets	4.2% (2,784)	1.8% (2,149)	0.5% (2,220)	0% (2,060)	0% (2,204)	0% (2,107)	0% (2,043)	0% (2,090)	0% (2,086)	0% (3,316)	0.3% (3,480)	2.9% (3,325)
Sacramento Trawel at Sherwood Harbor	1/3/2000-7/29/2022	Trawl Tows	7.3% (3,402)	1.9% (3,273)	0.1% (3,524)	0.1% (3,502)	0% (2,908)	0% (2,316)	0% (2,700)	0% (2,637)	0% (2,591)	0% (2,664)	0.1% (2,631)	6.7% (3,349)
Midwater Trawel at Chipps Island	1/2/2000-7/29/2022	Trawl Tows	23.6% (4,225)	9.3% (3,257)	2.7% (3,445)	0.3% (4,738)	0.1% (6,348)	0% (3,539)	0% (2,441)	0% (2,264)	0% (2,290)	0% (2,704)	0.6% (2,612)	18.3% (3,718)
Salvage	1/1/1993-8/10/2023	Days	27.5% (955)	2.4% (874)	0% (954)	0% (930)	0% (960)	0% (930)	0% (960)	0% (940)	0% (900)	0% (929)	0.1% (900)	25.2% (930)

Note: RST = Rotary Screw Trap. Frequency of occurrence is percentage of sampling units with at least one late fall-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Intensity of shading increases with increasing frequency of occurrence. Numbers in parentheses indicate number of sampling units.

Table 6A-10e. Frequency of Occurrence (Percent) of Fall-Run Chinook Salmon Juveniles (Based on Length-at-Date Criteria) in the Mossdale Trawl Sampling Program

Adipose Fin	Sampling Dates	Sampling Units	January	February	March	April	May	June	July	August	September	October	November	December
Clipped	1/3/2000-7/29/2022	Trawl Tows	0% (2,427)	0% (2,300)	0% (2,623)	9.1% (5,232)	17.6% (6,239)	2.7% (3,503)	0% (2,665)	0% (2,601)	0% (2,339)	0% (2,147)	0% (2,231)	0% (2,118)
Unclipped	1/3/2000-7/29/2022	Trawl Tows	2.7% (2,427)	5.5% (2,300)	6.7% (2,623)	29.4% (5,232)	48.5% (6,239)	18.3% (3,503)	0.6% (2,665)	0% (2,601)	0% (2,339)	0% (2,147)	0% (2,231)	0.2% (2,118)

Note: Frequency of occurrence is percentage of sampling units with at least one late fall-run Chinook Salmon juvenile (based on length-at-date criteria) collected. Numbers in parentheses indicate number of sampling units.

6A.1.6 Steelhead—Central Valley DPS

The California Central Valley steelhead DPS was originally listed as threatened under the ESA on March 19, 1998 (63 FR 13347), and the listing was reaffirmed January 5, 2006 (71 FR 834) and updated April 14, 2014 (79 FR 20802). The DPS includes naturally spawned anadromous *Oncorhynchus mykiss* (steelhead) originating below natural and human-made impassable barriers from the Sacramento and San Joaquin rivers and their tributaries; excludes such fish originating from San Francisco and San Pablo bays and their tributaries. This DPS includes steelhead from two artificial propagation programs: the Coleman National Fish Hatchery Program, and the FRFH Program (79 FR 20802–20810). Factors contributing to listing for West Coast steelhead, including the Central Valley DPS, include decline primarily caused by destruction and modification of habitat, overutilization for recreational purposes, and natural and human-made factors (63 FR 13347–13354). NMFS (2014a:60) described some of the most important stressors to the Central Valley steelhead DPS to be passage impediments and barriers, warm water temperatures for rearing, hatchery effects, limited quantity and quality of rearing habitat, predation, and entrainment.

Upstream migration of Central Valley steelhead begins with estuarine entry from the ocean as early as July and continues through February or March in most years (McEwan and Jackson 1996; National Marine Fisheries Service 2009). Populations of steelhead occur primarily within the watersheds of the Sacramento River Basin, although not exclusively. Steelhead can spawn more than once, with postspawn adults (typically females) potentially moving back downstream through the Delta after completion of spawning in their natal streams.

Upstream migrating adult steelhead enter the Sacramento River and San Joaquin River basins through their respective mainstem river channels. Steelhead entering the Mokelumne River system (including Dry Creek and the Cosumnes River) and the Calaveras River system to spawn are likely to move up the mainstem San Joaquin River channel before branching off into the channels of their natal rivers, although some may detour through the south Delta waterways and enter the San Joaquin River through the Head of Old River.

Steelhead entering the San Joaquin River Basin appear to have a later spawning run, with adults entering the system starting in late October through December, indicating that migration up through the Delta may begin a few weeks earlier. During fall, warm water temperatures in the south Delta waterways and water quality impairment because of low dissolved oxygen (DO) at Stockton have been suggested as potential barriers to upstream migration (National Marine Fisheries Service 2009). Reduced water temperatures, as well as rainfall runoff and flood control release flows, provide the stimulus to adult steelhead holding in the Delta to move upriver toward their spawning reaches in the San Joaquin River tributaries. Adult steelhead may continue entering the San Joaquin River Basin through winter.

Juvenile steelhead can be found in all waterways of the Delta, but particularly in the main channels leading from their natal river systems (National Marine Fisheries Service 2009). Juvenile steelhead are recovered in trawls from October through July at Chipps Island and at Mossdale. Chipps Island catch data indicate there is a difference in the outmigration timing between wild and hatchery-reared steelhead smolts from the Sacramento and eastside tributaries. Hatchery fish are typically recovered at Chipps Island from January through March, with a peak in February and March corresponding to the schedule of hatchery releases of steelhead smolts from the Central Valley hatcheries (Nobriga and Cadrett 2001; U.S. Bureau of Reclamation 2008:3-11). The timing of wild (unmarked) steelhead outmigration is more spread out, and based on salvage records at the CVP and SWP fish collection facilities, outmigration occurs over approximately six months with the highest levels of recovery in February through June (Aasen 2011, 2012). Steelhead are salvaged annually at the project export facilities (e.g., 4,631 fish were salvaged in 2010, and 1,648 in 2011) (Aasen 2011, 2012).

Outmigrating steelhead smolts enter the Delta primarily from the Sacramento or San Joaquin rivers. Mokelumne River steelhead smolts can either follow the north or south branches of the Mokelumne River through the central Delta before entering the San Joaquin River, although some fish may enter farther upstream if they diverge from the south branch of the Mokelumne River into Little Potato Slough. Calaveras River steelhead smolts enter the San Joaquin River downstream of the Port of Stockton. Although steelhead have been routinely documented by CDFW in trawls at Mossdale since 1988 (San Joaquin River Group Authority 2011), it is unknown whether successful outmigration occurs outside the historical seasonal installation of the barrier at the Head of Old River (between April 15 and May 15 in most years). Prior to the installation of the Head of Old River Fish Control Gate, steelhead smolts exiting the San Joaquin River Basin could follow one of two routes to the ocean, either staying in the mainstem San Joaquin River through the central Delta, or entering the Head of Old River and migrating through the south Delta and its associated network of channels and waterways.

Central Valley steelhead use the San Francisco and San Pablo Bays as a migration corridor to and from the ocean. The juveniles move quickly through the bays on their way to the ocean, preying on a variety of macroinvertebrates and small fish.

Steelhead are broadly divided into two life history types, summer-run steelhead and winter-run steelhead, based on their state of sexual maturity at the time of river entry. Only winter-run steelhead are currently found in Central Valley rivers and streams. 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 system in Wet years) (McEwan 2001). Presently, Central Valley steelhead are found in the Sacramento River downstream of Keswick Dam, in major tributary rivers and creeks in the Sacramento River watershed, and in major tributaries of the San Joaquin River (Stanislaus, Tuolumne, and Merced rivers) and Delta (Mokelumne and Calaveras rivers). 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.

Adult steelhead migrate upstream past the Fremont Weir between August and March, but primarily from August through October, and they migrate upstream past RBDD during all months of the year, but primarily during September and October (National Marine Fisheries Service 2009). The primary spawning area used by steelhead in the Sacramento River is the area from Keswick Dam downstream to RBDD. Unlike Pacific salmon, steelhead may live to spawn more than once and generally rear in freshwater streams for two to four years before outmigrating to the ocean. Both spawning areas and migratory corridors are used by juvenile steelhead for rearing prior to outmigration. The Sacramento River functions primarily as a migration channel, although some rearing habitat remains in areas with setback levees (primarily upstream of Colusa) and flood bypasses (e.g., Yolo Bypass) (National Marine Fisheries Service 2009).

Recent steelhead monitoring data are scarce for the upper portion of the Sacramento River system. Hallock (1989) reported that steelhead had declined drastically in the Sacramento River upstream of the Feather River confluence. In the 1950s, the average estimated spawning population size upstream of the Feather River confluence was 20,540 fish (McEwan and Jackson 1996). In 1991–1992, the annual run size for the total Sacramento River system was likely fewer than 10,000 adult fish (McEwan and Jackson 1996). From 1967 to 1993, the estimated number of steelhead passing the Red Bluff Pumping Plant ranged from a low of 470 to a high of 19,615 (California Hatchery Scientific Review Group 2012). Steelhead escapement surveys at Red Bluff ended in 1993.

Both steelhead and resident (non-anadromous) rainbow trout (*O. mykiss*) occur in Clear Creek. Adult Central Valley steelhead populations in Clear Creek have been relatively stable between 2003 and 2011, with redd counts ranging from 42 to 409, with an average of 176 (Giovannetti et al. 2013; Provins and Chamberlain 2019). Adult Central Valley steelhead spawn in Clear Creek from early December to mid-March. Steelhead rear in Clear Creek year-round, and outmigration can occur in any month, although peak outmigration in 2011 was from February to June (Schraml et al. 2018).

California Central Valley steelhead adults migrate into the Feather River between July and March, and redd construction occurs from late December to March, peaking in late January (Federal Energy Regulatory Commission 2007:178; McEwan 2001). Spawning in the Feather River primarily occurs within the LFC between the Fish Barrier Dam and the Thermalito Afterbay outlet, although a small amount of spawning occurs downstream of the Thermalito Afterbay outlet (Federal Energy Regulatory Commission 2007:169, 178). Nearly half of all observed redds are constructed in the uppermost mile of the LFC (Federal Energy Regulatory Commission 2007:178). Fry begin to outmigrate in February, soon after emerging, with the majority outmigrating between March and mid-April. Most juveniles outmigrate by September, but a small portion of juveniles that do not outmigrate rear in the river for up to one year, most often in secondary channels of the LFC (Federal Energy Regulatory Commission 2007:169).

Although some spawning by steelhead in the American River occurs naturally (Hannon and Deason 2008), the population is supported primarily by the Nimbus Fish Hatchery. The total estimated steelhead return to the river (spawning naturally and in the hatchery) has ranged from 946 to 3,426 fish, averaging 2,184 fish per year from 2002 to 2010 (California Hatchery Scientific Review Group 2012). Steelhead spawning surveys have shown approximately 300 steelhead spawning in the river each year (Hannon and Deason 2008). Lindley et al. (2007) classifies the listed (i.e., naturally spawning) population of American River steelhead at a high risk of extinction because it is reportedly mostly composed of winter-run steelhead originating from Nimbus Fish Hatchery; possibly up to 90 percent of spawners are of hatchery origin (Hannon and Deason 2008). NMFS considers the American River population to be important to the survival and recovery of the species (National Marine Fisheries Service 2014a).

Steelhead from the American River (collected from both the Nimbus Fish Hatchery and the American River) are genetically more similar to Eel River and Mad River steelhead than other Central Valley steelhead stocks because individuals from these rivers were used as broodstock for Nimbus Hatchery (Nielsen et al. 2005; California Hatchery Scientific Review Group 2012:30). American River steelhead exhibit a slightly later upstream migration period than other Central Valley steelhead (Lee and Chilton 2007).

Adult steelhead migrate up the American River from October through April with a peak occurring from December through March (Surface Water Resources 2001). Adult steelhead have been caught in the Nimbus Fish Hatchery trap as early as the first week of October. Spawning typically occurs in the lower American River between late December and early April, with the peak occurring in late February to early March (Hannon and Deason 2008). Spawning occurs between Nimbus Dam and Paradise Beach, although approximately 90 percent of spawning occurs upstream of the Watt Avenue Bridge (Hannon and Deason 2008). Embryo incubation occurs shortly after spawning in late December and generally extends through May, although incubation can occur into June in some years (Surface Water Resources 2001). Although steelhead embryo and alevin mortality from high flows in the American River has not been documented, flows high enough to mobilize spawning gravels and scour or entomb redds have been recorded during the spawning and embryo incubation periods (National Marine Fisheries Service 2009). Juvenile *O. mykiss* are present year-round throughout the lower American River, with rearing generally upstream of spawning areas. Juveniles can rear in the lower American River for a year or more before outmigrating as smolts from January through June (Snider and Titus 2000; Surface Water Resources 2001), although it is rare to find individuals older than YOY fry and parr (Snider and Titus 2002; Pacific States Marine Fisheries Commission 2014). Peak juvenile steelhead outmigration occurs from March through May (McEwan and Jackson 1996; Surface Water Resources 2001; Pacific States Marine Fisheries Commission 2014). Juvenile steelhead rear in the lower American River from Nimbus Dam to Paradise Beach. During summer months, juveniles occur in most major riffle areas, with the highest densities near the higher-density spawning areas (U.S. Bureau of Reclamation 2008:3-21). The number of juveniles in the American River decreases throughout summer (U.S. Bureau of Reclamation 2008:3-20). Juveniles experience water temperature-related stress during summer and early fall (Water Forum 2005; National Marine Fisheries Service 2014a) despite laboratory studies indicating that American River steelhead may be more tolerant of high temperatures than steelhead from other rivers (Myrick and Cech 2004).

Central Valley steelhead were thought to be extirpated from the San Joaquin River system (National Marine Fisheries Service 2009). However, monitoring has detected small self-sustaining (i.e., of natural origin, not of hatchery origin) populations of steelhead in the Stanislaus River and other streams previously thought to be devoid of steelhead (Stanislaus River Fish Group 2003; McEwan 2001). There is a catch-and-release steelhead fishery in the lower Stanislaus River between January 1 and October 15. Surveys of *O. mykiss* (resident rainbow trout and the anadromous steelhead) abundance and distribution conducted annually since 2009 have documented a relatively stable population. River-wide abundance estimates from 2009 to 2014 have averaged just over 20,220 (all life stages combined) and have never been estimated to be less than about 14,000 (2009). The highest densities and abundances of *O. mykiss* are consistently found in Goodwin Canyon. Key factors that may contribute to higher than average abundances in the Stanislaus River (relative to other San Joaquin River tributaries) include high gradient reaches that are typically associated with fast-water habitats, particularly in Goodwin Canyon (State Water Resources Control Board 2015).

Historically, the distribution of steelhead extended into the headwaters of the Stanislaus River (Yoshiyama et al. 1996). Steelhead currently can migrate more than 58 miles (93.3 km) up the Stanislaus River to the base of Goodwin Dam. In the Stanislaus River, there is little data regarding the migration patterns of adult steelhead since adults generally migrate during periods when river flows and turbidity are high, making fish difficult to observe with standard adult monitoring techniques. Stanislaus River weir data indicate that steelhead migrate upstream, through the south Delta and lower San Joaquin River, between September and March (U.S. Bureau of Reclamation

2014). High Delta export rates relative to San Joaquin River flows at Vernalis, when adults are migrating through the Delta (presumably December through May), may result in adults straying to the Sacramento River Basin.

It is believed that steelhead spawn primarily between December and March in the Stanislaus River. Although few steelhead spawning surveys have been conducted in the Stanislaus River, spawning *O. mykiss* were documented between Goodwin Dam and Horseshoe Bar in a 2014 spawning survey (U.S. Bureau of Reclamation and California Department of Water Resources 2015:56). The spawning adults require holding and feeding habitat with cover adjacent to suitable spawning habitat. These habitat features are relatively rare in the lower Stanislaus River because of in-river gravel mining and the scouring of gravel from riffles in Goodwin Canyon.

Juvenile steelhead rear in the Stanislaus River for at least one year, and usually two years, before migrating to the ocean. As a result, flow, water temperature, and DO concentration in the reach between Goodwin Dam and the Orange Blossom Bridge (their primary rearing habitat) are critical during summer (U.S. Bureau of Reclamation 2012:11).

Small numbers of Central Valley steelhead smolts have been captured in rotary screw traps at Caswell State Park and near Oakdale (FISHBIO Environmental 2007:31; Watry et al. 2007, 2012), and data indicate that steelhead outmigrate primarily from February through May. Rotary screw traps are generally not considered efficient at catching fish as large as steelhead smolts, and the number captured is too small to estimate capture efficiency, so no steelhead smolt outmigration population estimate has been calculated. The capture of these fish in downstream migrant traps and the advanced smolting characteristics exhibited by many of the fish indicate that some steelhead/rainbow trout juveniles might outmigrate to the ocean in spring. However, it is not known whether the parents of these fish were anadromous or fluvial (i.e., migrate within fresh water). Resident populations of steelhead/rainbow trout in large streams are typically fluvial, and migratory juveniles look much like smolts.

Steelhead were historically present in the San Joaquin River, though data on their population levels are lacking (McEwan 2001). The current steelhead population in the San Joaquin River is substantially reduced compared with historical levels, although resident rainbow trout occur throughout the major San Joaquin River tributaries. Additionally, small populations of steelhead persist in the lower San Joaquin River and tributaries (e.g., Stanislaus, Tuolumne, and possibly Merced rivers) (Zimmerman et al. 2009; McEwan 2001). Steelhead/rainbow trout of anadromous parentage occur at low numbers in all three major San Joaquin River tributaries. These tributaries have a higher percentage of resident rainbow trout compared to the Sacramento River and its tributaries (Zimmerman et al. 2009). Presence of steelhead smolts from the San Joaquin River Basin is estimated annually by CDFW based on the Mossdale Trawl (San Joaquin River Group Authority 2011). The sampling trawls capture steelhead smolts, although usually in small numbers. One steelhead smolt was captured and returned to the river during the 2009 sampling period (San Joaquin River Group Authority 2010), and three steelhead were captured and returned in both 2010 and 2011 (Speegle et al. 2013).

General life stage timing for Central Valley steelhead is summarized in Tables 6A-11 and 6A-12.

Table 6A-11. Temporal Occurrence of Central Valley Steelhead by Life Stage

	Month																							
Location	**Jan**		**Feb**		**Mar**		**Apr**		**May**		**Jun**		**Jul**		**Aug**		**Sep**		**Oct**		**Nov**		**Dec**	
Adults																								
Sacramento at Fremont Weir	L	L	L	L	L	N	N	N	N	N	N	L	L	L	L	M	H	H	H	M	L	L	L	L
Sacramento River at Reb Bluff	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	M	M	H	M	L	L	L	L
San Joaquin River	H	H	M	M	L	L	N	N	N	N	N	N	L	L	L	L	M	M	M	M	M	M	H	H
Juvenile																								
Sacramento at Fremont Weir	L	L	L	L	M	M	M	M	M	M	M	M	L	L	L	L	L	L	M	M	M	M	L	L
Sacramento River at Knights Landing	H	H	H	H	M	M	M	M	L	L	L	L	N	N	N	N	N	N	N	N	L	L	L	L
Chipps Island (clipped)	M	M	H	H	M	M	L	L	L	L	N	N	N	N	N	N	N	N	N	N	N	N	L	L
Chipps Island (unclipped)	M	M	M	M	H	H	H	H	H	H	M	M	L	L	N	N	N	N	N	N	N	L	L	L
San Joaquin River at Mossdale	N	N	L	L	M	M	H	H	H	H	L	L	N	N	N	N	N	N	L	L	N	N	N	N

Source: National Marine Fisheries Service 2019:100.

Relative Abundance: H = high (blue); M = medium (green); L = low (yellow); N = none.

Table 6A-12. Temporal Occurrence of Central Valley Steelhead by Life Stage in the Delta

	Month											
Life Stage	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**	**Sep**	**Oct**	**Nov**	**Dec**
Adult	M	M	M	M	H	N	L	M	H	M	M	M
Juvenile	L	M	M	H	H	L	L	N	L	N	N	L
Salvaged	M	H	H	M	L	L	N	N	N	N	L	L

Source: National Marine Fisheries Service 2019:101.

Relative Abundance: H = high (blue); M = medium (green); L=low (yellow); N = none.

6A.1.7 Green Sturgeon—Southern DPS

As summarized by NMFS (2018:14), the southern DPS of North American Green Sturgeon was listed as threatened in 2006 (71 FR 17757) because of the following factors: (1) the Sacramento River contains the only known spawning population; (2) there has been a substantial loss of spawning habitat in the upper Sacramento and Feather rivers; (3) the Sacramento River and Delta system face mounting threats to habitat quality and quantity; and (4) fishery-independent data indicated a decrease in observed numbers of juvenile Green Sturgeon collected. The NMFS (2018:22–26) recovery plan for the species noted numerous threats for the species, among which those of very high rank were altered water flow as a result of channel control structures and impoundments in the San Francisco Bay-Delta Estuary, altered prey base as a result of nonnative species in coastal bays and estuaries and the nearshore marine environment, and global climate change in coastal bays and estuaries.

Green Sturgeon reach maturity around 14 to 16 years of age and can live to be 70 years old, returning to their natal rivers every three to five years for spawning (Van Eenennaam et al. 2005). Adult Green Sturgeon move through the Delta from February through April, arriving at holding and spawning locations in the upper Sacramento River between April and June (Heublein 2006; Kelly et al. 2007). Following their initial spawning run upriver, adults may hold for a few weeks to months in the upper river before moving back downstream in fall (Vogel 2008; Heublein et al. 2009), or they may migrate immediately back downstream through the Delta. Colborne et al. (2022) identified the role of minimum discharge as the primary factor cueing two out-migration life history variations, "early" and "late". These life history variations were similar to behavior exhibited by adult Green Sturgeon on the Rogue River and Klamath River systems (Erickson and Webb 2007; Benson et al. 2007). Miller et al. (2020) observed two adults that remained in the Sacramento River spawning reach for nearly a year, exiting in January or February in the year following spawning. However, the longer holding observed by Miller et al. (2020) of nearly a year has not been observed in other systems and was suggested by the authors as possibly being a feature of the Sacramento River population, individual variation, or related to the Dry years during their study that may have delayed the flow cues needed to outmigrate. Anglers have reported catching a few Green Sturgeon in recent years in the San Joaquin River (California Department of Fish and Game 2012b), and this has been confirmed by capture during research surveys (Anderson et al. 2018; Root et al. 2020).

Similar to other estuaries along the West Coast of North America, adult and subadult Green Sturgeon frequently congregate in the San Francisco Estuary during summer and fall (Lindley et al. 2008). Specifically, adults and subadults may reside for extended periods in the central Delta as well as in Suisun and San Pablo bays, presumably for feeding, because bays and estuaries are preferred feeding habitat rich in benthic invertebrates (e.g., amphipods, bivalves, insect larvae). In part because of their bottom-oriented feeding habits, sturgeon are at risk of harmful accumulations of toxic pollutants in their tissues, especially pesticides such as pyrethroids and heavy metals such as selenium and mercury (Israel and Klimley 2008; Stewart et al. 2004). Subadult and adult Green Sturgeon occupy a diversity of depths within bays and estuaries for feeding and migration. Tagged adults and subadults within the San Francisco Estuary and Delta occupy waters over shallow depths of less than 33 feet, either swimming near the surface or foraging along the bottom, although recent studies suggested adults tend to be benthically oriented in the San Francisco Estuary and Delta (Chapman et al. 2019). Juvenile Green Sturgeon are largely oriented at or near the bottom (Thomas et al. 2019).

Juvenile Green Sturgeon and White Sturgeon are periodically (although rarely) collected from the lower San Joaquin River at south Delta water diversion facilities and other sites (National Marine Fisheries Service 2009; Aasen 2011, 2012). Green Sturgeon are salvaged from the south Delta diversion facilities and are generally juveniles greater than 10 months but less than three years old (U.S. Bureau of Reclamation 2008:8-17). NMFS (2005) suggested that the high percentage of San Joaquin River flows contributing to the Tracy Fish Collection Facility could mean that some entrained Green Sturgeon originated in the San Joaquin River Basin.

Green Sturgeon larval distribution is estimated to extend at least 100 km (62 miles) downstream from spawning habitats on the Sacramento and Feather rivers in high-flow years. This estimated downstream distribution corresponds with the Colusa area on the Sacramento River (RM 157) and the confluence of the Sacramento and Feather rivers near Verona (RM 80) for larvae originating in the Sacramento River and Feather River, respectively (Heublein et al. 2017a:14). Juveniles are believed to use the Delta for rearing for the first one to three years of their lives before moving out to the ocean and are likely to be found in the main channels of the Delta and the larger interconnecting sloughs and waterways, especially within the central Delta and Suisun Bay and Suisun Marsh. Miller et al. (2020) found the greatest number of detections of acoustically tagged juvenile Green Sturgeon in the central Delta, with relatively few occurring in the Sacramento River mainstem and north Delta sloughs (Sutter, Steamboat, Miner). Project operations at the DCC have the potential to reroute Green Sturgeon as they outmigrate through the lower Sacramento River to the Delta (Israel and Klimley 2008; Vogel 2011). When the DCC is open, there is no passage delay for adults, but juveniles could be diverted from the Sacramento River into the interior Delta. This has been shown to reduce the survival of juvenile Chinook Salmon (Brandes and McLain 2001:69-70; Newman and Brandes 2010; Perry et al. 2012), but it is unknown whether it has similar effects on Green Sturgeon.

The Sacramento River provides habitat for Green Sturgeon spawning, adult holding, foraging, and juvenile rearing. Sturgeon spawn in deep pools (averaging about 28 feet deep) (National Marine Fisheries Service 2018). Suitable spawning temperatures and spawning substrate exist for Green Sturgeon in the Sacramento River upstream and downstream of RBDD (U.S. Bureau of Reclamation 2008:8-7–8-8). Although the historical upstream extent of Green Sturgeon spawning in the Sacramento River is unknown, the observed distribution of sturgeon eggs, larvae, and juveniles indicates that spawning occurs from Hamilton City to as far upstream as the Inks Creek confluence and possibly up to the Cow Creek confluence (Brown 2007; Poytress et al. 2013). Adult Green Sturgeon that migrate upstream in April, May, and June are completely blocked by the Anderson-Cottonwood Irrigation District diversion dam (National Marine Fisheries Service 2009), rendering approximately 3 miles (4.8 km) of spawning habitat upstream of the diversion dam inaccessible. Based on the distribution of sturgeon eggs, larvae, and juveniles in the Sacramento River, California Department of Fish and Game (2002), now known as CDFW, indicated that Green Sturgeon spawn in late spring and early summer, although they periodically spawn in late summer and fall (as late as October) (Heublein et al. 2009, 2017b; National Marine Fisheries Service 2018). Green Sturgeon eggs are believed generally to hatch about a week after fertilization (Heublein et al. 2017b). The number of Green Sturgeon accessing the upper Sacramento River appears to have increased following the decommissioning of RBDD (Steel et al. 2019a).

Green Sturgeon from the Sacramento River are genetically distinct from their northern counterparts, indicating a spawning fidelity to their natal rivers (Israel et al. 2004), even though individuals can range widely (Lindley et al. 2008). Larval Green Sturgeon have been regularly captured during their dispersal stage at about two weeks of age (24–34 mm [0.95–1.34 inches] FL) in rotary screw traps at RBDD (California Department of Fish and Game 2002:7) and at about three weeks old when captured at the Glenn-Colusa Irrigation District intake (Van Eenennaam et al. 2001). Juvenile Green Sturgeon can spend extended periods in the upper river reach before they begin their outmigration to the Delta when river discharge and turbidity increase (Poytress et al. 2024).

The current population status is unknown (Beamesderfer et al. 2007; Adams et al. 2007). A genetic analysis of Green Sturgeon juveniles captured in the Sacramento River resulted in an estimate of the annual number of spawners upstream of RBDD ranging from 10 to 28 individuals between 2002 and 2006 (Israel and May 2010b). Using results from acoustic telemetry and dual-frequency identification sonar studies to locate Green Sturgeon in the Sacramento River to derive an adult spawner abundance estimate of 2,106 fish (95 percent confidence interval = 1,246–2,966), Mora et al. (2018) applied a conceptual demographic structure to the adult population estimate and generated a subadult southern DPS Green Sturgeon population estimate of 11,055 (95 percent confidence interval = 6,540–15,571), together with an estimate of 4,387 juveniles (95 percent confidence interval = 2,595–6,179). The estimate does not include spawning adults in the lower

Feather or Yuba rivers (National Marine Fisheries Service 2019), with spawning confirmed in the Feather River (Seesholtz et al. 2015) and Yuba River (National Marine Fisheries Service 2018). Mora et al. (2018) cautioned that their juvenile and subadult Green Sturgeon estimates are less reliable than their adult estimates because the former were based on the ratios from a modeling study; the percentage of juvenile sturgeon is particularly uncertain because so little is known about this life stage. Additionally, the modeling study upon which the juvenile and subadult estimates are based requires four assumptions that Mora et al. (2018) admitted are rarely met: (1) constant recruitment, (2) population equilibrium, (3) stable size and age structure, and (4) a lack of density dependence. Mora et al. (2018) suggested, however, that their study provided a rough estimate of total abundance that is suitable for assessing the impacts of take, such as that observed in coastal trawl fisheries and at large water diversions.

NMFS (2009) noted that, similar to winter-run Chinook Salmon, the restriction of spawning habitat for Green Sturgeon to only one reach of the Sacramento River increases the vulnerability of this spawning population to catastrophic events, which is one of the primary reasons that the southern DPS of Green Sturgeon was federally listed as a threatened species in 2006. However, there is evidence that Green Sturgeon spawn in the Feather River, although perhaps irregularly (Seesholtz et al. 2015).

Southern DPS North American Green Sturgeon are thought to have historically spawned in the Sacramento, Feather, and San Joaquin rivers (Adams et al. 2007). Mora et al. (2009) estimated that large dams had blocked access to Keswick Dam blocks access to approx. 39 ± 14 km of habitat in the Pit, McCloud and Little Sacramento rivers, Nimbus Dam blocks access to approx. 22 ± 8 km of habitat in the American River, Oroville Dam blocks access to approx. 16 ± 4 km of habitat in the Feather River, Friant Dam blocks approx. 12 ± 4 km of habitat in the San Joaquin River and Daguerre Dam blocks approx. 4 ± 2 km of habitat in the Yuba River. After hatching, Green Sturgeon larvae possess limited swimming ability and generally seek refuge in low-velocity and complex habitats, such as large cobble substrate (Kynard et al. 2005). While little is known about Green Sturgeon rearing, it is likely that juveniles rear near spawning habitat for a few months or more before migrating to the Delta (Heublein et al. 2017b:15).

Green Sturgeon are also present in the San Joaquin River, but at considerably lower numbers than White Sturgeon. Between 2007 and 2012, anglers reported catching six Green Sturgeon in the San Joaquin River (Jackson and Van Eenennaam 2013). In 2017, environmental DNA testing confirmed the identity of a Green Sturgeon observed in the Stanislaus River at Knights Ferry (Anderson et al. 2018). Although the reported presence of Green Sturgeon in the San Joaquin River coincides with the spawning migration period of Green Sturgeon within the Sacramento River, no evidence of spawning has been detected (Jackson and Van Eenennaam 2013).

General life stage timing for southern DPS Green Sturgeon is summarized in Tables 6A-13 and 6A-14.

Table 6A-13. Temporal Occurrence of Southern Distinct Population Segment Green Sturgeon by Life Stage

Location	Month																							
	Jan		Feb		Mar		Apr		May		Jun		Jul		Aug		Sep		Oct		Nov		Dec	
Adult – sexually mature																								
Sacramento River (river mile 332.5-451)	L	L	L	L	M	M	H	H	H	H	H	H	H	H	H	H	H	H	H	H	H	M	M	M
Sacramento River (<river mile 332.5)	L	L	L	M	M	M	M	M	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L	L
Sacramento-San Joaquin-San Francisco Estuary	L	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	L	L
Larva																								
Sacramento River (<river mile 332.5)	N	N	N	N	N	L	M	M	H	H	H	H	H	H	M	M	M	M	L	L	N	N	N	N
Juvenile (≤5 months old)																								
Sacramento River (<river mile 332.5)	N	N	N	N	N	N	N	L	M	M	M	M	H	H	H	H	H	H	H	M	M	M	M	M
Juvenile (≤5 months old)																								
Sacramento River (<river mile 391)	M	M	M	M	L	L	L	L	L	L	L	M	M	M	M	H	H	H	H	H	H	H	H	L
Sub-adults and non-spawning adults																								
Sacramento-San Joaquin-San Francisco Estuary	M	M	M	M	M	M	M	M	M	M	H	H	H	H	H	H	H	H	H	H	H	H	M	M
Pacific Coast	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M	M
Coastal Bays and Estuaries	M	M	M	M	M	M	M	M	M	M	H	H	H	H	H	H	H	H	H	H	H	H	M	M

Source: National Marine Fisheries Service 2019:113–114.

Relative Abundance: H = high (blue); M = medium (green); L = low (yellow); N = none.

Table 6A-14. Temporal Occurrence of Southern Distinct Population Segment Green Sturgeon by Life Stage in the Delta

Life Stage	Month											
	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult	M	M	M	M	M	M	M	M	M	M	M	M
Juvenile	M	M	M	M	M	M	M	M	M	M	M	M
Salvaged	L	L	L	L	L	N	M	H	L	L	L	L

Source: National Marine Fisheries Service 2019:115.

Relative Abundance: H = high (blue); M = medium (green); L = low (yellow); N = none.

6A.1.8 White Sturgeon

White Sturgeon are generally similar to Green Sturgeon in terms of their biology and life history. Like Green Sturgeon and other sturgeon species, White Sturgeon are late-maturing and infrequent spawners, which makes them vulnerable to overexploitation and other sources of adult mortality. White Sturgeon are believed to be most abundant within the San Francisco Estuary and Delta region (Moyle 2002). Both nonspawning adults and juveniles can be found throughout the Delta year-round (Radtke 1966; Kohlhorst et al. 1991; Moyle 2002; California Department of Water Resources et al. 2013:11-535). When not undergoing spawning or ocean migrations, adults and subadults are usually most abundant in brackish portions of the San Francisco Estuary and Delta (Kohlhorst et al. 1991; Miller et al. 2020). White Sturgeon is not presently listed under the ESA or CESA but is a California Species of Special Concern (Moyle et al. 2015:102–117). However, a petition to list the White Sturgeon as threatened under CESA was submitted to the California Fish and Game Commission on November 29, 2023 (California Regulatory Notice Register 2024). Overall, information on trends in adults and juveniles suggests that numbers are declining (Moyle 2002; Moyle et al. 2015:4–5 of the White Sturgeon evaluation).

The Central Valley population of White Sturgeon spawns mainly in the Sacramento and Feather rivers, with occasional spawning in the San Joaquin River (Moyle 2002; Jackson 2013). Spawning-stage adults generally move into the lower reaches of rivers during winter prior to spawning and migrate upstream in response to higher flows to spawn from February to early June (McCabe and Tracy 1994; Schaffter 1997). Miller et al. (2020) detected the greatest number of adult White Sturgeon in their middle Sacramento River spawning reach during February–April, although individuals were present from October to June.

After absorbing yolk sacs and initiating feeding, YOY White Sturgeon make an active downstream migration that disperses them widely to rearing habitat throughout the lower rivers and the Delta (McCabe and Tracy 1994). White Sturgeon larvae have been observed to be flushed farther downstream in the Delta and Suisun Bay in high outflow years, but are restricted to more interior locations in low outflow years (Stevens and Miller 1970). White Sturgeon larvae are periodically collected in various locations throughout the Delta in general larval fish monitoring (e.g., 20-mm survey) in late winter and spring and in salvage at federal and state Delta pumping facilities, so that larval distribution generally is suggested to range from downriver of spawning habitats (primarily in the Sacramento and San Joaquin rivers) to the approximate downstream extent of the Delta at Chipps Island (Heublein et al. 2017a). From 1995 through 2021, 60% of all White Sturgeon were collected in the 20-mm survey in Wet water years (Table 6A-15). Only 2% of White Sturgeon collected during this time period were in Critical and Dry water years (Table 6A-15), consistent with previous observations from Stevens and Miller (1970).

Table 6A-15. Annual Number of White Sturgeon Collected in the 20-mm Survey, 1995–2021

Year	Water Year Type	Number of White Sturgeon Larvae Collected
1995	W	8
1996	W	5
1997	W	2
1998	W	81
1999	W	7
2000	AN	16
2001	D	1
2002	D	0
2003	AN	2
2004	BN	0
2005	AN	0
2006	W	16
2007	D	0
2008	C	2
2009	D	1
2010	BN	0
2011	W	8
2012	BN	0
2013	D	0
2014	C	1
2015	C	0
2016	BN	137
2017	W	73
2018	BN	6
2019	W	34
2020	D	0
2021	C	0

Source: Interagency Ecological Program et al. (2021).

Salinity tolerance increases with increasing age and size (McEnroe and Cech 1985), allowing White Sturgeon to access a broader range of habitat in the San Francisco Estuary (Israel et al. 2008). During Dry years, White Sturgeon have been observed following brackish waters farther upstream, while the opposite occurs in Wet years (Kohlhorst et al. 1991). Adult White Sturgeon tend to concentrate in deeper areas and tidal channels with soft bottoms, especially during low tides, and typically move into intertidal or shallow subtidal areas to feed during high tides (Moyle 2002). These shallow water habitats provide opportunities for feeding on benthic organisms, such as opossum shrimp (mysids), amphipods, and even invasive overbite clams (*Potamocorbula amurensis*), and small fishes (Israel et al. 2008; Kogut 2008). White Sturgeon also have been found in tidal habitats of medium-sized tributary streams to the San Francisco Estuary, such as Coyote Creek and Guadalupe River in the South Bay and Napa and Petaluma rivers and Sonoma Creek in the North Bay (Leidy 2007). Miller et al. (2020) described some detections of acoustically tagged adult White Sturgeon year-round in central and south San Francisco Bay, but many more individuals were detected year-round in San Pablo Bay, Suisun Bay, and the Delta. Subadults were mostly detected in San Pablo Bay, Suisun Bay, and the Delta (Miller et al. 2020). Patton et al. (2020) found that White Sturgeon 26.6–63 inches (675–1,600 mm) long were very rare in small wetland channels relative to large channels or shoals in the lower Delta.

Numerous factors likely affect the White Sturgeon population in the Delta, similar to those for Green Sturgeon. Survival during early life history stages may be adversely affected by insufficient flows, lack of rearing habitat, predation, warm water temperatures, decreased DO, chemical toxicants in the water, and entrainment at diversions (Cech et al. 1984; Israel et al. 2008). Historical habitats, including shallow intertidal feeding habitats, have been lost in the Delta because of channelization. Overexploitation by recreational fishing and poaching also likely has been an important factor adversely affecting numbers of adult sturgeon (Moyle 2002), although new regulations were implemented in 2007 by CDFW to reduce harvest. The relatively high current levels of exploitation (annually nearly 14 percent) will likely continue to decrease the population size in the future (albeit with considerable uncertainty); maintaining a stable population would likely require low levels of exploitation (<3 percent) (Blackburn et al. 2019). Like Green Sturgeon, there have historically been substantial passage problems for White Sturgeon such as at the Fremont Weir (Sommer et al. 2014). Upstream-migrating White Sturgeon enter the Yolo Bypass more often than Green Sturgeon (Miller et al. 2020), but have a high probability of exiting at the Bypass's southern end when passage to the Sacramento River at the northern end of the Yolo Bypass is not available (Johnston et al. 2020). Positive correlations exist between White Sturgeon year class strength indices and Delta outflow (Fish 2010; Gingras et al. 2013). Vessel strikes are a source of injury and mortality to White Sturgeon in the San Francisco Estuary and Delta, although the proportion of fish affected is not known (Hildebrand et al. 2016; Demetras et al. 2020). Recent research has shown tissue from adult White Sturgeon in the San Francisco Estuary to contain multiple metal and organic contaminants, including some (selenium, mercury, cadmium, arsenic, and copper) at levels known to impair fish health and likely negatively affecting fitness (Gundersen et al. 2017). Laboratory experiments have demonstrated that predators abruptly reduce foraging activity, possibly reducing growth rates and extending the period of juvenile vulnerability to predation, suggesting that introduced predators and degraded habitats may have interacting effects (Steel et al. 2019b).

Central Valley White Sturgeon are most abundant within the San Francisco Estuary and Delta, but the population spawns mainly in the Sacramento River (Moyle 2002). White Sturgeon larvae rear primarily in the Sacramento River and the Delta (Moyle 2002; Israel et al. 2008). White Sturgeon are found in the Sacramento River primarily downstream of RBDD (Tehama-Colusa Canal Authority 2008), with most spawning between Knights Landing and Colusa (Schaffter 1997).

The population status of White Sturgeon in the Sacramento River is unclear. Overall, limited information on trends in adult and juvenile abundance in the Delta population suggests that numbers are declining (Reis-Santos et al. 2008). Adults ready to spawn generally move into the lower reaches of the Sacramento River during winter, and then migrate upstream in response to higher flows and spawn from February to early June (Schaffter 1997; McCabe and Tracy 1994). Most spawning in the Sacramento River occurs in April and May between Knights Landing and Colusa (Kohlhorst 1976). As previously noted, recent acoustic telemetry studies found the greatest number of adult White Sturgeon in their middle Sacramento River spawning reach during February–April (Miller et al. 2020), confirming prior investigations. The acoustic telemetry studies also suggest that north Delta sloughs (Miner Slough, Steamboat Slough, and Sutter Slough) are used more than the mainstem for upstream migration, although the mainstem is the main downstream migration route following spawning (Miller et al. 2020). YOY White Sturgeon juveniles make an active downstream migration that disperses them widely to rearing habitat throughout the lower Sacramento River and Delta (McCabe and Tracy 1994; Israel et al. 2008). Statistical analysis has demonstrated a positive relationship between Delta outflow and recruitment of White Sturgeon year classes (Kohlhorst et al. 1991).

White Sturgeon are known to use the lower Feather River primarily for spawning, embryo development, and early rearing. Limited quantitative information is available on the status of White Sturgeon in the lower Feather River, but the spawning population was most likely much larger prior to construction of Oroville Dam in 1961 (Israel et al. 2008). Sixteen White Sturgeon were recorded from creel surveys and sightings during 2006, and more were captured by anglers in 2007 (Israel et al. 2008). Numerous factors likely limit the success of the White Sturgeon population in the lower Feather River, but loss of historical habitat, alteration of temperatures and flows caused by the Oroville Project, and recreational fishing and poaching are expected to be among the most important factors.

Small numbers of White Sturgeon inhabit the American River, as evidenced by White Sturgeon report cards submitted to CDFW by anglers in recent years (e.g., DuBois and Danos 2017, 2018). Very little other information about use of the American River by White Sturgeon is available.

Little is known about White Sturgeon populations inhabiting the San Joaquin River. Spawning-stage adults generally move into the lower reaches of rivers during winter prior to spawning, then migrate upstream to spawn in response to higher flows (Schaffter 1997; McCabe and Tracy 1994). Based on tag returns from White Sturgeon tagged in the Delta and recovered by anglers, Kohlhorst et al. (1991) estimated that over 10 times as many White Sturgeon spawn in the Sacramento River as in the San Joaquin River. CDFW fisheries catch information for the San Joaquin River obtained from fishery report cards (California Department of Fish and Game 2008, 2009b, 2010, 2011, 2012b; California Department of Fish and Wildlife 2013, 2014) documented that anglers upstream of State Route 140 annually caught between 8 and 25 mature White Sturgeon between 2007 and 2013. Below State Route 140 downstream to Stockton, anglers annually caught between 2 and 35 mature White Sturgeon over the same period. Most of the White Sturgeon caught were released.

White Sturgeon spawning in the San Joaquin River was documented for the first time in 2011 and confirmed in 2012. Viable White Sturgeon eggs were collected in 2011 at one sampling location downstream of Laird Park (Gruber et al. 2012) and in 2012 at four sampling locations generally between Laird Park and the Stanislaus River confluence (Jackson and Van Eenennaam 2013). Although the majority of sturgeon likely spawn in the Sacramento River, the results of these surveys confirm that White Sturgeon do spawn in the San Joaquin River in both Wet- and Dry-year conditions and may be an important source of production for the White Sturgeon population in the Sacramento–San Joaquin River system.

White Sturgeon are uncommon in the Klamath and Trinity rivers (National Research Council 2004). Although historically there may have been small spawning runs in these rivers, there are no recent reports of White Sturgeon spawning in this system. However, Welch et al. (2006) stated that a total of 186 juvenile and adult White Sturgeon were caught in the Klamath River native sturgeon fisheries between 1980 and 2002. The presence of juveniles suggests White Sturgeon spawning may occur in this system. Currently almost all sturgeon found in the Klamath River Basin above the estuary are Green Sturgeon (Moyle 2002).

6A.1.9 Pacific Lamprey

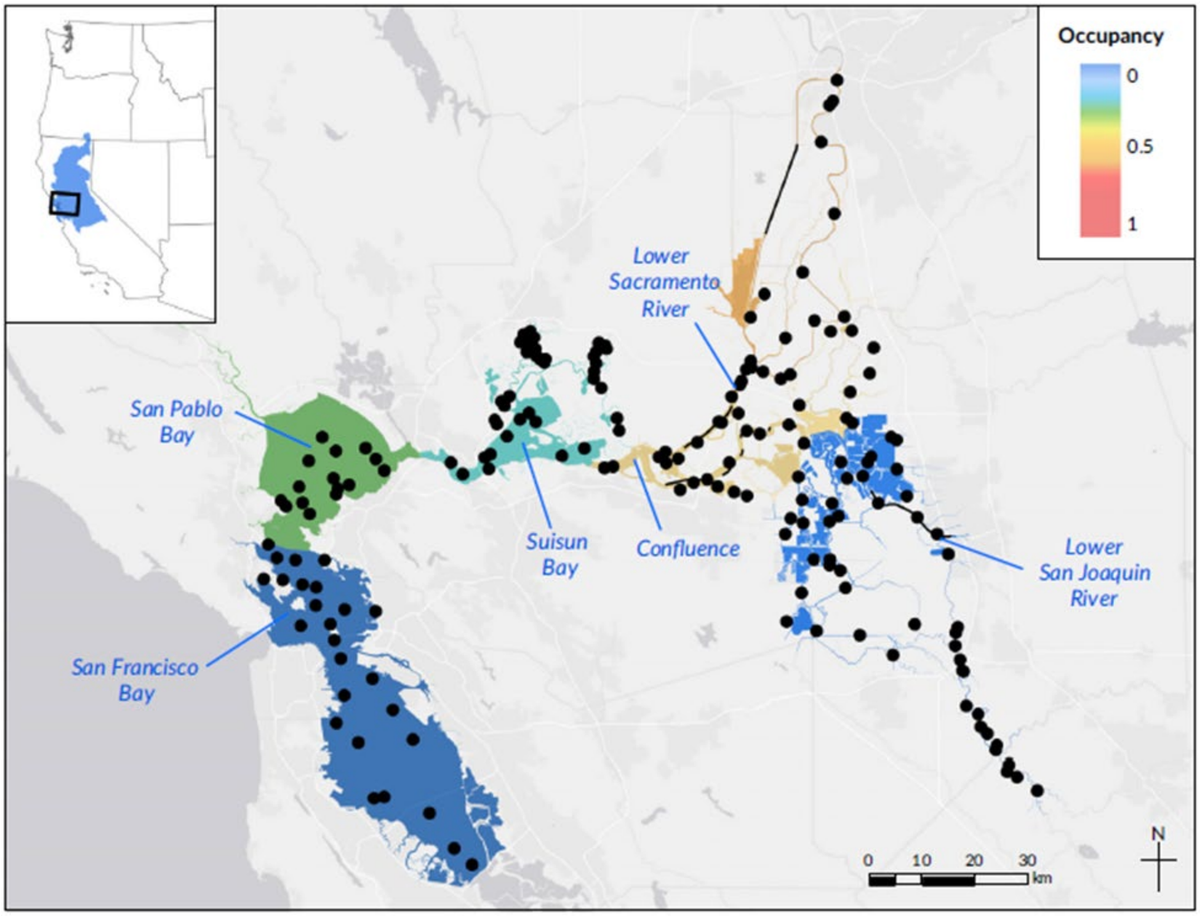
Pacific Lamprey is a widely distributed species that uses the Delta for upstream migration as adults, for downstream migration as juveniles, and for rearing as ammocoetes (the larval stage of lamprey) (Hanni et al. 2006; Moyle et al. 2009). Pacific Lamprey is present in the north, central, and south Delta, and ammocoetes are present year-round in all of the regions (California Department of Water Resources et al. 2013:11-592). Limited information on status of Pacific Lamprey in the Delta exists, but the number of lamprey inhabiting the Delta is likely greatly suppressed compared with historical levels, as suggested by the loss of access to historical habitat and apparent population declines throughout California and the Sacramento and San Joaquin River basins (Moyle et al. 2009). Pacific Lamprey is listed as a California Species of Special Concern (Moyle et al. 2015).

Limited data indicate most adult Pacific Lamprey migrate though the Delta en route to upstream holding and spawning grounds in the early spring through early summer (Hanni et al. 2006). As documented in other large river systems, it is likely that some adult migration through the Delta occurs from late fall and winter through summer and possibly over an even broader period (Robinson and Bayer 2005; Hanni et al. 2006; Moyle et al. 2009; Clemens et al. 2012; Lampman 2011). Data from the FMWT survey in the lower Sacramento and San Joaquin rivers and Suisun Bay suggest that peak outmigration of Pacific Lamprey through the Delta coincides with high-flow events from fall through spring (Hanni et al. 2006). Some outmigration likely occurs year-round, as observed at sites farther upstream (Hanni et al. 2006) and in other river systems (Moyle 2002). Some Pacific Lamprey ammocoetes likely spend part of their extended (five to seven years) freshwater residence rearing in the Delta, particularly in the upstream, freshwater portions.

Sacramento River Pacific Lamprey adults enter the Sacramento River from the Delta primarily during about March through June and hold in the river for about a year prior to spawning (Moyle et al. 2015). Spawning occurs in gravel redds in the upper river from March through July. The eggs and pro-larvae incubate for about 1 to 1.5 months. After the larvae (ammocoetes) emerge, they drift downstream and burrow into fine sediments primarily in off-channel habitats, where they rear (Schultz et al. 2014; Moyle et al. 2015). After five or more years, the ammocoetes metamorphose to the macropthalmia (juvenile) stage and migrate downstream to the Delta and ocean. Migration downstream is closely associated with rainfall events, with most migrants sampled in the upper Sacramento River being collected on the day of a rainfall event or the following two days (Goodman et al. 2015).

River flow potentially affects survival of Pacific Lamprey eggs and larvae, and the migratory habitat of the juveniles and adults. Pacific Lamprey build their spawning redds in shallow water (about 0.5 to 3.5 feet) (Gunckel et al. 2009; Schultz et al. 2014; Moyle et al. 2015), so reductions in water level can dewater the redds. The larvae select habitats, often off-channel, with fine sediments, low flow velocity, and shallow depths (approximately 1 foot), so they are vulnerable to stranding by reductions in water level.

In the San Francisco Estuary and Delta, occupancy of habitat by lamprey (including Pacific Lamprey and Western River Lamprey combined) was found to be greatest in the north and central Delta, with zero to low occupancy in the south Delta, San Joaquin River, and San Francisco Bay (Goertler et al. 2020; Figure 6A-17). Predicted occupancy of habitat declines with increasing temperature; for example, in the lower Sacramento River, occupancy probability ranges from nearly 1 at very low temperature (below 10 °C [50 °F]) to below 0.25 at temperature of around 25 °C (77 °F) (Goertler et al. 2020).



Source: Goertler et al. 2020.
Note: Data examined were for 2006–2016. Filled black circles are sampling sites.

Figure 6A-17. Mean Modeled Lamprey Occupancy Estimates Mapped by Region with Sites Used in the Single-Season Lamprey Occupancy Model

There are no data on Pacific Lamprey spawning specific to Clear Creek, but they are assumed to occur throughout the accessible portion of Clear Creek downstream of Whiskeytown Dam. Pacific Lamprey are inferred to spawn and rear in Clear Creek because ammocoetes have been routinely collected in the screw trap at RM 1.7. Lamprey life cycles in Clear Creek are assumed to follow those elsewhere in California. Pacific Lamprey inhabit accessible reaches of the lower Feather River (California Department of Water Resources 2003). Little information is available on factors limiting Pacific Lamprey populations in the lower Feather River, but they are likely affected by many of the same factors as salmon and steelhead because of parallels in their life cycles. Hannon and Deason (2008) have documented Pacific Lamprey spawning in the nearby American River between early January and late May, with peak spawning typically occurring in early April. Pacific Lamprey ammocoetes rear in the lower Feather River or American River for all or part of their five- to seven-year freshwater residence.

Limited information on Pacific Lamprey status in the Stanislaus River exists, but the species has experienced loss of access to historical habitat and apparent population declines throughout California and the Sacramento and San Joaquin River basins (Moyle et al. 2009). Pacific Lamprey ammocoetes are expected to rear in the Stanislaus River for all or part of their five- to seven-year freshwater residence. Data from rotary screw trapping in the nearby Mokelumne and Tuolumne rivers suggest that outmigration of Pacific Lamprey generally occurs from early winter through early summer (Hanni et al. 2006). Catches of juvenile Pacific Lampreys in trawl surveys of the mainstem San Joaquin River, near the mouth of the Stanislaus River at Mossdale, occurred during winter and spring. Significant numbers of lampreys of unknown species and unspecified life stage have been captured during rotary screw trapping on the Stanislaus River at Oakdale (FISHBIO Environmental 2007:38) and Caswell (Watry et al. 2007).

Pacific Lamprey are an anadromous species that is important to local tribes and supports a subsistence fishery on the lower Trinity River. Adult Pacific Lamprey may begin their upstream migration during all months of the year, but peak upstream migration typically occurs from December through June (Larson and Belchik 1998; Petersen Lewis 2009). After entering fresh water, Pacific Lamprey hold through summer and most of the winter before reaching sexual maturity. Pacific Lamprey undergo a secondary migration in the late winter or early spring from holding areas to spawning grounds; spawning occurs during the spring (Robinson and Bayer 2005; Clemens et al. 2012; Lampman 2011). Therefore, adult Pacific Lamprey can be found in the Trinity River throughout the year. Ammocoetes rear within fine substrates in depositional areas and remain in the Trinity River and tributaries for up to seven years before outmigrating to the ocean (Moyle 2002:98; U.S. Bureau of Reclamation and Trinity County 2006:3.6-20). Limited data are available on the distribution and abundance of Pacific Lamprey in the Trinity River. They are expected to have a distribution similar to anadromous salmonids that use the mainstem Trinity River and accessible reaches of larger tributaries. Pacific Lamprey abundance in the Trinity River is believed to be declining based on information from Tribal fishermen who catch lamprey in the lower Klamath River (Petersen Lewis 2009). Parallels in the life cycle of Pacific Lamprey make them susceptible to many of the same factors as salmon and steelhead. Reduced access to historical spawning and rearing habitats in the Trinity River above Lewiston Dam, degraded spawning and rearing habitat resulting from operations of dams and water diversions, impacts from historic mining practices, and predation by nonnative invasive species (e.g., brown trout) have likely contributed to adverse effects on the Trinity River Pacific Lamprey population.

6A.1.10 Western River Lamprey

River Lamprey are found in large coastal streams from just north of Juneau, Alaska, to the San Francisco Bay (Vladykov and Follett 1958; Wydoski and Whitney 1979). The Sacramento and San Joaquin River basins are at the southern edge of their range (Moyle et al. 2009). River Lamprey seem to be primarily associated with the lower portions of certain large river systems, and most records for California are from the lower Sacramento–San Joaquin system, especially the Stanislaus and Tuolumne rivers (Moyle et al. 1995; Moyle 2002). In the Sacramento River, they have been documented upstream to RBDD (Hanni et al. 2006; Moyle et al. 2009). River Lamprey have also been collected in the Feather and American rivers and Mill and Cache creeks (Vladykov and Follett 1958; Hanni et al. 2006; Moyle et al. 2009). Quantitative data on populations are extremely limited, but loss and degradation of historical habitats suggest populations may have declined. The River Lamprey is considered a California Species of Special Concern (Moyle et al. 2015).

River Lamprey life history is poorly known, especially in California (Moyle et al. 2015). The adults migrate from the ocean to spawning areas during the fall and late winter (Beamish 1980). Spawning is believed to occur February through May in small tributary streams (Moyle 2002). The redds are built at the upstream end of small riffles (Moyle 2002). After the larvae (ammocoetes) emerge, they drift downstream and burrow into sediments in pools or side channels where they rear. After several years, the larvae metamorphose in late July and the juveniles (macropthalmia) migrate downstream in the following year from May to July (Moyle 2002).

River flow potentially affects survival of River Lamprey eggs and larvae, and migratory habitat of the juveniles and adults. River Lamprey build their spawning redds in shallow water (Moyle et al. 2015), so reductions in water level can dewater the redds. Assuming River Lamprey larvae habitat requirements are similar to those of Pacific Lamprey, the larvae select habitats that are often off-channel, with low flow velocity and shallow depths; therefore, they are vulnerable to stranding by reductions in water level.

River Lamprey have been collected in the Feather River, but there is little information about their use of the Feather River (Vladykov and Follett 1958; Hanni et al. 2006; Moyle et al. 2009). Spawning is generally in spring, at least in other locations (Beamish 1980), with adult-sized River Lamprey collected in the Feather River from mid-November to early May (Hanni et al. 2006). There are no monitoring programs that target River Lamprey. Quantitative data on populations are limited, but loss and degradation of historical habitats suggest populations may have declined. River Lamprey are inconspicuous, often overlooked, and ammocoetes can be difficult to distinguish from ammocoetes of the co-occurring Pacific Lamprey. Hanni et al. (2006) summarized distribution data and did not include specific information for the American River or Stanislaus River. It is possible that the species occurs in these areas based on available habitat. River Lamprey have been collected in the San Joaquin River at Mossdale, suggesting spawning somewhere in the San Joaquin River Basin.

6A.1.11 Sacramento Hitch

Sacramento Hitch is a California Species of Special Concern (Moyle et al. 2015). The species historically occurred in low-elevation streams throughout the Sacramento and San Joaquin valleys and in the Delta, but are now extirpated from the San Joaquin River and tributaries between Friant Dam and the Merced River (Moyle et al. 2015). Within the San Francisco Estuary and Delta vicinity, there are historical records (from before the water development of the 1950s) for Sacramento Hitch in Coyote Creek and Alameda Creek. However, the species may have been introduced to Arroyo Valle through water transfers from the Central Valley. Furthermore, it is unknown if populations in these streams are reproducing or are sustained from reservoir or historical stream populations (Leidy 2007). The species occurs in some urban streams and may tolerate highly altered habitats. Sacramento Hitch face continued threats from population fragmentation (e.g., dams), agriculture (e.g., flow alteration and pollution), estuary alteration, and nonnative species. The species appears to be in long-term decline and mainly consists of scattered, small populations over a broad area. However, there is only moderate concern for overall species extinction in part due to its fairly secure establishment in some areas (Moyle et al. 2015).

Moyle et al. (2015:2) summarized habitat and distribution: "Spawning takes place mainly in riffles of streams tributary to lakes, river, and sloughs after flows increase in response to spring rains, but spawning requirements are in need of further documentation." Moyle et al. (2015:2) indicate that YOY hitch spend the two months after hatching shoaling in shallow water or staying close to aquatic plants before moving out into more open water at around 50 mm FL. The species is distributed in the Sacramento River Basin but no longer in the San Joaquin River Basin and in some locations in the north Delta (Moyle et al. 2015:2–3).

6A.1.12 Sacramento Splittail

Sacramento Splittail are found primarily in marshes, turbid sloughs, and slow-moving river reaches throughout the Delta subregion (Sommer et al. 1997, 2008). Sacramento Splittail are most abundant in moderately shallow, brackish tidal sloughs and adjacent open-water areas, but they also can be found in freshwater areas with tidal or riverine flow (Moyle et al. 2004).

Adult Sacramento Splittail typically migrate upstream from brackish areas in January and February and spawn in fresh water, particularly on inundated floodplains when they are available, in March and April (Sommer et al. 1997, 2008; Moyle et al. 2004). A substantial amount of splittail spawning occurs in the Yolo and Sutter Bypasses and the Cosumnes River area of the Delta (Moyle et al. 2004). Spawning also can occur in the San Joaquin River during high-flow events (Sommer et al. 1997, 2008). However, not all adults migrate significant distances to spawn, as evidenced by spawning in the Napa and Petaluma rivers (Feyrer et al. 2005). Sommer et al. (2002) found that larval and juvenile splittail of 15–20-mm FL in a floodplain were concentrated in edge habitat near an inflow during the day but moved at night into deeper habitats, including open water and habitats with submerged vegetation, whereas larger splittail (28–34-mm FL) used a broad range of habitats both during the day and at night.

Although juvenile Sacramento Splittail are known to rear in upstream areas for a year or more (Baxter 1999), most move to the Delta after only a few weeks or months of rearing in floodplain habitats along the rivers (Feyrer et al. 2006). Juveniles move downstream into the Delta from April to August (Meng and Moyle 1995; Feyrer et al. 2005). Juvenile size at outmigration downstream from the Yolo Bypass was found to generally be around 30–40-mm FL (Feyrer et al. 2006) and 25–40-mm total length (TL) from the Cosumnes River floodplain (Moyle et al. 2004). Sacramento Splittail recruitment is largely limited by extent and period of inundation of floodplain spawning habitats, with abundance observed to spike following Wet years and dip after Dry years (Moyle et al. 2004). However, the five- to seven-year life span buffers the adult population abundance (Sommer et al. 1997; Moyle et al. 2004). Other factors that may adversely affect the splittail population in the Delta include entrainment, predation, changed estuarine hydraulics, nonnative species (Moyle et al. 2004), pollutants (Greenfield et al. 2008), and limited food.

Historically, Sacramento Splittail were widespread in the Sacramento River from Redding to the Delta (Rutter 1908, as cited in Moyle et al. 2004). This distribution has become somewhat reduced in recent years (Sommer et al. 1997, 2007b). During drier years there is evidence that spawning occurs farther upstream (Feyrer et al. 2005). Adult splittail migrate upstream in the lower Sacramento River to above the mouth of the Feather River and into the Sutter and Yolo Bypasses (Sommer et al. 1997; Feyrer et al. 2005; Sommer et al. 2007b). Each year, mainly during the spring spawning season, a small number of individuals have been documented at the Red Bluff Pumping Plant and the entrance to the Glenn-Colusa Irrigation District intake (Moyle et al. 2004).

As previously noted, nonreproductive adult splittail are most abundant in moderately shallow, brackish areas in the Delta and Suisun Bay, but can also be found in freshwater areas with tidal or riverine flow (Moyle et al. 2004). Adults typically migrate upstream from brackish areas in January and February and spawn in fresh water on inundated floodplains in March and April (Moyle et al. 2004; Sommer et al. 2007b). In the Sacramento drainage, the most important spawning areas appear to be the Yolo and Sutter Bypasses, in years that they are inundated. However, some spawning occurs almost every year along inundated river edges and backwaters created by small increases in flow. Splittail spawn in the Sacramento River from Colusa to Knights Landing in most years (Feyrer et al. 2005).

Most juvenile Sacramento Splittail move from upstream areas downstream into the Delta from April through August (Meng and Moyle 1995; Sommer et al. 2007b). The production of YOY Sacramento Splittail is largely influenced by extent and duration of inundation of floodplain spawning habitats, with abundance spiking following Wet years and declining after Dry years (Sommer et al. 1997; Moyle et al. 2004; Feyrer et al. 2006). Other factors that may affect the Sacramento Splittail adult population include pumping at the CVP and SWP south Delta export facilities, flood control operations and infrastructure, entrainment by irrigation diversions, recreational fishing, changed estuarine hydraulics, pollutants, and nonnative species (Moyle et al. 2004; Sommer et al. 2007b).

Sacramento Splittail enter the lower Feather River primarily in Wet years. On the lower Feather River, February through May is believed to encompass the period of splittail spawning, egg incubation, and initial rearing (Sommer et al. 2008; California Department of Water Resources 2004:4-3). Splittail use shallow flooded vegetation for spawning. Most spawning in the Feather River is thought to occur downstream of the Yuba River confluence (Federal Energy Regulatory Commission 2007:125). The primary factor that likely limits the lower Feather River splittail population is availability of spawning and rearing habitats as related to inundation of floodplains (Moyle et al. 2004; California Department of Water Resources 2004:6-1).

Splittail likely spawn in the lower reaches of the American River, particularly in wetter conditions (Moyle et al. 2004). Mature individuals begin a gradual upstream migration toward spawning areas between late November and late January (Moyle et al. 2004). Spawning typically occurs between late February and early July (Wang 1986). Although juvenile splittail can rear in upstream areas for a year or more (Baxter 1999), most move downstream after only a few weeks of rearing (Feyrer et al. 2006). Most juveniles move downstream into the Delta from April to August (Meng and Moyle 1995).

In Wet years Sacramento Splittail have been found in the San Joaquin River as far upstream as Salt Slough (Baxter 1999, 2000; Brown and Moyle 1993; Saiki 1984). Historically, Sacramento Splittail were widespread in the San Joaquin River and found upstream to Tulare and Buena Vista lakes, where they were harvested by native peoples (Moyle et al. 2004). Spawning typically takes place on inundated floodplains from February through June, with peak spawning in March and April. Today, Sacramento Splittail likely ascend the San Joaquin River to Salt Slough during Wet years (Baxter 1999). During Dry years, Sacramento Splittail are uncommon in the San Joaquin River and occur only downstream of the Tuolumne River (Moyle et al. 2004). Most spawning takes place in the flood bypasses, along the lower reaches of the Sacramento and San Joaquin rivers and major tributaries, and lower Cosumnes River and similar areas in the western Delta.

6A.1.13 Hardhead

Hardhead (*Mylopharodon conocephalus*) is a California Species of Special Concern (Moyle et al. 2015). They exist throughout the Sacramento–San Joaquin River Basin and are fairly common in the Sacramento River and the lower reaches of the American and Feather rivers. In other parts of their range, populations have declined or have become increasingly isolated (Moyle 2002). Hardhead also inhabit reservoirs and are abundant in a few impoundments where water level fluctuations prevent black bass from reproducing in large numbers (Moyle 2002). Hardhead tend to be absent from areas that have been highly altered (Moyle et al. 1995) or that are dominated by introduced fish species, especially centrarchids (species of the black bass and sunfish) (Moyle et al. 1995). Hardhead are omnivorous, their diet consisting mostly of benthic invertebrates and aquatic plants, but also including drifting insects. In reservoirs, Hardhead also prey on zooplankton (Moyle et al. 1995).

Hardhead spawn mainly in April and May, but some may spawn as late as August in the foothill regions of the upper San Joaquin River (Wang 1986). They migrate upstream and into tributary streams as far as 45 miles (72.4 km) to spawning sites. Spawning behavior has not been documented, but it is assumed to be similar to that of the Sacramento Pikeminnow (*Ptychocheilus grandis*), which deposit their eggs over gravel-bottomed riffles, runs, and at the head of pools (Moyle et al. 1995). Spawning substrates may also include sand and decomposed granite (Wang 1986).

Hardhead larvae and juveniles likely inhabit stream margins with abundant cover and move into deeper habitats as they grow larger. Adults occupy the deepest part of pools. Juvenile and adult Hardhead are present in the Sacramento River year-round. They tend to prefer water temperatures near 67 °F (19.4 °C) (Thompson et al. 2012), but have been captured at RBDD, where water temperatures are generally much cooler (Tucker et al. 1998).

Hardhead are present in very low abundance in the San Francisco Estuary and Delta, as reflected by electrofishing in the 1980s and 2000s (Brown and Michniuk 2007) and very few individuals being collected at the SWP/CVP south Delta fish salvage facilities (California Department of Water Resources 2020:4-62).

In 2004 and 2005, Hardhead were found at only 2 of 31 sampling locations upstream of Whiskeytown Dam (both in the mainstem Clear Creek) (Wulff et al. 2012). Hardhead are also found in Whiskeytown Reservoir (National Park Service 1999).

Hardhead are fairly common year-round in the lower reaches of the American River (Moyle 2002). Although migratory behavior of Hardhead in the American River individuals is unknown, individuals from other large rivers, such as the Sacramento River, migrate into tributary streams during April and May. Hardhead typically spawn between April and May (Moyle 2002). Although Hardhead early life history is largely unknown, young individuals likely remain along stream edges with dense cover and move into deeper water as they grow, which allows migrants to move back downstream in the current (Moyle 2002).

In the San Joaquin drainage, Hardhead are present throughout tributary streams, but are largely absent from the mainstem San Joaquin River as a result of periodic desiccation during the dry season. Hardhead are widely distributed in foothill streams and may be found in a few reservoirs such as Redinger and Kerkhoff reservoirs upstream of Millerton Lake on the San Joaquin River.

6A.1.14 Central California Roach

Central California Roach is a small (usually less than 10 cm TL), stout-bodied minnow that occurs in tributaries to the Sacramento and San Joaquin rivers and tributaries to San Francisco Bay. Their historic distribution in the upper Sacramento River Basin is poorly understood, but their upstream range limit is thought to have been Pit River Falls (Moyle et al. 2015).

Central California Roach are found in small, high-gradient, often intermittent tributaries but appear to be poorly adapted to lakes and reservoirs. Where dams have been constructed on Central Valley streams, Central California roach persist only in small tributaries to the resultant reservoirs (Moyle et al. 2015). Their absence from reservoirs is likely due both to habitat alteration and to the presence of introduced predatory fish species. Central California Roach is a California Species of Special Concern (Moyle et al. 2015). They primarily inhabit small streams, but may occur in backwaters with dense riparian cover along the mainstem rivers (Baumsteiger and Moyle 2019). Central California Roach frequent a wide variety of habitats, which are often isolated by downstream barriers. They are adaptable fish and tolerate relatively high water temperatures and low oxygen levels (Moyle et al. 2016). They spawn from March through early July, usually when water temperatures exceed about 61 °F (16 °C) (Moyle 2002). Hatching takes place in two to three days, and fry remain in crevices until they can actively swim. Roach are omnivores, eating such items as terrestrial insects, filamentous algae, aquatic insect larvae and adults, crustaceans, and detritus.

Central California Roach are present in very low abundance in the San Francisco Estuary and Delta, as reflected by electrofishing in the 1980s and 2000s (Brown and Michniuk 2007) and none being collected at the SWP/CVP south Delta fish salvage facilities (California Department of Water Resources 2020:4-62).

In 2004 and 2005, Central California Roach were found in 3 of 11 surveyed streams upstream of Whiskeytown Dam. They were confined to Clear Creek and its tributary Cline Gulch upstream of the Carr Powerhouse and Grizzly Gulch upstream of Whiskeytown Reservoir. They were also found in Paige-Boulder Creek, a tributary of Clear Creek downstream of Whiskeytown Dam. Where found, Central California Roach may be locally abundant, second in abundance only to Riffle Sculpin and, in some locations, Sacramento Sucker (Wulff et al. 2012).

6A.1.15 Starry Flounder

Starry Flounder is a species for which essential fish habitat (EFH for Pacific Groundfish) exists in the San Francisco Estuary and Delta. The overall extent of Pacific Groundfish EFH includes all water and substrate in depths that are less than or equal to 11,483 feet (3,500 meters or 1,914 fathoms) to the mean higher high water level or the upriver extent of saltwater intrusion (upstream area and landward where waters have salinities less than 0.5 ppt), known spawning habitat and thermal refugia, complex channels and floodplains and areas containing estuarine and marine submerged aquatic vegetation.

Starry Flounder is a flatfish that belongs to the family *Pleuronectidae* (Moyle 2002). Starry Flounder range from north of the Bering Strait south to Los Angeles Harbor. Older juveniles and adults are found from 120 km (74.5 miles) upstream to the outer continental shelf at 375-meter (1,230-foot) depth, but most adults are found at less than 150-meter (492-foot) depth. Most juvenile fish are found in shallow, fresh to brackish water, and shift to salinities of 10–15 ppt as they mature, but appear to remain within estuaries through at least their second year (Baxter et al. 1999; Moyle

2002). During the late fall and winter, mature Starry Flounder probably migrate to shallow coastal waters to spawn (Orcutt 1950). Adults primarily inhabit coastal marine waters (Orcutt 1950; Haertel and Osterberg 1967; Bottom et al. 1984:89; Hieb and Baxter 1993). Distribution of age-0 juveniles within the San Francisco Estuary and Delta is primarily in Suisun Bay and San Pablo Bay, with lower abundance in the west Delta (Baxter et al. 1999:410). Starry Flounder older than age 1 (age-1+ fish) occur principally in San Pablo Bay, Suisun Bay, and Central Bay (Baxter et al. 1999:411–412).

In general, abundance indices from the past decade suggest a decline relative to several decades ago, consistent with declines in commercial and recreational catch (ICF International 2016a:5.E-12 and 5.E-13). Starry Flounder are found on different substrates including: gravel, clean shifting sand, hard stable sand, and mud substrata, but fishermen report the largest catches over soft sand. Prey from mud (sternapsid worms) and sand (*Siliqua patula* clams) habitats have been observed in the stomach of a single individual, suggesting fish move freely from one habitat type to another (Orcutt 1950). Starry Flounder also consume crabs, shrimps, worms, clams and clam siphons, other small mollusks, small fishes, nemertean worms, and brittle stars (Hart 1973). Starry Flounder can tolerate a wide range of salinities. In the Sacramento and San Joaquin rivers, Starry Flounder have been observed in salinities of 0.02–0.06 ppt (i.e., essentially fresh water) (Orcutt 1950), and have been collected 75 miles (120.7 km) upstream in the Columbia River. Age-0 and age-1+ Starry Flounder are a common species in estuarine habitats along the West Coast (see Orcutt 1950; Pearson 1989; Emmett et al. 1991:266; Baxter et al. 1999; Kimmerer 2002). Spawning occurs primarily during the winter months of December and January (Orcutt 1950). Starry Flounder reach approximately 110 mm (4.34 inches) in length by the end of their first year. By the time they reach age 2 many fish have migrated to ocean habitats adjacent to their natal estuaries. Starry Flounder become reproductively mature at age 2 for males and age 3 for females, which equates to approximately 28 cm (11.02 inches) in males and approximately 35 cm (13.78 inches) in females. Adults may move seasonally into shallow coastal waters to spawn, perhaps in proximity to estuaries to take advantage of estuarine circulation which would advect fertilized eggs near the bottom into nursery areas.

Hieb and Baxter (1993) established specific habitat criteria for Starry Flounder YOY (<70 mm) in the San Francisco Estuary: 90 percent were collected from intertidal and subtidal habitats <7 meters (23 feet) in depth, and with accompanying salinities of <22 ppt. The exclusivity of fresh and brackish water rearing habitat in age-0 and age-1 Starry Flounder coupled with the relationship between freshwater outflow and abundance makes a strong case for estuarine dependence (Emmett et al. 1991:267; Hieb and Baxter 1993). However, spawning in coastal areas and variation in abundance during high outflow years suggest that coastal ocean conditions as well as high outflow work in conjunction to determine year class abundance (Hieb and Baxter 1993). There is a significant correlation between Delta outflow (X2) and indices of Starry Flounder abundance in the San Francisco Estuary and Delta, although the mechanism underlying the correlation does not appear to be related to extent of habitat and may be related to enhanced transport to estuarine rearing grounds by increased residual circulation with increased outflow (Kimmerer et al. 2009). It is unknown the extent to which this potential enhanced transport and apparent greater abundance in the San Francisco Estuary and Delta with greater outflow may contribute to overall coastwide Starry Flounder abundance (Grimaldo 2018:13–14).

6A.1.16 Northern Anchovy

Northern Anchovy (*Engraulis mordax*) is a species for which EFH exists in the San Francisco Estuary and Delta, as one of managed Coastal Pelagics. The overall extent of Coastal Pelagic EFH is based on a thermal range bordered by the geographic area where Coastal Pelagic Species occur at any life stage, where Coastal Pelagic Species have occurred historically during periods of similar environmental conditions, or where environmental conditions do not preclude colonization by Coastal Pelagic Species. Species diversity and abundance declines on an upstream gradient as determined by the tolerance of individual species for low and variable salinity conditions.

Northern Anchovy is distributed along the West Coast from British Columbia to Baja California (Miller and Lea 1972:56). The central subpopulation, which is present in the Proposed Project area, ranges from approximately San Francisco, California, to Punta Baja, Baja California. Members of the central population move north during the summer and south during the winter (Haugen et al. 1969). Northern Anchovy is an important forage fish for other resident and migratory species in the San Francisco Estuary and Delta, including salmon, Jacksmelt (*Atherinopsis californiensis*), and Striped Bass (Baxter et al. 1999:167). It supports a moderate commercial fishery for live bait (Smith and Kato 1979). The annual abundance of Northern Anchovy is highly variable between years. Surveys have shown that the greatest densities occur in Central, San Pablo, and South Bays, and only in late summer were they collected in appreciable numbers in Suisun Bay (Baxter et al. 1999).

Northern Anchovy is a small fish typically found in schools near the surface of the water. They are short-lived, rarely living past four years of age. A portion of the population reaches maturity at the end of their first year, about 50 percent by the end of their second year, and all are mature by their third or fourth year (Clark and Phillips 1952). Female anchovy are batch spawners, spawning 20 to 30 thousand eggs a year in two or three events (Baxter 1967). Spawning can occur during every month of the year and is temperature dependent, increasing in late winter and early spring and peaking from February to April. They spawn in nearshore areas across their entire range, in the upper 50 meters (164 feet) of the water column. Both Northern Anchovy eggs and larvae are found near the surface, and eggs need two to four days to hatch, depending on water temperatures. The San Francisco Bay is a very productive nursery area because of high abundance of food for both larvae and adults, advective losses are lower than in adjacent coastal waters, and the bay is warmer, with varying salinity allowing for eggs and larvae throughout the year (U.S. Bureau of Reclamation 2008:16-3). Anchovies feed diurnally either by filter feeding or biting, depending on the size of the food. Juvenile and adult anchovy feed at a higher trophic level than larvae, selectively feeding on larger zooplankton (mysids), fish eggs, and fish larvae and have been observed to eat small fish at times, even their own species (Baxter 1967).

Larvae eat phytoplankton and dinoflagellates, while larger larvae pick up copepods and other zooplankton. Larger female anchovies can consume up to 4–5 percent of their total body weight per day. Competitors with the anchovy for food include sardines and other schooling planktivores, such as Jacksmelt and Topsmelt. These species are also potential predators of young anchovy life stages (Goals Project 2000).

Factors affecting anchovy production are mostly natural influences, such as ocean temperature (Bergen and Jacobson 2001: 305). Offshore within the California Current, temperature, upwelling, and stable stratification of the water column are believed to work together to produce conditions that are favorable to anchovy larvae (Lasker 1975). Investigation of the correlations between Delta outflow (X2) and indices of abundance and habitat did not find statistically significant relationships

(Kimmerer et al. 2009). The distribution of Northern Anchovy shifted toward higher salinity when *Potamocorbula* invaded in the mid- to late 1980s, reducing summer abundance by >90 percent in the low-salinity region of the San Francisco Estuary and Delta (Kimmerer 2006).

6A.1.17 Striped Bass

Striped Bass is a recreationally important anadromous species introduced into the Sacramento and San Joaquin River basins between 1879 and 1882 (Moyle 2002). Despite their nonnative status and piscivorous feeding habits, Striped Bass are considered important because they are a major game fish in the Delta. Striped Bass use the Delta as a migratory route and for rearing and seasonal foraging. Striped Bass spend the majority of their lives in saltwater, returning to fresh water to spawn. When not migrating for spawning, adult Striped Bass in the San Francisco Estuary and Delta are found in San Pablo Bay, San Francisco Bay, and the Pacific Ocean (Moyle 2002). Adult Striped Bass spend about six to nine months of the year in San Francisco and San Pablo bays (Hassler 1988). Striped Bass also use deeper areas of many of the larger channels in the Delta, in addition to large embayments such as Suisun Bay.

Spawning occurs in spring, primarily in the Sacramento River between Sacramento and Colusa and in the San Joaquin River between Antioch and Venice Island (Farley 1966). Eggs are free-floating and negatively buoyant and hatch as they drift downstream, with larvae occurring in shallow and open waters of the lower reaches of the Sacramento and San Joaquin rivers, the Delta, Suisun Bay, Montezuma Slough, and Carquinez Strait. According to Hassler (1988), the distribution of larvae in the estuary depends on river flow. In low-flow years, all Striped Bass eggs and larvae are found in the Delta, while in high-flow years, the majority of eggs and larvae are transported downstream into Suisun Bay.

YOY Striped Bass distribute themselves in accordance with the estuarine salinity gradient (Kimmerer 2002; Feyrer et al. 2007), indicating that salinity is a major factor affecting their habitat use and geographic distributions. Kimmerer (2002) found that distributions of fish species, including Striped Bass, substantially overlapped with the low-salinity zone. Older Striped Bass are increasingly flexible about their distribution relative to salinity (Moyle 2002). Statistically significant correlations between indices of age-0 abundance or survival in the San Francisco Estuary and Delta have been found with Delta outflow (X2) (e.g., Kimmerer et al. 2009; Mac Nally et al. 2010), with evidence for greater extent of habitat as the mechanism (Kimmerer et al. 2009). However, subsequent density-dependent survival after the first summer dampens the effects of flow on subsequent recruitment (Kimmerer et al. 2000).

The entrainment of Striped Bass has been observed at the south Delta export facilities, including Clifton Court Forebay (Stevens et al. 1985; Bowen et al. 1998:7; Aasen 2012). In WY 2011, salvage of Striped Bass at export facilities (approximately 550,000 fish) continued a generally low trend observed since the mid-1990s. Prior to 1995, annual Striped Bass salvage was generally above 1 million fish (Aasen 2012). DWR et al. (2013) reported that Striped Bass longer than 24 mm (0.95 inch) were effectively screened at the Tracy Fish Collection Facility (TFCF) and bypassed the pumps. However, planktonic eggs, larvae, and juveniles smaller than 24 mm (0.94 inch) in length received no protection from entrainment. Although the percentage entrainment of YOY juveniles during June through September at the south Delta export facilities was estimated to be appreciable (median of 33 percent, maximum of 99 percent), any population-level effect may have been obscured by variability in total mortality and possibly salvage operations success, or the density-dependent effect during and after this life stage (Kimmerer et al. 2000, 2001).

Striped Bass, primarily YOY, are one of the pelagic fish of the upper estuary that have shown substantial variability in their populations, with evidence of long-term declines (Kimmerer et al. 2000; Sommer et al. 2007a). A substantial proportion of the variability in abundance index patterns has been associated with variation of outflow in the estuary (Jassby et al. 1995; Kimmerer et al. 2001; Loboschefsky et al. 2012), although this is disputed by some interested parties (Bourez 2011). However, surveys showed that population levels for YOY Striped Bass began to decline sharply around 1987 and 2002 (Thomson et al. 2010), despite relatively moderate hydrology, which typically supports at least modest fish production (Sommer et al. 2007a). Moyle (2002) cites causes of decline in Striped Bass to include climatic factors, entrainment at south Delta export facilities in the south Delta, other diversions, pollutants, reduced estuarine productivity, invasions by alien species, and human exploitation. Kimmerer et al. (2000, 2001) attribute the decline in juvenile YOY Striped Bass to declining carrying capacity, likely related to food limitation. Loboschefsky et al. (2012) showed that there had been no long-term decline for age-1 and older Striped Bass as of 2004.

Striped Bass occur in the lower Feather River and have been reported to occur in the Thermalito Forebay (Federal Energy Regulatory Commission 2007:125). Striped Bass are a popular sport fish in the lower Feather River during periods when they migrate upstream to spawn. Little is known about Striped Bass use of the Feather River, although acoustically tagged Striped Bass released in the Feather River spent more time in the river than acoustically tagged Striped Bass released in the Sacramento River (Sabal et al. 2019). These acoustically tagged Striped Bass generally seemed to follow seasonal prey sources, with the most time spent in the San Francisco and San Pablo bays during the summer, and greatest detections in Delta in the winter, with an overall variability to behavior that may have allowed persistence since the original introduction of the species (Sabal et al. 2019).

Striped Bass are year-round inhabitants of the American River from the confluence with the Sacramento River to Nimbus Dam, with highest densities during summer (Surface Water Resources 2001; Moyle 2002). Although specific spawning locations in the American River are not well understood, the river is believed to serve as a nursery area for YOY and subadult Striped Bass (Surface Water Resources 2001). They provide a locally important sportfishing resource.

Striped Bass occur in the Stanislaus River, and they support a sport fishery when adult fish migrate upstream to spawn. Striped Bass have been observed at Lovers Leap and at Knights Ferry from May through the end of June. These adult fish were observed in all habitats (U.S. Fish and Wildlife Service 2002; Kennedy and Cannon 2005). The distribution of Striped Bass in the Stanislaus River is thought to be limited to downstream of the historic Knights Ferry Bridge because of a set of falls about 3 feet tall in the area (U.S. Fish and Wildlife Service 2002). Ainsley et al. (2013) reported that Striped Bass were collected in May at two locations on the mainstem San Joaquin River between the Head of Old River and the mouth of the Stanislaus River.

6A.1.18 American Shad

American Shad is a recreationally important anadromous species introduced into the Sacramento and San Joaquin River basins in the 1870s (Moyle 2002). American Shad spend most of their adult life at sea and may make extensive migrations along the coast. American Shad become sexually mature while in the ocean and migrate through the Delta to spawning areas in the Sacramento, Feather, American, and Yuba rivers. Some spawning also takes place in the lower San Joaquin, Mokelumne, and Stanislaus rivers (U.S. Fish and Wildlife Service 1995). The spawning migration may begin as early as February, but most adults migrate into the Delta in March and early April (Skinner 1962). Migrating adults generally take two to three months to pass through the Delta (Painter et al. 1979). They enter the Feather River annually in spring to spawn and are present in the lower Feather River from May through mid-December during the adult immigration, spawning, and outmigration periods of their life cycle (California Department of Water Resources 2003). Adult American Shad migrate into the lower American River to spawn during the late spring, typically during April through early July (California Department of Fish and Game 1986). American Shad migrate up the Stanislaus River to spawn in the late spring and support a sport fishery during that period. American Shad have been observed on occasion from June through July at Lovers Leap (U.S. Fish and Wildlife Service 2002; Kennedy and Cannon 2005). American Shad were found primarily in the faster habitats and were observed in schools of 20 or more (U.S. Fish and Wildlife Service 2002). Little is known about American Shad in the San Joaquin River. They may spawn in the San Joaquin River system, but their abundance is unknown. Sport fishing for American Shad occurs seasonally in the San Joaquin River. A unique, successfully reproducing landlocked population of American Shad exists in Millerton Lake.

Water temperature is an important factor influencing the timing of spawning. American Shad are reported to spawn at water temperatures ranging from approximately 46 to 79 °F (7.8 to 26 °C) (U.S. Fish and Wildlife Service 1967), although optimal spawning temperatures are reported to range from about 60 to 70 °F (15.6 to 21 °C) (Bell 1986:95; Leggett and Whitney 1972; Painter et al. 1979). Spawning takes place mostly in the main channels of rivers, and generally about 70 percent of the spawning run is made up of first-time spawners (Moyle 2002).

Shad have remarkable abilities to navigate and to detect minor changes in their environment (Leggett 1973). Although homing is generally assumed in the Sacramento River and its tributaries, there is some evidence that numbers of first-time spawning fish are proportional to flows of each river at the time the shad arrive. When suitable spawning conditions are found, American Shad school and broadcast their eggs throughout the water column. The optimal temperature for egg development is reported to occur at 62 °F (16.7 °C). At this temperature, eggs hatch in five to eight days; at temperatures near 75 °F (23.9 °C), eggs would hatch in three days (MacKenzie et al. 1985). Egg incubation and hatching, therefore, are coincident with the spawning period.

Fertilized eggs are slightly negatively buoyant, are not adhesive, and drift in the current. Newly hatched larvae are found downstream of spawning areas and can be rapidly transported downstream by river currents because of their small size. Juvenile American Shad rear in the Sacramento River from Colusa to Sacramento, the lower Feather River below the Yuba River, and the Sacramento River portion of the Delta (Stevens et al. 1987). As previously noted, rearing also takes place in the Mokelumne River near the DCC. Based on density, juvenile rearing in the American and Yuba rivers appears less than other areas (Stevens et al. 1987). Overall, in contrast to Striped Bass, an appreciable portion of the American Shad population appears to rear upstream of the Delta based on density in seine catches (Stevens et al. 1987). Some juvenile shad may rear in the Delta for up to a year before outmigrating to the ocean (U.S. Fish and Wildlife Service 1995). Outmigration from the Delta begins in late June and continues through November (Painter et al. 1979).

Juvenile American Shad are frequently encountered in the Delta during the FMWT survey and in fish salvage monitoring at the south Delta SWP and CVP fish facilities. American Shad use of the Delta has been observed to vary with salinity (e.g., X2) and outflows (Kimmerer 2002). Statistically significant negative correlations exist between X2 and indices of abundance in the San Francisco Estuary and Delta, with the mechanism potentially being related to the extent of available habitat (Kimmerer et al. 2009).

American Shad are entrained at the TFCF (Bowen et al. 1998) and in the Clifton Court Forebay, mostly during May through December when young American Shad migrate downstream. The American Shad population in the Sacramento and San Joaquin River basins has declined since the late 1970s, most likely because of increased diversion of water from rivers and the Delta, combined with changing ocean conditions, and possibly pesticides (Moyle 2002). Salvage of American Shad at project export facilities in WY 2011 represented nearly 659,000 fish (Aasen 2012), with similar but slightly lower salvage in 2010 (545,125 fish) (Aasen 2011).

6A.1.19 Threadfin Shad

Threadfin Shad were intentionally introduced to provide forage for game fish. Threadfin Shad were planted by CDFW in reservoirs throughout California, with the Sacramento and San Joaquin River basins planted in 1959. From these transplants, they have become established in the Sacramento–San Joaquin River system and the Delta. Threadfin Shad live mainly in fresh water and become progressively less abundant as salinity increases. Juveniles form dense schools and, in estuaries, are found in water of all salinities, although they are most abundant in fresh water. Threadfin Shad are fast-growing but short-lived. Few live longer than two years. Spawning takes place in California in April through August, peaking in June and July when water temperatures exceed 68 °F (20 °C), although spawning has been observed at 14 to 18 °C (57 to 64 °F). The embryos hatch in three to six days and larvae immediately assume a planktonic existence.

As noted by Baxter et al. (2010:75), Threadfin Shad is widely distributed but is most commonly encountered and most abundant in the southeastern Delta, especially the San Joaquin River near and just downstream of Stockton, where suitable abiotic habitat coincides with high prey abundance (Feyrer et al. 2009); these regions also have a relatively high density of submerged aquatic vegetation, which provides important spawning and larval rearing habitat (Grimaldo et al. 2004). Baxter et al. (2010) also noted that historic surveys by Turner (1966) found relatively high abundance in the northeast Delta in dead-end sloughs.

Threadfin Shad are susceptible to entrainment in water diversions and the species is salvaged at the SWP and CVP south Delta fish salvage facilities in higher abundance than any other fish species. Herbold et al. (2005) estimated annual salvage from approximately 1.5 million to about 10 million during 1994–2005. Some evidence for correlations with abundance indices has been found for water clarity, indices of predator and prey abundance, and south Delta exports (Mac Nally et al. 2010; Thomson et al. 2010).

6A.1.20 Black Bass

6A.1.20.1 Largemouth Bass

Largemouth Bass is a nonnative species to California. They were introduced in California in 1891 (Dill and Cordone 1997:171; Moyle 2002) and have since been introduced to suitable waters, including streams and reservoirs, throughout the state. They are an important invasive species in the Delta.

Largemouth Bass first spawn during their second or third spring at about 7 inches in length. Spawning is limited to fresh water (Moyle 2002). The males begin building nests when the water temperature reaches about 60 °F (15.6 °C). Spawning takes place from April through June, at temperatures up to 75 °F (23.9 °C). Nests are shallow pits in depths of 3 to 7 feet and are often built next to submerged objects. Females lay their eggs in one or more nests. The eggs hatch in two to five days and sac fry usually spend five to eight days in or near the nest (Moyle 1976:315). For the first month or two after hatching, fry feed mainly on zooplankton. YOY bass stay close to shore in schools that swim in the open water. By the time they reach about 2 inches in length, the juveniles feed largely on aquatic insects and fish fry, including other Largemouth Bass, and after they reach 4 to 5 inches, they prey primarily on fish and crayfish. Largemouth Bass are thought to be a major predator of juvenile Chinook Salmon and other native fish species in the Delta (Nobriga and Feyrer 2007). In addition to the Delta, they occur in lower riverine habitats such as in the Feather River and American River. Their growth rate is highly variable, depending on genetic background, food availability, inter- and intra-specific competition, temperature regimes, and other environmental factors. Maximum size for the species is approximately 30 inches TL and the maximum age is 16 years (Moyle 1976:314).

Largemouth Bass prefer warm, quiet waters with aquatic vegetation and low turbidity. They are known to tolerate DO levels as low as 1 milligram of oxygen per liter (mg/L) (Lee et al. 1980; Moyle 2002). The species thrives in areas with high levels of infestation by nonnative aquatic plants (Brown and Michniuk 2007). Recent studies suggest juvenile and larger Largemouth Bass abundance is positively correlated with water temperature, whereas juvenile Largemouth Bass abundance is greatest at intermediate levels of submerged aquatic vegetation but larger fish are widespread even in areas with limited submerged aquatic vegetation (Conrad et al. 2016). A study in the San Joaquin River between the Head of Old River and Stockton in 2015–2016 found a mean of 333 Largemouth Bass per km, which were estimated to consume three to five fall-run Chinook Salmon per day per km during the peak of the salmon outmigration period, compared to 0 to 24 salmon consumed per day per km by Striped Bass (Michel et al. 2018).

6A.1.20.2 Smallmouth Bass

Smallmouth Bass (*Micropterus dolomieu*) is a nonnative species to California, which may have been first introduced in California in the 1870s (Dill and Cordone 1997:164). Smallmouth Bass are most common in large, clear lakes and cool, clear streams with large amounts of cover. In streams they prefer complex habitat with a variety of pools, riffles, runs, rocky bottoms, and overhanging trees, while lake populations concentrate in narrow bays along shores where rocky shelves project under water.

Optimal water temperature differs with age, as adults tend to stay in areas 25 to 27 °C (77 to 80.6 °F), while younger fish prefer areas 29 to 31 °C (84.2 to 87.8 °F), reflecting their shallower water environment. Regardless of age, however, temperatures greater than 35 °C (95 °F) are metabolically stressful, and temperatures over 38 °C (100.4 °F) are lethal. Smallmouth Bass are also restricted in their habitat choice by the amount of DO in the water. Although they can survive in areas with 1 to 3 mg/L oxygen, they require at least 6 mg/L for normal growth rates (University of California, Agriculture and Natural Resources 2019a). Juveniles and populations in crowded lakes may school, but this is rare and the majority are solitary hunters that stalk around some kinds of submerged debris. This localizes populations to such a degree that several reproductively independent groups can exist within a single lake. Foraging occurs throughout the day but is most intense in the evening and the early morning. Crustaceans and aquatic insects make up the majority of a Smallmouth Bass's diet until it reaches 3 to 5 cm (1.18–1.97 inches) TL, at which point crayfish and fish become more important. By the time an individual reaches 10 to 15 cm (3.94–5.91 inches) TL, these larger food items dominate the diet. Smallmouth Bass are opportunistic, however, and insects, amphibians, and small mammals are not uncommon sources of food (University of California, Agriculture and Natural Resources 2019a).

Smallmouth Bass reach maturity in their third or fourth spring, at which point they move into shallower water. Spawning begins in May and can continue into June or July. Males construct nests 30 to 60 cm (11.81–23.62 inches) in diameter, preferably in rubble, gravel, or sand 1 meter (3 feet) deep with submerged logs, boulders, and other submerged objects acting as cover. This is only the optimal environment, however, and nests can be found on a variety of substrates varying in depth from 0.5 to 5 meters (1.64–16.4 feet). These nests may be built close together, but they are not colonial and males defend the nests against other males as vigorously as they would against predators. Spawning is initiated by a female repeatedly swimming by a nest, changing colors, and keeping her head down in a mating posture. Eventually the pair circle the nest, with the male nipping at the female and the female occasionally rubbing her abdomen on the nest floor. The pair then settle into the nest and release their eggs and milt simultaneously. Smallmouth Bass are mostly monogamous, but the larger fish spawn earlier in the season and may have the opportunity to spawn again. Each female may release 2,000 to 21,000 eggs into her nest. The males guard the embryos and fan water over them to provide more oxygen. After hatching, it takes one to two weeks before fry become free swimming, and the male still guards them for another one to four weeks after that until they are too active to be herded. At 2 to 3 cm TL (0.79–1.18 inches), the young disperse to shallow water where high mortality rates are suffered because of predation and high stream flows. Those that survive generally grow to between 6 cm (2.36 inches) and 18 cm (7.09 inches) in their first year, and between 25 cm (9.84 inches) and 41 cm (16.14 inches) in their fourth, while stream populations grow at a decidedly slower rate. The largest individual on record weighed 4.1 kilograms (9.04 pounds) (University of California, Agriculture and Natural Resources 2019a). All life stages of Smallmouth Bass can occur in the freshwater regions of the Delta and lower portions of Central Valley rivers (Moyle 2002; Brown and May 2006; Brown and Michniuk 2007; Seesholtz et al. 2004).

6A.1.20.3 Spotted Bass

Spotted Bass are most common in moderately sized, clear, low-gradient rivers and reservoirs. In streams they spend most of their time hiding in pools, avoiding riffles or backwaters with heavy plant growth. Reservoir populations stay along steep rocky banks toward the upstream end of the reservoir. During the summer they can be found in temperatures between 24 °C (75.2 °F) and 31 °C (87.8 °F), and despite a low tolerance for brackish water, they have been found in salinities up to 10 ppt. Juveniles can easily be seen schooling in shallow areas close to shore, but adults are more solitary and spend most of their time 1–4 meters (3–12 feet) deep or even farther down when temperatures equalize in winter. Like most fish, the Spotted Bass's diet expands as a fish gets older. Fry eat mostly zooplankton and small insects, and then move on to crustaceans and larger aquatic insects as juveniles. Individuals between 75 and 150 mm (2.95–5.9 inches) feed on aquatic insects, fish, crayfish, and terrestrial insects, eventually preferring crayfish (University of California, Agriculture and Natural Resources 2019b).

Maturity is reached in the second or third year and spawning occurs when temperatures reach 15 to 18 °C (59 to 64.4 °F), continuing until temperatures reach 22 to 2 3°C (71.6 to 73.4 °F) in early June. Males move to shallow water in March and early April, where they construct nests 40 to 80 cm (31.5 inches) in diameter. Lake nests are built in areas 0.5 to 4.5 meters (1.64–14.76 feet) deep with large rocks and rubble or gravel, while nearly any area with low current can be used in rivers. These nests may be built close together, but Spotted Bass are not colonial and males defend the nests as vigorously against other males as they would against predators. Spawning is initiated by a female repeatedly swimming by a male's nest, changing colors, and keeping her head down in a mating posture. Eventually the pair circles the nest, with the male nipping at the female and the female occasionally rubbing her abdomen on the nest floor. The pair then settle into the nest and release their eggs and milt simultaneously. Spotted Bass are mostly monogamous, but some males may have more than one nest. Each female lays 2,000 to 14,000 eggs per nest. The male tends to and defends the nest for up to four weeks until the fry disperse at 30 mm TL (1.18 inches). Growth varies with habitat. Warmwater reservoirs support the highest growth, and cold streams support the slowest. On average, however, individuals reach 65–170 mm TL (2.56–6.69 inches) in their first year and 245–435 mm TL (9.65–17.13 inches) in their fourth. Few live longer than four or five years, and the largest recorded individual for California was 450 mm TL (17.72 inches) (University of California, Agriculture and Natural Resources 2019b). All life stages of Spotted Bass can occur in the freshwater regions of the Delta.

6A.1.21 California Bay Shrimp

A summary of California Bay Shrimp was provided by Baxter et al. (1999:78–79), upon which this account is largely based. Bay shrimp include several species of *Crangon*, primarily *C. franciscorum*. They are fished commercially by trawlers in the San Francisco Estuary west of a line joining Port Edith to the south extending through Buoy 6 to the shoreline on the north and sold as bait to sport anglers. From 1980 to 1995, the fishery annually landed between 100,000 and 200,000 pounds, although landings were considerably greater (2–3 million pounds) in the 1920s and 1930s, when bay shrimp were sold for human consumption. The fishery was concentrated in South Bay in the late 1980s to early 1990s probably because of lower-salinity water from sewage treatment plant discharges. There appears to have been a general decline in landings in the past two decades, with only one year above 100,000 pounds and most years since 2005 having 60,000 pounds or less in landings (ICF International 2016b:4–295).

Bay shrimp migrate seasonally in response to salinity, temperature, and maturity or life stage: for example, *C. franciscorum* larvae hatch in winter/early spring in Central Bay or the Gulf of the Farallones, with post-larvae and juveniles migrating upstream to rear in lower-salinity, warmer areas such as San Pablo and Suisun bays during the summer, before migrating downstream in fall/winter to complete the life cycle. Diet is variable by location and size and may consist of mysid shrimps, amphipods, bivalves, and copepods, for example. Bay shrimp are preyed upon by many fish in the estuary, including Striped Bass, Pacific Staghorn Sculpin (*Leptocottus armatus*), and Green and White Sturgeon, as well as other taxa such as harbor seals and diving ducks. The overall distribution is broad: for example, the dominant species *C. franciscorum* ranges from southeast Alaska to San Diego, California.

As with some other species in the San Francisco Estuary and Delta, statistically significant negative correlations between abundance index and X2 have been found, which does not appear to be related to extent of habitat and may be related to enhanced transport to estuarine rearing grounds by increased residual circulation with increased outflow (Kimmerer et al. 2009).

6A.1.22 Southern Resident Killer Whale

Southern resident killer whales are found primarily in the coastal waters offshore of British Columbia and Washington and Oregon in summer and fall (National Marine Fisheries Service 2008). During winter, southern resident killer whales are sometimes found off the coast of central California and more frequently off the Washington coast (Hilborn et al. 2012).

The 2005 listing (70 FR 69903) of southern resident killer whale DPS as endangered lists several factors that may be limiting the recovery of killer whales, including the quantity and quality of prey, accumulation of toxic contaminants, and sound and vessel disturbance. The Recovery Plan for Southern Resident Killer Whales (National Marine Fisheries Service 2008) posits that reduced prey availability forces whales to spend more time foraging, which may lead to reduced reproductive rates and higher mortality rates. Reduced food availability may lead to mobilization of fat stores, which can release stored contaminants and adversely affect reproduction or immune function (National Marine Fisheries Service 2008).

The Independent Science Panel reported that southern resident killer whales depend on Chinook Salmon as a critical food resource (Hilborn et al. 2012). Hanson et al. (2010) analyzed tissues from predation events and feces to confirm that Chinook Salmon were the most frequent prey item for the southern resident killer whale in two regions of the whale’s summer range off the coast of British Columbia and Washington State, representing more than 90 percent of the diet in July and August. Samples indicated that when southern resident killer whales are in inland waters from May through September, they consume Chinook Salmon stocks that originate from regions that include the Fraser River, Puget Sound, the Central British Columbia Coast, West and East Vancouver Island, and California’s Central Valley (Hanson et al. 2010).

Significant changes in food availability for southern resident killer whale have occurred over the past 150 years, largely due to human impacts on prey species. Salmon abundance has been reduced over the entire range of southern resident killer whale, from British Columbia to California. NMFS (2008) indicates that wild salmon have declined primarily due to degraded aquatic ecosystems, overharvesting, and production of fish in hatcheries. NMFS (2008) supports restoration efforts, 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.

Central Valley streams produce Chinook Salmon that contribute to the diet of southern resident killer whale. The number of Central Valley Chinook Salmon that annually enter the ocean and survive to a size susceptible to predation by southern resident killer whale is not known. A 2018 report evaluated 30 stocks of West Coast Chinook Salmon for recovery priority to increase southern resident killer whales' prey base, based on each stock's contribution to diet, degree of spatiotemporal overlap, and whether it would be consumed during times of killer whale reduced body condition or diversified diet. Central Valley stocks ranked 13 (spring-run Chinook Salmon), 16 (fall- and late-fall-run Chinook Salmon), and 21 (winter-run Chinook Salmon) (National Marine Fisheries Service and Washington Department of Fish and Wildlife 2018:7–8). Genetic sampling of Chinook Salmon prey of southern resident killer whale during October to May 2004–2017 found that Central Valley fall-run Chinook Salmon made up a mean of 4.8 percent of stocks contributing to the diet for whales in Puget Sound during fall/early winter, whereas in outer coast waters during mid-winter/early spring, Central Valley fall-run contributed a mean of 8.0 percent and Central Valley spring-run contributed a mean of 11.0 percent of the Chinook Salmon diet of the whales (Hanson et al. 2021).

6A.2 References Cited

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6A.2.2 Personal Communications

None cited.