Chapter 2
Covered Species

This chapter contains species accounts for all listed species addressed in this document. The species habitat models described herein were developed in coordination with the *Delta Conveyance Project Final Environmental Impact Report* (Final EIR) (California Department of Water Resources 2023).

California Department of Fish and Wildlife (CDFW) incidental take permit (ITP) regulations (14 California Code of Regulations [Cal. Code Regs.] §783.2(a)(2)) require identification of the common and scientific names of the species to be covered by the permit, including the species' status under the California Endangered Species Act (CESA). The species for which California Department of Water Resources (DWR) seeks a CESA ITP are shown in Table 2.0-1. These species have the potential to occur in the area that could be affected by the proposed project and could be subject to incidental take. A description of each of the covered species is provided below, including known population trends and known threats to the species. Non-covered CESA-listed plant species Boggs Lake hedge-hyssop (*Gratiola heterosepala*), Colusa grass (*Neostapfia colusana*), and Solano grass (*Tuctoria mucronata*) do not occur in the project area and do not have potential to be affected by the proposed project. These species are included in the analysis of take to disclose potential impacts from tidal restoration mitigation actions described in the Compensatory Mitigation Plan (CMP; Appendix 5A, *Compensatory Mitigation Plan for Special-Status Species and Aquatic Resources*). Tidal restoration mitigation actions would be covered by separate, site-specific ITP applications (as described in Appendix 5A).

Non-covered CESA-listed wildlife species least that do not have potential to be affected by the proposed project include Bell's vireo (*Vireo bellii pusillus*), western yellow-billed cuckoo (*Coccyzus americanus occidentalis*), western bumble bee (*Bombus occidentalis*), and San Joaquin kit fox (*Vulpes macrotis mutica*). DWR is not seeking CESA take coverage because take, as defined by CESA, will not occur. The rationale for the "no take" conclusion is listed below for each of these species.

- Least Bell's vireo—The species is not currently known to occur in the project area. There are no California Natural Diversity Database (CNDDB) records of least Bell's vireos breeding in the project area since at least the 1970s and there are no extant occurrences that overlap with the project footprint (California Department of Fish and Wildlife 2020b, 2020c). Sporadic occurrences of individuals displaying breeding behavior in the Delta (California Department of Fish and Wildlife 2020c; eBird 2021) suggests the species is attempting to recolonize the Central Valley. For example, Bell's vireo was reported at Bradford Island in 2023 and at San Joaquin River National Wildlife Refuge in 2016 (eBird 2024a). These occurrences are considered rare events and, as of the writing of this application, least Bell's vireo is not known to occur within the project area and therefore not at risk of take.
- Westen yellow-billed cuckoo—Current nesting populations do not occur in the project area and there are no known western yellow-billed cuckoo breeding pairs in the project area (California Department of Fish and Wildlife 2020a). Most riparian corridors in the project area do not support sufficiently large riparian patches or the natural, geomorphic processes that provide suitable breeding habitat (Greco 2013:711–715). Several remnant riparian patches in the vicinity of Mandeville and Medford Islands provide riparian vegetation suitable for western yellow-billed cuckoo but do not provide a sufficiently large patch size to support breeding

cuckoos. Historic and recent sightings of western yellow-billed cuckoo near the project area are presumed to be migrating birds that are capable of easily avoiding construction areas and therefore any potential for take.

- Western bumble bee—The species is not expected to occur within the project area of the proposed project. As shown on the species' current and historic species range map (California Department of Fish and Wildlife 2023b), the project area is outside of the current range for the species. The western bumble bee range is predicted to continue to decline across the Mediterranean California ecoregion (Janousek and Graves 2022; Janousek et al. 2023) and the species is not expected to recolonize the Central Valley.
- San Joaquin kit fox—The species is not expected to occur within the project area of the proposed project and the project is not expected to directly affect the species. Potentially suitable habitat within the project area is considered to be in a "very low" condition which is defined by the USFWS as showing "no evidence of a current population" (U.S. Fish and Wildlife Service 2020c:50). Based on CNDDB records, the species has not been detected in the region since 2000.

Table 2.0-1. Species to be Covered by 2081(b) Incidental Take Permit

Common Name	Scientific Name	State Status
Delta smelt	*Hypomesus transpacificus*	Endangered
Longfin smelt	*Spirinchus thaleichthys*	Threatened
Winter-run Chinook salmon	*Oncorhynchus tshawytscha*	Endangered
Spring-run Chinook salmon	*Oncorhynchus tshawytscha*	Threatened
White sturgeon	*Acipenser transmontanus*	Species of Special Concern, proposed listing as Threatened
California tiger salamander	*Ambystoma californiense*	Threatened
Giant garter snake	*Thamnophis gigas*	Threatened
Swainson's hawk	*Buteo swainsoni*	Threatened
Tricolored blackbird	*Agelaius tricolor*	Threatened
Crotch bumble bee	*Bombus crotchii*	Candidate endangered
Mason's lilaeopsis	*Lilaeopsis masonii*	Rare [a]

[a] Take of Mason's lilaeopsis is being requested through a rare plant permit, not through a 2081(b) permit. Information on this species is provided in this document to support the rare plant permit application.

GIS-based habitat models have been used to estimate impacts to California tiger salamander, giant garter snake, Swainson's hawk, tricolored blackbird, Crotch bumble bee, and Mason's lilaeopsis. These habitat models are described in the appropriate subsections, below. Impacts on fish species were based on the assumption of potential presence throughout fish-bearing waters in the project area, subject to timing constraints and habitat use as described for each species in the appropriate subsections that follow. Impacts on fish species also considered relevant studies, surveys, and species impact models.

2.1 Delta Smelt

2.1.1 Legal Status

Delta smelt (*Hypomesus transpacificus*) was listed as a threatened species under the CESA and ESA in 1993 (58 *Federal Register* [FR] 12854) and USFWS designated critical habitat for the species in December 1994 (59 FR 65256). 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 the CESA (Bay Institute et al. 2007). On March 4, 2009, the California Fish and Game Commission elevated the status of delta smelt to endangered under the CESA (California Department of Fish and Wildlife 2021). A 12-month finding on a petition to reclassify the delta smelt as an endangered species 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 the 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 of reclassifying delta smelt to endangered from threatened 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 FR 73175). As of May 03, 2022, USFWS has declared that, although the species may warrant reclassification to endangered, because of its current protections as a threatened species, it will not be reclassified (87 FR 26172).

2.1.2 Life History and General Ecology

Delta smelt are endemic to the San Francisco Estuary and Sacramento–San Joaquin Delta (Delta) where the species primarily occupies open-water habitats in Suisun Bay, Suisun Marsh, and the Delta. 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 1 year, which typically occurs from April to the following April. In captivity, delta smelt can survive to spawn at 2 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 either within the low-salinity zone of the Upper San Francisco Estuary, the tidal freshwater region of the Cache Slough Complex, or move between the two regions of fresh water and low salinity (Bennett 2005; Sommer and Mejia 2013; Hobbs et al. 2019; Merz et al. 2011).[1] Komoroske et al. (2016) found that 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 (92% of fish caught 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), or the discrepancy between field observations and laboratory observations may be evidence that delta smelt's distribution in the wild is due to a factor or factors other than salinity per se (U.S. Fish and Wildlife Service 2019a:122).

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[1] The low-salinity zone is frequently defined as waters with a salinity range of about 0.5 to 6 parts per thousand (Kimmerer 2004). As discussed further below, delta smelt are not solely restricted to the low salinity zone.

Although delta smelt are physiologically euryhaline (i.e., are able to tolerate 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 substrate (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*), as well as experimental trials, suggest that sand or small pebbles may be the preferred substrate (Bennett 2005; Lindberg et al. 2020). Hatching success peaks at water temperatures of 15 degrees Celsius (°C) to 16°C, ceasing when water temperatures exceed 20°C (60 degrees Fahrenheit [°F]) (Bennett 2005). Water temperatures suitable for spawning occur most frequently during the months of March to May, but ripe female delta smelt have been observed as early as January and larvae have been collected as late as July (Damon et al. 2016). Most spawning occurs at 9°C to 18°C (48.2°F to 64.4°F), 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 and during Spring Kodiak Trawl sampling from 2003 to 2015 with 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 suggested that spawning locations change based on hydrological conditions (reviewed by Bennett 2005:13). However, a more recent study indicated that the majority of regional movement 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, as well as 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 of 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 is conducted by the Enhanced Delta Smelt Monitoring program throughout the fall and winter, including December when maturity is generally reached (Kurobe et al. 2016). Many delta smelt disperse to landward[2] habitats sometime after the first significant precipitation event of the winter, staging while sexual maturity is completed (Grimaldo et al. 2009; Sommer et al. 2011; Polansky et al. 2018). Some adult delta smelt exhibit very limited dispersal during the spawning season (Murphy and Hamilton 2013; Polansky et al. 2018).

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[2] Note that "landward" in this context does not necessarily mean "upstream, " as there could be lateral movements (Murphy and Hamilton 2013).

In the wild, larval delta smelt are presumed to be surface-oriented, exhibiting greater dispersion during the night (Bennett 2005). Juvenile delta smelt 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). Adult delta smelt appear to use tidal migration and/or move horizontally toward shore during spawning migrations to upstream habitats (Bennett and Burau 2015). Laboratory studies of delta smelt of 32–68 millimeters (mm) standard length (SL) gave mean critical swimming velocity of around 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 first on juvenile and then adult stages of the calanoid copepods *Eurytemora affnis* and *Pseudodiaptomus forbesi*, as well as cladocerans (Nobriga 2002; Hobbs et al. 2006; Slater and Baxter 2014) and *Sinocalanus doerrii*. Nobriga (2002) found that delta smelt larvae expressed positive selection for *E. affinis* and *P. forbesi*, consuming these prey species in greater proportion than available in the environment. Such selection was not noted for other zooplankton prey. Regional differences in food use occurs, with *E. affinis* and *P. forbesi* being major prey items downstream in the low-salinity zone with a transition to *S. doerrii* and cyclopoid copepods as 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, cladocerans, and amphipods (Moyle et al. 1992; Lott 1998; Feyrer et al. 2003; Slater et al. 2019). The presence of several epibenthic species in diets therefore indicates that food sources for this species are not solely connected to pelagic pathways.

2.1.3 Distribution and Abundance

CDFW conducts three annual fish surveys (20-mm Survey, Summer Townet Survey, and Fall Midwater Trawl Survey) from which it develops indices of delta smelt's relative abundance. Each survey has variable capture efficiency (Mitchell et al. 2017), and in each, the frequency of zero catches of delta smelt is very high, largely due to the species' rarity (Latour 2016; Polansky et al. 2018) or because the 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). In addition, 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 high detection probability of delta smelt larvae and early juveniles by the 20-mm Survey at the level of replication (three samples) at each site.

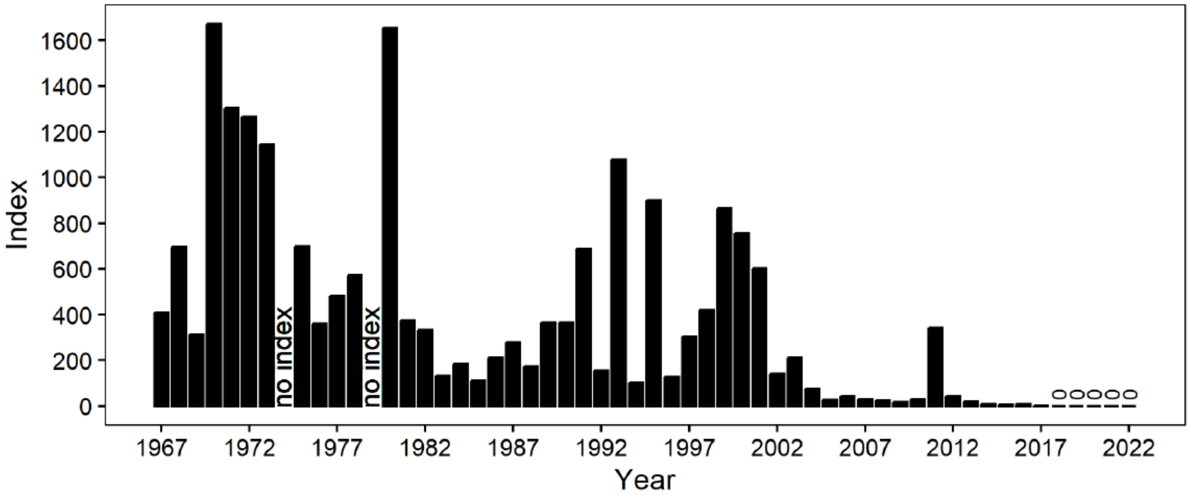
USFWS implemented a new smelt monitoring program in 2016, called the Enhanced Delta Smelt Monitoring (EDSM) program. This new program is used to measure the abundance and distribution of all life stages of delta smelt using a generalized random tessellation stratified design. Delta smelt population estimates 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 over 10 Nephelometric Turbidity Units (NTU) (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 generally distributed farther seaward than in drier years (Sommer and Mejia 2013). In contrast, during drier years, larval delta smelt are distributed further upstream and in greater abundance in the Delta (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).

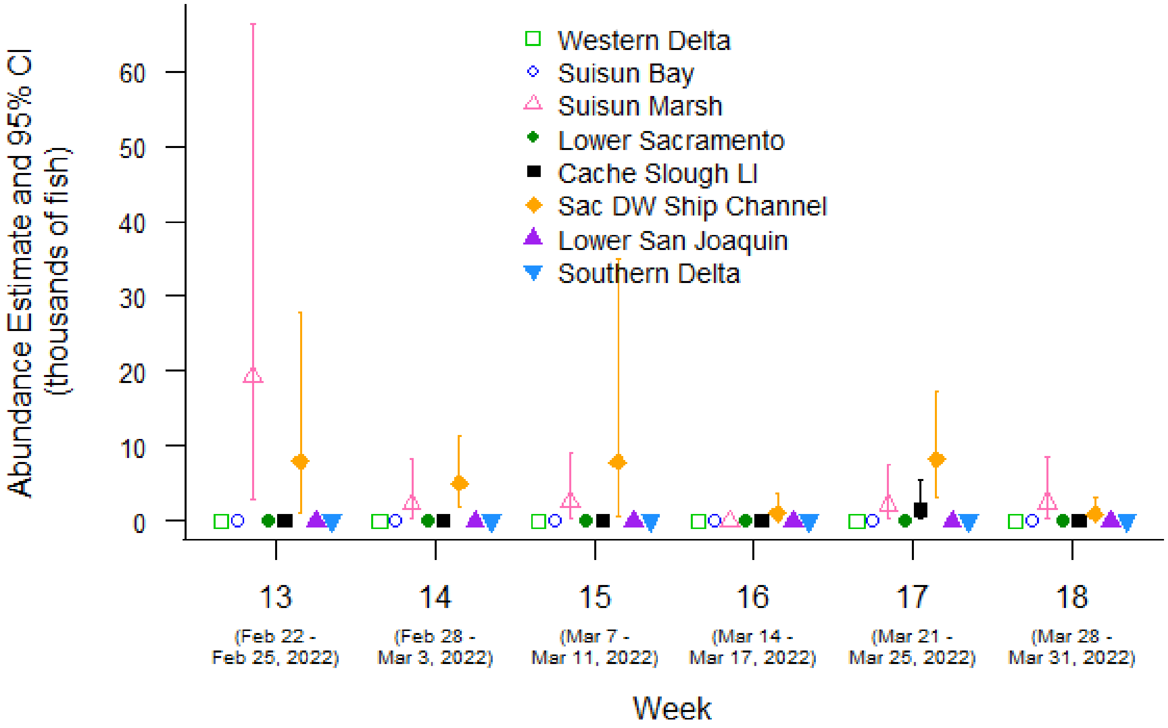
Trawl abundance indices indicate that the relative abundance of delta smelt has declined substantially since the 1980s. The observed decline in delta smelt abundance is generally consistent with declines of other pelagic species (longfin smelt and juvenile striped bass) in the Delta (Sommer et al. 2007; Baxter et al. 2010; Stompe et al. 2020).

The CDFW Fall Midwater Trawl (FMWT) delta smelt annual abundance index has been zero every year from 2018 through 2022 (Water Years 2019–2023), the lowest on record (Figure 2.1-1). All CDFW relative abundance indices show a declining trend since the early 2000s. USFWS implemented a new smelt monitoring program in 2016, called the Enhanced Delta Smelt Monitoring (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 2.1-2 for 2022 as an example).



Source: White 2022.
Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.1-1. Time Series of the Fall Midwater Trawl Water Years 1967–2022 Abundance Index for Delta Smelt



Source: U.S. Fish and Wildlife Service 2022.
Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.1-2. 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 60,000 adult equivalents of surplus hatchery-origin delta smelt each year into a portion of the current range of the species for a 3-year period (2021–2024). For example, in water year 2023, nearly 44,000 marked delta smelt reared at the UC 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 2023). The purpose of the Experimental Release of Delta Smelt Project is as part of an early experimental release effort to inform the feasibility of potential future supplementation efforts. The hatchery 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 have only 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 in Section 2.1.4, *Species Threats*), especially when these stressors are occurring in consecutive years (e.g., drought) or across sequential life stages (e.g., high water temperatures).

2.1.4 Species Threats

Delta smelt are believed to be limited by a number of stressors, including water temperature, water quality, prey availability, entrainment at water diversions, increasing frequency and duration of droughts and contaminants (Sommer et al. 2007; 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 the delta smelt to identify and describe the reasons for 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 that limit delta smelt populations, but water diversions and predation may also have significant impacts as well. The threats discussed below may be directly or indirectly affected by water operations.

All life stages of delta smelt are vulnerable to entrainment at the south Delta export facilities. In general, delta smelt salvage increases when certain conditions occur, principally when adult delta smelt move into the south Delta, when turbidity exceeds 10 to 12 NTU, and with increasing net Old and Middle River (OMR) flow reversal (i.e., more negative net OMR flows) (Grimaldo et al. 2021). However, delta smelt movement leading to entrainment cannot only be predicted by migration, turbidity and OMR flows, but is complicated by unknown behaviors and needs further study (Smith 2019). Based on field and salvage data, Kimmerer (2008, 2011) calculated that from near 0% to 25% of the larval and juvenile delta smelt population, and from 0% to 38% of the adult delta smelt population, can be entrained at the Central Valley Project (CVP) and State Water Project (SWP) annually in years with periods of high exports. Methods to calculate proportional loss estimates have been debated (Kimmerer 2011; Miller 2011) and work on entrainment estimation has continued (e.g., Smith 2019; Smith et al. 2020). Korman et al. (2021) provided preliminary estimates that adult delta smelt entrainment loss in 2002 was 35%, more than double the original estimate from Kimmerer (2008). 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), reflecting historical estimates of entrainment that were episodically high (Kimmerer 2008). More recently, delta smelt salvage has decreased due to declining abundance and salvage may no longer reflect the number of fish entrained in the south Delta or that enter the CVP and SWP fish facilities (U.S. Fish and Wildlife Service 2019a). Data on the distribution of the population (see 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 (e.g., U.S. Fish and Wildlife Service 2008:280–282; U.S. Fish and Wildlife Service 2019a:40–49).

Delta smelt are most vulnerable to entrainment when, as adults, they move from brackish water into fresh water or as larvae, when they move from fresh water in the southern and central Delta into the brackish water of Suisun Bay. While some delta smelt live year-round in fresh water far from the CVP and SWP, most rear in the low-salinity regions of the estuary, also at a relatively safe distance from the SWP and CVP pumps. The timing, direction, and geographic extent of the spawning movements of adult delta smelt affect their entrainment risk (Sweetnam 1999; Sommer et al. 2011). Unlike the years prior to the 1990s, when high salvage of adult and juvenile delta smelt occurred at high, intermediate, or low export levels, the risk of entrainment for fish that move into the central

Delta and south Delta is currently highest when net Delta outflow is at intermediate levels (about 20,000 to 75,000 cubic feet per second [cfs]), turbidity is greater than 12 NTU in the Delta (U.S. Fish and Wildlife Service 2019a), and OMR flow is more negative than -5,000 cfs (U.S. Fish and Wildlife Service 2008). In contrast, when adult delta smelt move upstream to the Sacramento River and into the Cache Slough region 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 and because there is relatively less hydrodynamic influence of the south Delta export facilities (i.e., greater OMR flow; Grimaldo et al. 2009). Hierarchical modeling has recently been developed in order 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 delta smelt south Delta entrainment loss during 1994 through 2016 to range from 53 fish in 2014 to just over 119,000 fish in 2004. Smith et al. (2020) estimated post-larval delta smelt south Delta entrainment loss during April through June 1995 through 2015 to range 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 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 (Sommer et al. 2007) era (Interagency Ecological Program Management Analysis and Synthesis Team 2015:153–162). This report stressed that the “results are preliminary and included for illustrative purposes only; peer-reviewed publications of these analyses need to be completed before they can 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 the 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). During the summer and fall, it has been hypothesized that environmental conditions improve for delta smelt as X2 moves seaward and the low-salinity zone expands habitat area (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 a habitat index calculated by Feyrer et al. (2011). However, others (e.g., Manly et al. 2015) have questioned the use by Feyrer et al. (2011) of outflow and X2 location as an indicator of delta smelt habitat because other factors may be influencing survival. Some analyses have shown no relationship of fall X2 (ICF 2017b) 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 related 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) but did find the previous fall’s mean X2 to be statistically supported as a potential negative influence on delta smelt recruitment. 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 are typically higher. As both these 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. Thus, for example, in addition to salinity, other habitat attributes increasing the suitability of juvenile delta smelt habitat include lower water clarity and greater prey density (Hamilton and Murphy 2020:110). 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 San Francisco Estuary and Delta (discussed further below). Detailed examination of a fall flow action in 2017 did not provide evidence for an increase in delta smelt prey with increased outflow resulting in X2 farther downstream (Schultz et al. 2019). 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). The Delta Smelt Summer-Fall Habitat Action under 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 considered a pelagic species, and their physical habitat is generally 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 water body 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). Feyrer et al. (2013) found that 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 in oblique trawl tows covering the full water column, suggesting strong surface orientation possibly because of visual feeding; the authors noted that these their results applied primarily to flood-tide sampling and that 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. As previously mentioned, after first flush and initial dispersal, adult delta smelt 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 may also be a key reason that juvenile delta smelt now rarely occur in the south Delta during summer (Nobriga et al. 2008). Thomson et al. (2010) found decreases in turbidity were a significant predictor of delta smelt decline in abundance over time. Grimaldo et al. (2009) found that the presence of adult delta smelt at the fish salvage facilities was linked, in part, with high turbidity associated with first flush events. Recent modeling examining future climate scenarios predicts significant increases in large flow events and sediment loading to the Delta from the Sacramento River over the next century for two representative greenhouse gas concentration pathways, which could increase turbidity (Stern et al. 2020). Turbidity may also serve as a behavioral cue for small-scale (lateral and vertical movements in the water column) and larger-scale (migratory) delta smelt movements (Bennett and Burau 2015). The decline in turbidity appears to be attributable to a decline in sediment supply from upstream, trapping 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 temperatures higher than 25.4°C (77.7°F) (Swanson et al. 2000). However, for juvenile delta smelt acclimated to 11.9°C, 15.7°C, and 19.7°C (53.4°F, 60.3°F, and 67.5°F), 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 Summer Townet and Fall Midwater Trawl. 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. Based on the Summer Townet Survey (Nobriga et al. 2008) and the 20-mm survey (Sommer and Mejia 2013), most juvenile delta smelt 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 delta smelt 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*, are hypothesized to potentially have negative effects on delta smelt (Brooks et al. 2012). While recent research has resulted in improved understanding of the factors influencing the quantity, toxicity, and location of harmful algal blooms, there are still many uncertainties about their direct and indirect effects on delta smelt relative to other factors and about what can be done to prevent them. There is no routine quantitative monitoring program in place that specifically targets harmful algae. The Summer Townet and Fall Midwater Trawl surveys have included qualitative, visual assessment of *Microcystis* since 2007. Available studies in the Delta suggest that retention time and water temperature are key environmental correlates with *Microcystis* bloom amplitude and that, once established, *Microcystis* is likely to be resistant to even very high flows as long as water quality (in particular water temperature) is favorable (Lehman et al. 2020).

Changes in phytoplankton production and phytoplankton species abundances observed and the invasion of *Potamocorbula* (overbite clam, Asian clam, Amur River clam) may have had important consequences for consumer species preyed upon by delta smelt. For example, there has been a decrease in mean zooplankton size (Winder and Jassby 2011) and a long-term decline in calanoid copepods, including a major step-decline in the abundance of the copepod *E. affinis*. These changes are possibly due to predation by *Potamocorbula* (Kimmerer et al. 1994) or to indirect effects of clam grazing on copepod food supply. Predation by *Potamocorbula* may also have been important for other zooplankton species (Kimmerer 2008).

The interaction of *Potamocorbula* grazing with the water's nutrient composition is also thought to have importance for delta smelt prey availability. As summarized by USFWS (2019a:115), diatoms (i.e., phytoplankton prey of delta smelt's zooplankton prey) preferentially take up ammonium over nitrate but grow more slowly using ammonium. Consumption of diatoms by *Potamocorbula* prevents sufficient diatom metabolization of ammonium to lower levels that would allow more rapid diatom growth rates and greater diatom abundance for consumption by delta smelt zooplankton prey. The largest source of dissolved ammonium in the Bay-Delta is the Sacramento Regional Wastewater Treatment Plant, for which facility upgrades reducing ammonium concentrations beginning in 2023 should demonstrate how important this nutrient source is to limiting diatom production in the Bay-Delta (U.S. Fish and Wildlife Service 2019a:115). A recent analysis concluded high ammonium loading is not a driver of low productivity in the San Francisco Bay-Delta (Strong et al. 2021).

In addition to a long-term decline in calanoid copepods and mysids (Orsi and Mecum 1986) 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 somewhat 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 southern Delta (Winder and Jassby 2011).

Another introduced cyclopoid copepod, which may affect delta smelt growth, is *Limnoithona tetraspina.* Experimental studies addressing this issue suggest that larval delta smelt will consume and grow on *L. tetraspina*, but growth is slower than with *P. forbesi* (Kimmerer et al. 2011). This copepod significantly increased in the Suisun Bay region beginning in the mid-1990s. It is now the most abundant copepod species in the 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). 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 over the last decade (Hennessy 2010), although its suitability as food for delta smelt remains unclear.

In addition to the previously mentioned subsidy of *P. forbesi* to the low-salinity zone from the Delta that has been positively related to Delta outflow (Kimmerer et al. 2018, 2019), Hamilton et al.

(2020) conducted modeling of potential Delta 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 to 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 or zooplankton productivity (Hammock et al. 2019a; Kimmerer et al. 2019). Tidal wetlands have been suggested to confer substantial benefits to the foraging success of delta smelt, on the basis of observed stomach fullness increases with increasing adjacent tidal wetland area (Hammock et al. 2019b).

Modeling suggests that 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). These metrics are composites of the relative abundance of Mississippi Silverside (*Menidia audens*), largemouth bass (*Micropterus salmoides*), and other centrarchids; these species are potential predators of concern because of their increasing abundance (Bennett and Moyle 1996; Brown and Michniuk 2007; Thomson et al. 2010) and because of inverse correlations between largemouth bass abundance and delta smelt abundance (Nobriga and Feyrer 2007; Thomson et al. 2010; Maunder and Deriso 2011). These correlations could represent predation on delta smelt by largemouth bass or, alternatively, 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 exists (Ferrari et al. 2014). However, there is little evidence that largemouth bass are 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 that Mississippi silversides currently are the most important predators of delta smelt early life stages, as reflected in recent studies of delta smelt DNA in the prey consumed by silversides (Baerwald et al. 2012; Schreier et al. 2016), with two recent statistical examinations finding support for 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 be at 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 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 suggested 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 (*Morone saxatilis*) 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% of the age 2 striped bass diet by volume during the summer. Thomas (1967) reported on spatial variation in the striped bass diet composition based on collections throughout the San Francisco Estuary and the Sacramento River above tidal influence. Delta smelt accounted for 8% of the spring diet composition and about 16% of the summer diet composition in the Delta. Brandl et al. (2021) used genetic analysis and found 1.3% of striped bass had delta smelt in their guts, noting that this was higher than in previous reports (0%–0.4%; 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 to controlling delta smelt 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 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 of 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 through 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 as estimated by 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 similar to that 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 be a source of 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 temperatures likely present a significant conservation challenge for delta smelt (Brown et al. 2013, 2016). Mean annual water temperatures within the Delta are expected to increase steadily during the second half of this century (Cloern et al. 2011). 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). Flow Alteration–Management, Analysis, and Synthesis Team (2020:20) concluded that temperature is likely a primary factor in the lack of response of the delta smelt population to the high flows in 2017.

The frequency of occurrence (percentage of samples) of delta smelt by life stage and region from monitoring in the San Francisco Estuary and Delta as assessed by Merz et al. (2011) is provided in Table 2.1-1.

Table 2.1-1. Average Annual Frequency (Percent) of Delta Smelt Occurrence by Life Stage, Interagency Ecological Program Monitoring Program, and Region

	Average Annual Frequency (%)										
Region **Life Stage:**	**Larvae (<15 mm)**	**Sub-Juvenile (≥15, <30 mm)**		**Juvenile (30–55 mm)**			**Subadult (>55 mm)**	**Mature Adults (>55 mm)**		**Pre-Spawning** [a]	**Spawning** [a]
Monitoring Program:	20-mm	20-mm	STN	20-mm	STN	FMWT	FMWT	BS	BMWT	KT	KT
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
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; California Department of Fish and Wildlife n.d.

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 Spring Kodiak Trawl database were classified by CDFW per the description available at: http://www.dfg.ca.gov/delta/data/skt/eggstages.asp (California Department of Fish and Wildlife n.d.).

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

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

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2.2 Longfin Smelt

2.2.1 Legal Status

In June 2009, the California Fish and Game Commission approved regulations listing longfin smelt as threatened under the CESA. Specific to the federal ESA, the longfin smelt Bay-Delta DPS was determined to be a distinct population segment (DPS) that warranted listing as an endangered or threatened species under the ESA on April 2, 2012 (77 FR 19756), but the listing was precluded by higher priority listing actions. On October 7, 2022, USFWS published a proposed rule that would find the longfin smelt, Bay-Delta DPS as an endangered species under the ESA. This proposed rule's original comment period closed on December 6, 2022. On February 27, 2023, the USFWS reopened a 30-day comment period to allow for a public hearing held on March 14, 2023.

2.2.2 Life History and General Ecology

The longfin smelt (*Spirinchus thaleichthys)* is a small, euryhaline, anadromous, and largely semelparous[3] fish with a life cycle of approximately 2 to 3 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 San Francisco Estuary and Delta, although these 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 in 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 freshwater in tributaries to the 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 of longfin smelt 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 swim back into the estuary (Rosenfield and Baxter 2007). Longfin smelt 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 2.2-1) (Moyle 2002; Pinnix et al. 2004; Garwood 2017). 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).

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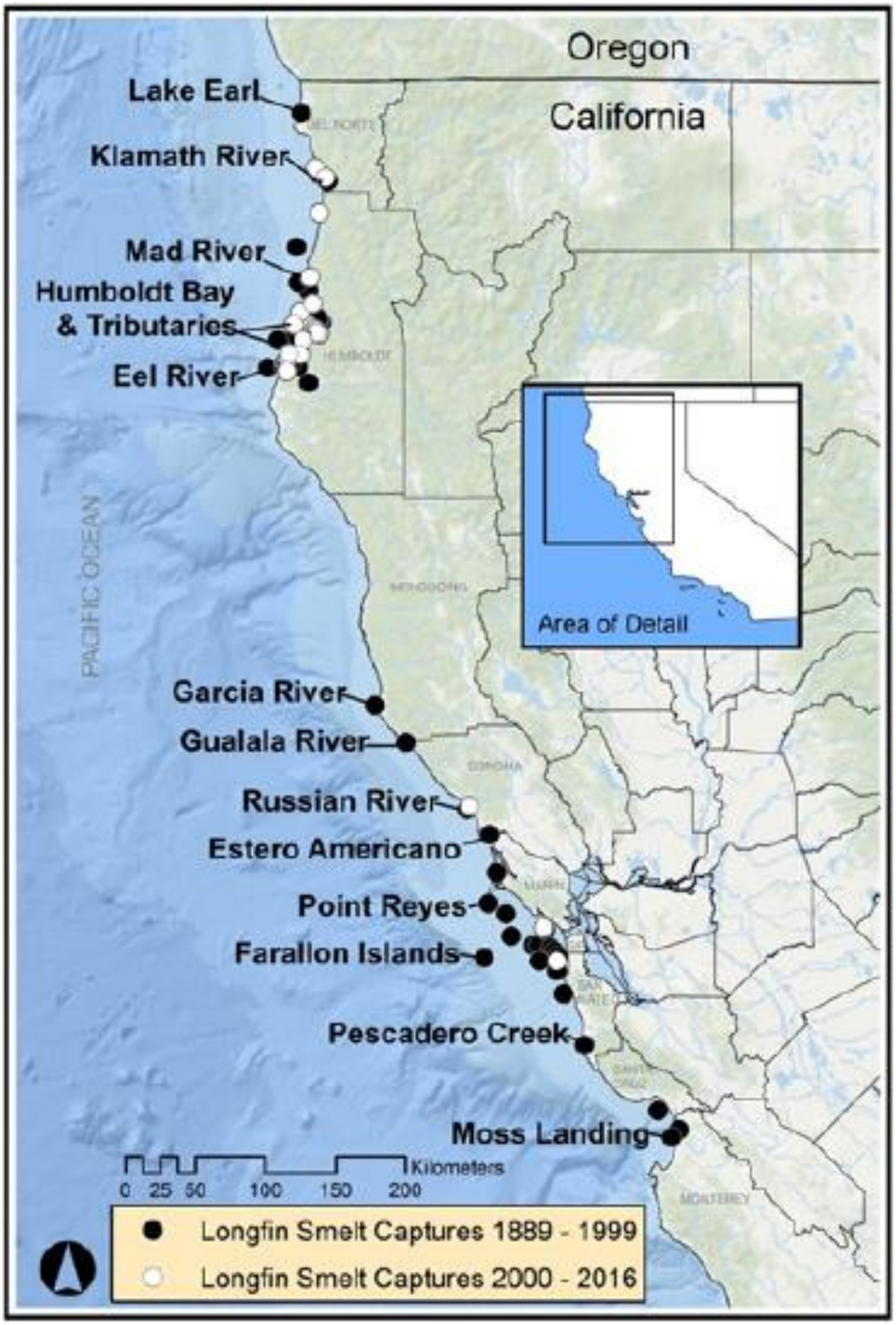
[3] Moulton (1974:50) stated, "Apparently a ripe female enters the river and spawns only once"; and Wang (2007: 39) cited Moyle (1976) in noting that "Most die after spawning, but a few females may live and spawn a second time."



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.

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.2-1. Locations of Longfin Smelt Captures, 1889–2016, Excluding the San Francisco Estuary and Delta

Genetic isolation exists between the population segment of longfin smelt in the San Francisco Estuary and populations north of the Columbia River estuary (Stanley et al. 1995; Israel and May 2010; Saglam et al. 2021). Saglam et al. (2021) found evidence for significant contemporary migration northward from the San Francisco Estuary population to Humboldt Bay and the Columbia River estuary, without significant southward migration. Due to the low likelihood of southward migration from more northern breeding populations as close as Humboldt Bay, USFWS determined that listing of the San Francisco Estuary population as a DPS is warranted (U.S. Fish and Wildlife Service 2012a). The Bay-Delta DPS of longfin smelt occurs 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, north as far as the town of Colusa on the Sacramento River, east as far as Lathrop on the San Joaquin River, and south as far 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).

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 to 42 days (Rosenfield 2010). Evidence for individuals spawning multiple times in a season has not been provided, although some females may undergo repeated spawning events (see the discussion above related to semelparity). Newly hatched larvae have been observed in salinities from freshwater up to 12 practical salinity units (psu) with peak observations occurring between 2 and 4 psu (Grimaldo et al. 2017). Early juvenile longfin smelt (20–40 mm SL) are found in salinities up to 30 psu, but most are found in salinities between 2 to 18 psu (MacWilliams et al. 2016). By late summer, late juveniles can tolerate full seawater.

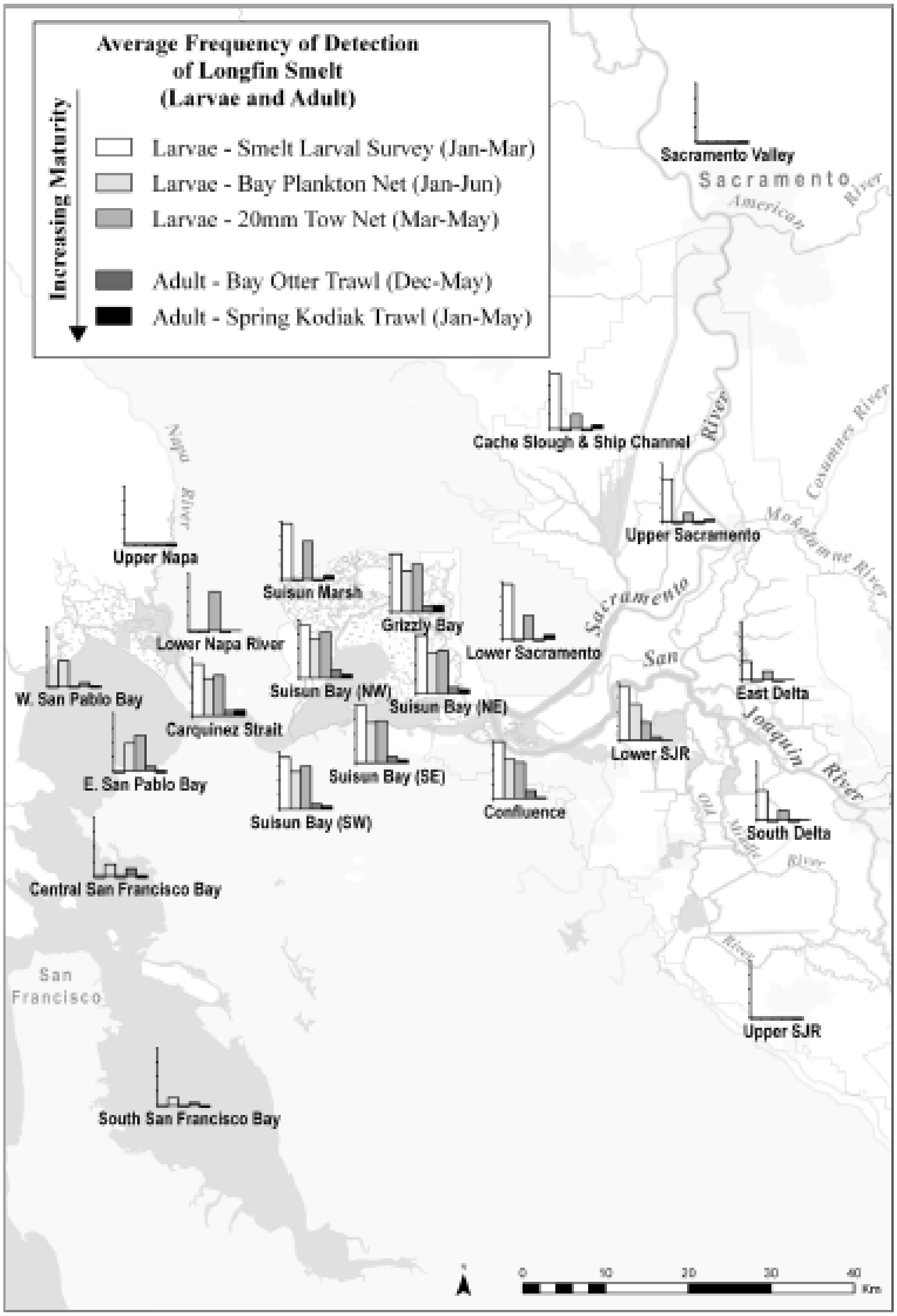
Longfin smelt are anadromous and semelparous, moving from saline to brackish or fresh water for spawning from November to May (Grimaldo et al. 2017; Lewis et al. 2020). Longfin smelt usually live for 2 years, spawn, and then die, although some individuals may spawn as 1-year-old or 3-year-old fish before 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). 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 reared in salinities around 2 ppt.

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 (~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) believed 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), longfin smelt 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). Mysid density is positively related to spring Delta outflow (negatively related to spring X2) (Mac Nally et al. 2010), although note that Kimmerer (2002) found a changing relationship to X2 for the mysid *Neomysis mercedis* (negative prior to 1987; positive following 1987).

2.2.3 Distribution and Abundance

During late summer and early fall, juvenile and adult longfin smelt within the San Francisco Estuary are more common throughout San Francisco Bay than in other landward areas (Rosenfield and Baxter 2007; MacWilliams et al. 2016), although the extent of marine migration has yet to be quantified. During the spawning period in late fall and early winter, adults are more commonly found in San Francisco Bay tributaries and marshes (Lewis et al. 2020; Grimaldo et al. 2020), Suisun Bay, and the Delta (Rosenfield and Baxter 2007). Larval longfin smelt are broadly distributed throughout San Francisco Bay and its associated tributaries during wet years (MacWilliams et al. 2016; Lewis et al. 2020; Parker et al. 2017; Grimaldo et al. 2020). Analyses of multiple surveys by Merz et al. (2013) found that larvae were more frequently detected in the Delta in drier years than in wet years (Figure 2.2-2); however, the limited extent of the Smelt Larval Survey to landward regions does not account for potential spawning in tributaries of San Francisco Bay (Lewis et al. 2020). In the SLS, albeit limited to regions landward of San Pablo Bay, more than 50% of the measured larval abundance in any given year between 2009 and 2015 occurred in Suisun Bay and Suisun Marsh (Grimaldo et al. 2017). In a two-year study with greater spatial coverage than long-term surveys, Grimaldo et al. (2020) found that during the low-flow year of 2016, larval and post-larval longfin smelt were mostly distributed in Suisun Bay, whereas in the high-flow year of 2017, larval longfin smelt were mostly found in San Pablo and South Bays. Some juveniles and adults are believed to move to the coastal ocean during the summer and fall (Rosenfield and Baxter 2007; MacWilliams et al. 2016).

Abundance indices for the longfin smelt population have undergone a decline over time. For example, there was an approximate thirty-fold reduction in the fall midwater trawl index since the early 1980s (Figure 2.2-3) (Rosenfield and Baxter 2007; Sommer et al. 2007; Kimmerer et al. 2009), and an index of 2-year-olds based on the San Francisco Bay Study midwater and otter trawl went down from a mean of 1,931 in 1980 through 1986 (prior to the *Potamocorbula* invasion) to a mean of 918 during 1987 through 2002, with a further decline following the onset of the Pelagic Organism Decline to a mean of 422 during 2003 through 2013 (Nobriga and Rosenfield 2016). The rate of decline of the population suggested by abundance indices has been particularly steep, especially since the onset of the Pelagic Organism Decline (Sommer et al. 2007; 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, the slope of the relationship between winter-spring flow and fall longfin smelt abundance indices remains unchanged, suggesting that flow or hydrological conditions may be strong drivers of their population abundance (Kimmerer et al. 2009; Maunder et al. 2015; Nobriga and Rosenfield 2016), although specific mechanisms are unknown. The intercept of this relationship has dropped nearly twofold, possibly because of declining food supply related to *Potamocorbula* (Kimmerer et al. 2009).

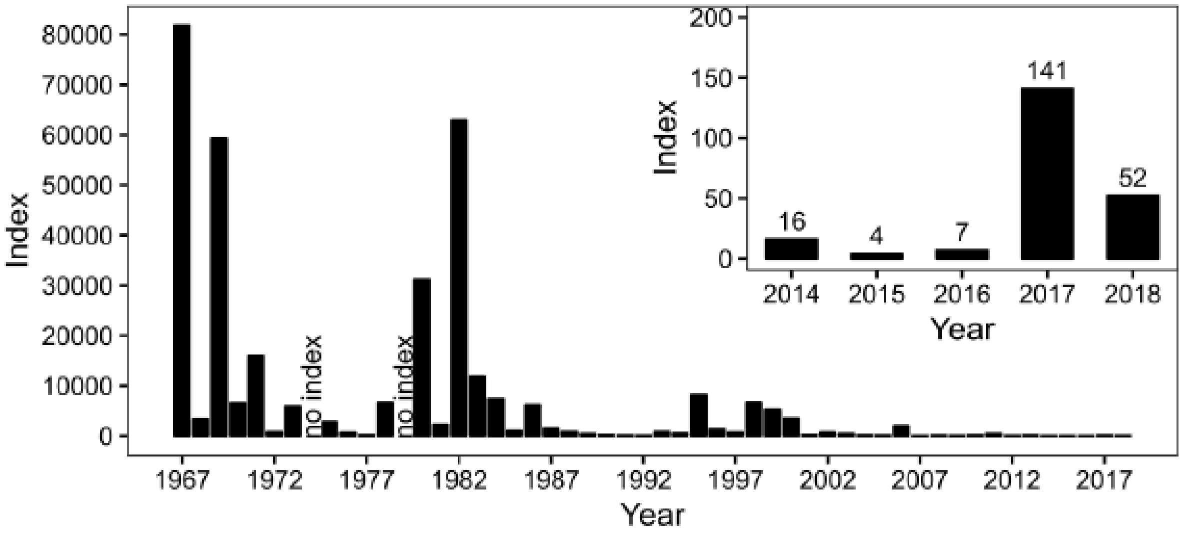


Source: Merz et al. 2013.

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).

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.2-2. Average Annual Frequency of Longfin Smelt Detection (%) for Larval and Adult Life Stages by Region and Interagency Ecological Program Survey Type



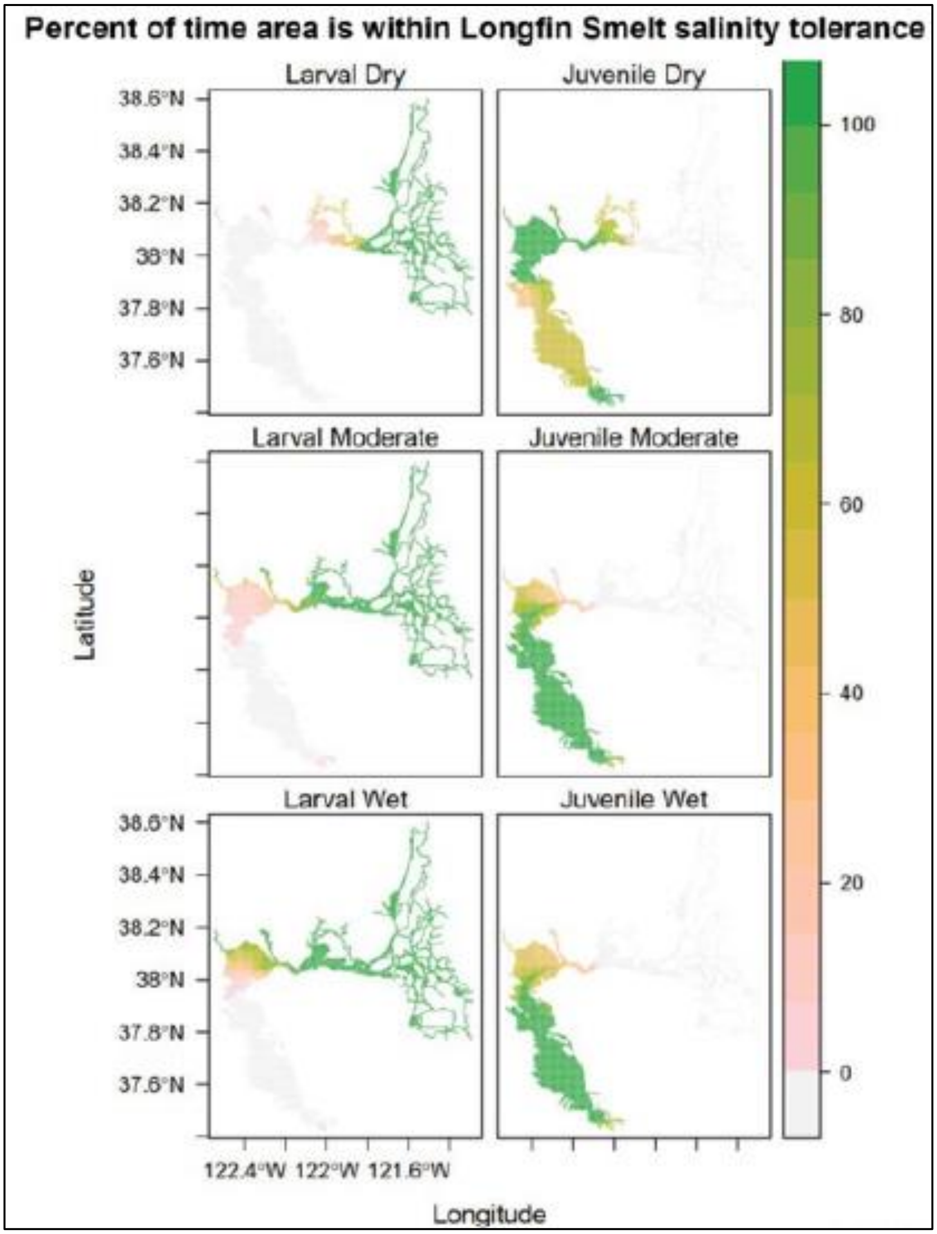
Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.2-3. Longfin Smelt Fall Midwater Trawl Abundance Index, 1967–2018

2.2.4 Species Threats

There are several threats to longfin smelt. The USFWS (77 FR 19756) 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 discussion below also describes other threats that have been noted (e.g., California Department of Fish and Game 2009a), although as discussed further below, not all have been concluded to be of significance to the species (e.g., entrainment; U.S. Fish and Wildlife Service 2012a).

Longfin smelt are vulnerable to entrainment at the south Delta export facilities. The annual number of longfin smelt salvaged has been generally low since the 1980s, except in some years (1988, 2002), as illustrated for the SWP salvage facility (see Figure 2-5 in 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 2-6 in California Department of Water Resources 2019:2-15), probably as a result of the landward shift in distribution in drier years. Figure 2.2-4 shows the distribution of larval and juvenile longfin smelt salinity tolerance in water years of varying runoff.



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]) (all runoff values based on California Data Exchange Center [CDEC] [2020]). 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.

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.2-4. Distribution of Larval and Juvenile Longfin Smelt Salinity “Tolerance” (10th–90th Percentile Salinity of Catch) 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, it is difficult to ascertain how many larvae are entrained (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 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% 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% of the population in the moderately dry year of 2013. Application of the same methods gave estimates of just under 1% larval entrainment in 2021 and 1.3% larval entrainment in 2022 (Resource Management Associates 2023).

As previously described, longfin smelt abundance indices are 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), but 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 with salinity around 0 to 7. 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.

Adult longfin smelt use tidal marshes for spawning (Lewis et al. 2020). Larval longfin smelt use marsh and shoal habitats as rearing habitat (Grimaldo et al. 2017; Grimaldo et al. 2020). Juvenile longfin smelt 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 2.2-4 shows the availability of habitat for larval and juvenile longfin smelt based on longfin smelt salinity *tolerance*[4] 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% of larval and juvenile longfin smelt were observed in the Smelt SLS and Bay Study surveys, respectively. Note that 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; although see 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 of longfin smelt 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 longfin smelt abundance indices in the population dynamics model of Maunder et al. (2015).

The effect of nonnative predators, such as Mississippi silversides (*Medinia beryllina*) and striped bass, has 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 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 (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).

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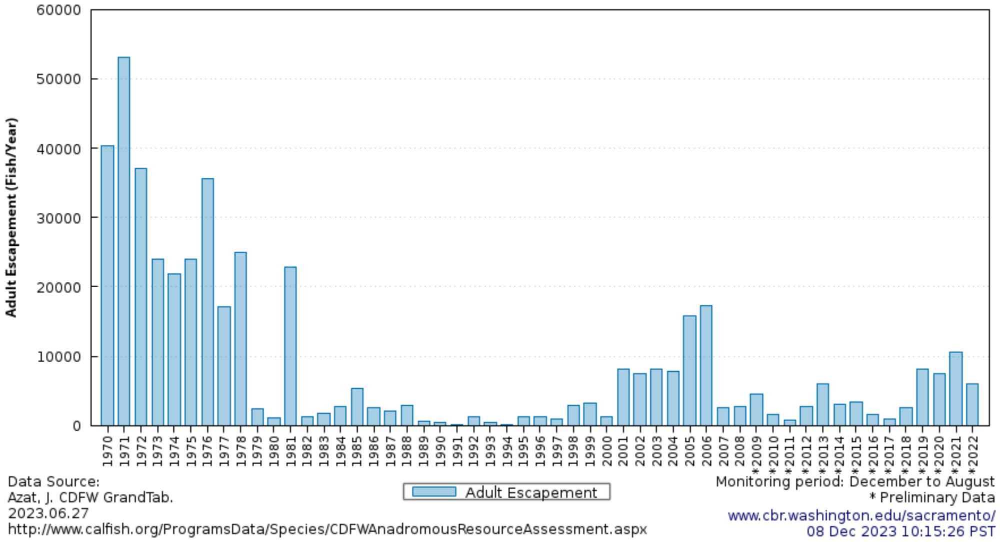
[4] As noted in Figure 2.2-4, *tolerance* is not taken to mean physiological tolerance, but as described above, the 10th–90th percentile salinity of longfin smelt catches.

2.3 Chinook Salmon – Winter-Run

2.3.1 Legal Status

On May 16, 1989, the California Fish and Game Commission listed the winter-run Chinook salmon (*Oncorhynchus tshawytscha*) as endangered under CESA due to persistent long-term declines (Figure 2.3-1). The National Marine Fisheries Service (NMFS), under an emergency interim rule, listed the winter-run Chinook salmon as a threatened species under the federal ESA in August 1989 (54 FR 32085). In 1994, NMFS reclassified the species' population as endangered due to several factors: the continued decline and increased variability of run sizes including expected weak returns due to small year classes in 1991 and 1993 and continuing threats to the species (59 FR 440). The ESU species' population 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. Although restoration actions in Battle Creek were not yet complete, there was adequate habitat for some fish to spawn and produce juveniles (U.S. Fish and Wildlife Service 2020a; see further discussion of Battle Creek restoration below). NMFS reaffirmed the listing of the winter-run Chinook salmon as endangered on June 28, 2005 (70 FR 37160) and expanded the species' population to include winter-run Chinook salmon produced by the Livingston Stone National Fish Hatchery artificial propagation program in the species' population. During 1970–2020, the highest estimated run sizes occurred in the early 1970s (~40,000–50,000 adults), declining to a low in the early 1990s (~200 adults), increasing to higher levels in mid-2000s (up to ~17,000 adults), before varying between just under 1,000 and ~10,000 adults since 2006 (Figure 2.3-1). Escapement in brood year 2022 was ~ 6,067 fish (Columbia Basin Research, University of Washington 2023), lower than in the previous few years (Marcinkevage 2023:4).



Source: Columbia Basin Research, University of Washington 2023 Note: Includes in-river and hatchery fish.

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.3-1. Winter-Run Chinook Salmon Adult Annual Escapement in the Central Valley, 1970–2022

2.3.2 Life History and General Ecology

Adult winter-run Chinook salmon enter the San Francisco Bay in November to begin their spawning migration and continue upstream from December through early August to the extent of anadromy at the base of Keswick Dam (Figure 2-7). Adults prefer to hold in deep, cold pools until they are sexually mature and ready to spawn in spring or summer. Upstream holding may last for a period of up to 8 months (Windell et al. 2017: 32). 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 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 at least 700 subadults and adults in 2020 (U.S. Fish and Wildlife Service 2020a). Current spawning is confined to the mainstem of the Sacramento River above Red Bluff Diversion Dam (RBDD) (River Mile [RM] 243) and below Keswick Dam (RM 302) (National Marine Fisheries Service 2014), with access to historical habitat in upper Sacramento River tributaries no longer available (Figure 2-7). Until recent years, salmon passage was not possible above the Coleman Hatchery barrier weir located on Battle Creek due to the Wildcat Diversion Dam and Canal, which were removed in 2010. Beginning in 2012 and completed in 2018, the construction of the automated fish screens and fish ladders at the Eagle Canyon Diversion Dam and North Battle Creek Feeder Diversion Dam sites were implemented to provide fish passage on North Fork Battle Creek (Bureau of Reclamation 2022). Windell et al. (2017:6–8) reviewed habitat characteristics and factors affecting egg survival, among which were in-river fishery and trampling, toxicity and contaminants; redd quality; stranding and dewatering; dissolved oxygen; pathogens; water temperature; sedimentation and gravel quantity; and predation risk (see Section 2.3.4, *Species Threats*, for more details). Following spawning, fry and juvenile downstream movement begins in July or August, as shown by monitoring at Red Bluff (Table 23-1).In addition to the Sacramento River, juveniles have also been found to rear in areas such as 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% of surviving adults reared in nonnatal habitats as juveniles. The lower reaches of the Sacramento River, the Delta, and San Francisco Bay serve as migration corridors for both smolts and adults and are thought to serve as juvenile rearing habitat. Juvenile winter-run Chinook salmon generally begin to enter the Delta in October[5] and smolt outmigration continues until April. Timing of smolt movement is thought to be correlated with winter rain events that result in pulse flows in the Sacramento River (del Rosario et al. 2013). Fry and smolts are known to use the Sacramento–San Joaquin Delta and Estuary as rearing habitat before entering the ocean (Sturrock et al. 2015). In addition to monitoring salvage of winter-run Chinook salmon at the Tracy Fish Collection Facility (TFCF) and the John E. Skinner Delta Fish Protective Facility (Skinner Fish Facility) in the south Delta, temporal occurrence of each life stage in the project area is monitored using screw trapping data in the rivers, trawls, and beach seines in the estuary and, more recently, acoustic tagging using a network of receivers located throughout the extent of their range, from Keswick Dam to the Golden Gate Bridge (e.g., Klimley et al. 2017).

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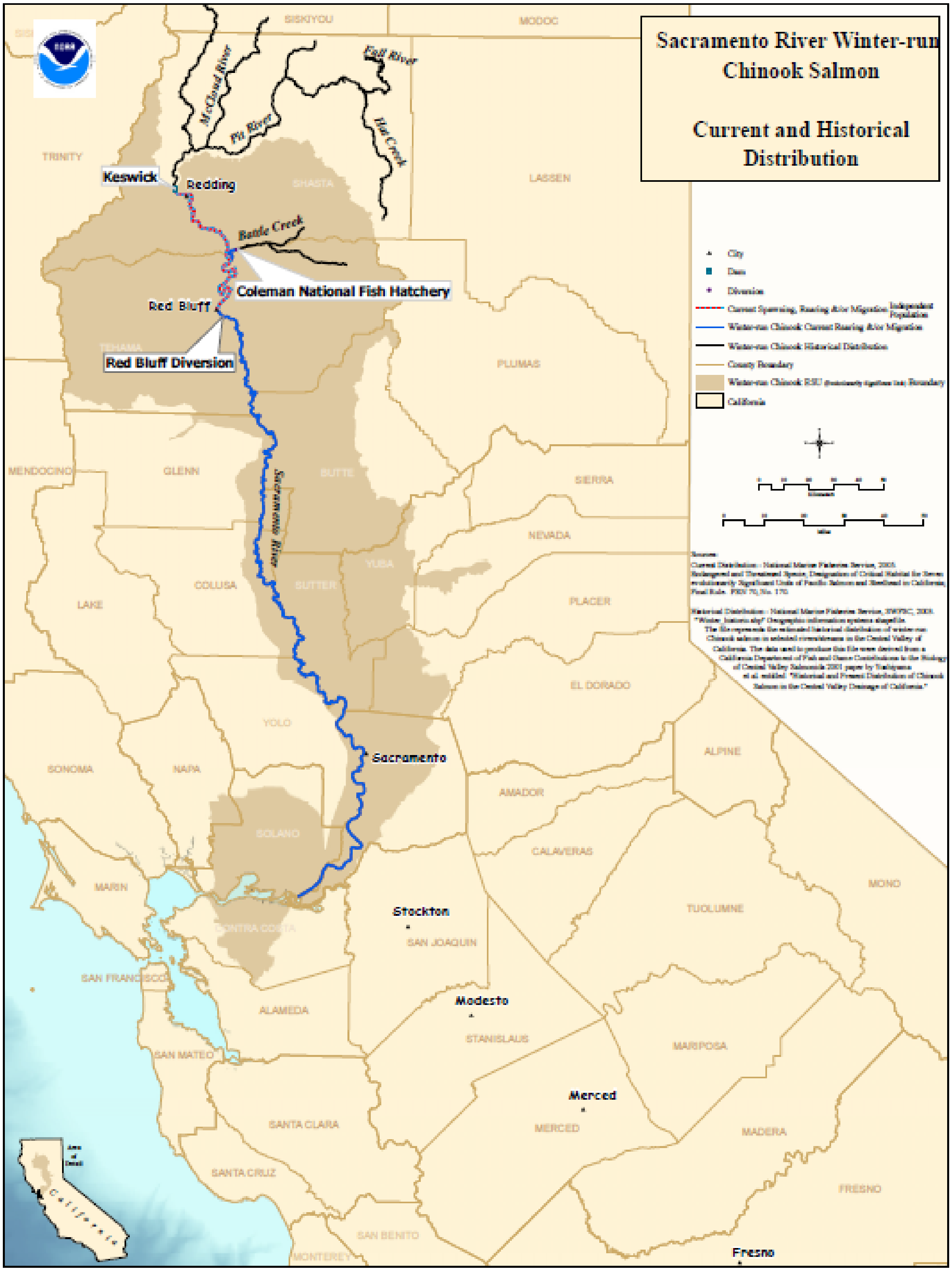
[5] Winter-run Chinook-sized juveniles have been collected as early as August in rotary screw traps at Knights Landing, and as early as September in Sacramento beach seines and trawls (see California Department of Water Resources 2023:Appendix 12A, Attachment 12A-1, Figures 12A.1-3, 12.1-4, and 12A.1-5). Juvenile Chinook salmon captured during trawling at Sacramento that were confirmed to be winter-run were captured as early as October (Brandes et al. 2021:7).



Source: National Marine Fisheries Service 2014:12.

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.3-2. Current and Historical Winter-Run Chinook Salmon Distribution

Generalized life stage timing for winter-run Chinook salmon is summarized in Tables 2.3-1 and 2.3-2, with a quantitative summary for juveniles in Tables 2.3-3 and 2.3-4. Appendix 12A, Attachment 12A.1, *Juvenile Salmonid Monitoring, Sampling, and Salvage Timing Summary,* of the Final EIR, provides additional summary information from the SacPAS database (California Department of Water Resources 2023).

Table 2.3-1. 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 2.3-2. 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

Note: Table reflects monitoring based on length-at-date classification of juvenile winter-run Chinook Salmon. See Tables 2.3-3 and 2.3-4 for quantitative summary. * The data in this category reflects juveniles entrained into the salvage facilities. See also annual summaries in California Department of Water Resources 2023: Appendix 12A, Attachment 12A.1, *Juvenile Salmonid Monitoring, Sampling, and Salvage Timing Summary from SacPAS*).

Source: National Marine Fisheries Service 2019:68.

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Table 2.3-3. 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% (588)
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 Trawl at Sherwood Harbor	1/3/2000–7/29/2022	Trawl tows	3.7% (3,402)	6.4% (3,273)	5% (3,524)	2.5% (3,402)	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 Trawl 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. Numbers in parentheses indicate number of sampling units.

Table 2.3-4. 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% (588)
Sacramento River RST at Tisdale	7/6/2010–12/18/2022	Days	5.7% (298)	20.4% (270)	8.8% (304)	0% (313)	0% (278)	0% (111)	0% (72)	0% (662)	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	% (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)	% (2,043)	% (2,090)	0% (2,086)	0% (3,316)	0% (3,480)	0.4% (3,325)
Sacramento Trawl 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)	% (2,700)	% (2,637)	0% (2,591)	0% (2,644)	0% (2,631)	0.5% (3,349)
Midwater Trawl 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)	% (2,441)	% (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)	% (960)	% (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. Numbers in parentheses indicate number of sampling units.

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2.3.3 Distribution and Abundance

Relative distribution, abundance, and migration timing in the Delta can be inferred from salvage monitoring data, the USFWS Delta Juvenile Fish Monitoring Program, Sherwood and Mossdale Trawls, and the Chipps Island Trawl. 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 identify variation in quality of rearing habitat. Johnson et al. (2017) give an overview of the various sampling programs providing monitoring of winter-run Chinook salmon.

2.3.4 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 grounds 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 is dependent on maintaining adequate coldwater pool in Shasta Reservoir to maintain suitable temperatures for winter-run Chinook salmon egg incubation, fry emergence, and juvenile rearing. Coldwater releases are facilitated by the operation of a temperature control device that allows selective withdrawal of water from different levels of the reservoir depending on temperature. Warm water 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 was low in brood year 2022 (2.2%), 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). Temperature-dependent egg mortality was estimated to have been 75% in 2021 (National Marine Fisheries Service Southwest Fisheries Science Center 2021), compared to 17% in 2022 with the more proactive water temperature management (National Oceanic and Atmospheric Administration Fisheries 2023).

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 Central Valley Project and State Water Project Biological Opinion Reasonable and Prudent Alternative Actions I.6.1 and I.7* (Yolo Bypass Restoration Plan) includes notching the Fremont Weir, which will provide access to floodplain habitat for juvenile salmon over a longer period (California Department of Water Resources and Bureau of Reclamation 2012:3) and the California Department of Fish and Wildlife (2020a) SWP ITP Condition of Approval 9.2.2 *Implement the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Project* requires that the project be complete by 2026. 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 may be affected by reverse Old and Middle River flows that are exacerbated by south Delta export facility operations at the CVP and SWP pumping plants (discussed further in Section 2.4, *Chinook Salmon—Spring-Run*). Bidirectional flow in the Sacramento River at Georgiana Slough associated with lower Sacramento River inflow to the Delta causes juvenile Chinook salmon to enter into the interior Delta in greater numbers than with unidirectional flow at high Sacramento River inflow, which results in greater travel times and lower survival (Perry et al. 2013, 2018; see additional discussion in Section 2.4). This and other studies have typically used hatchery-origin juvenile late fall–run Chinook salmon large enough to bear acoustic tags, with the general movement patterns often 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. 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 to winter-run Chinook salmon by NMFS (2014:27) include blockage of historical staging and spawning habitat by Shasta and Keswick Dams; flow fluctuations in the upper Sacramento River during spawning and incubation, water pollution, and water temperature impacts in the upper Sacramento River during embryo incubation; loss of juvenile rearing habitat in the form of lost natural river morphology and function, and lost riparian and instream cover; predation during juvenile rearing and outmigration; ocean harvest; and south Delta entrainment. A very recent potential threat identified for all runs of Chinook salmon is Thiamine Deficiency Complex, possibly the result of the oceanic diet of adults transferring negative effects to juveniles (National Oceanic and Atmospheric Administration Fisheries 2020a)[6]. Recent temperature modeling shows higher sensitivity to increases in water temperature because it leads to exponential increases in oxygen demand with a rise in temperature during the final weeks of egg-embryo maturation before the alevin stage (Martin et al. 2017; Anderson 2018). 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 the leading causes of mortality of eggs and juveniles (Dudley 2018). Turbidity reduces predation of migrating juveniles and carrying capacity for larger juveniles is often

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[6] 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% 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).

reached (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 that there is evidence for increased stranding risk 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 thereby 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). Note that 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 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% in 2017 and 2018, and remaining at around 40% 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 5-year species status evaluation to moderate risk at the time of the most recent (2015) evaluation (National Marine Fisheries Service 2016a:34).

Climate experts predict physical changes to ocean, river, and stream environments along the West Coast that include warmer atmospheric temperatures, diminished snow pack resulting in altered stream flow volume and timing, lower late summer flows, a continued rise in stream temperatures, and increased sea-surface temperatures and ocean acidity resulting in altered marine and freshwater food-chain dynamics (Williams et al. 2016). Climate change and associated 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 2014; Lindley et al. 2009). Because the winter-run Chinook salmon rely on the coldwater pool in Shasta Reservoir to maintain spawning conditions in the mainstem Sacramento River, this run is particularly at risk from global warming. Drought years are predicted to occur with greater frequency in the Sacramento Valley with climate change (Purkey et al. 2008). The years of 2012–2016 had drought conditions (Lund et al. 2018), with the mean annual percentage of land experiencing some level of drought conditions ranging from 87.2% in 2012 to nearly 100% in 2014 and 2015; further widespread drought began in 2020 (~68% of land experiencing some level of drought) and increased in extent so that in 2021 (99.9%) and 2022 (100%) all or nearly all of California was experiencing some level of drought.[7] High rainfall early in 2023 reduced the extent of California experiencing drought, with less than 4% of the state experiencing some level of drought by December 5, 2023. 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. Juvenile salmon are thought to use turbid water to avoid detection by

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[7] All statistics on percentage of California land area experiencing some level of drought cited in this section were based on data downloaded from US Drought Monitor, available https://www.drought.gov/states/california#historical-conditions, accessed December 12, 2023.

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 Nino 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 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).

2.4 Chinook Salmon – Spring-Run

2.4.1 Legal Status

Spring-run Chinook salmon (*Oncorhynchus tshawytscha*) were historically the most abundant run in the Central Valley (Yoshiyama et al. 1998). The Central Valley spring-run Chinook salmon has independent populations in Butte Creek, Mill Creek, and Deer Creek, with repopulation of a historically independent population in Battle Creek occurring; dependent populations occur in Antelope Creek, Big Chico Creek, Clear Creek, and Cottonwood/Beegum Creek (National Marine Fisheries Service 2016b; Goertler et al. 2020:2). There are also San Joaquin River spring-run Chinook salmon as a result of reintroduction efforts, and spring-running Chinook salmon in San Joaquin River tributaries. Spring-run Chinook salmon were extirpated from most rivers by mining or dam construction (Williams 2006). Due to the small number of these populations remaining 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 the 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 (Central Valley) spring-run Chinook salmon was listed as threatened under the ESA in 1999 because of the reduced range and small size of remaining spring-run Chinook salmon populations (64 FR 50393). On June 28, 2005, NMFS published the final hatchery listing policy (70 FR 37204) and reaffirmed the threatened status of the species' population (70 FR 37160). The species' population consists of naturally spawned spring-run Chinook salmon originating from the Sacramento River and its tributaries, and also from the Feather River Fish Hatchery (FRFH) Spring-Run Chinook Program (National Marine Fisheries Service 2016b).

2.4.2 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 adults and juveniles occupy freshwater habitat. Adult spring-run Chinook salmon enter fresh water as sexually immature fish between mid-February and July and remain in deep cold pools in proximity to spawning areas until late summer and early fall, when they are sexually mature and ready to spawn, depending on water temperatures (California Department of Fish and Game 1998; National Marine Fisheries Service 2009).

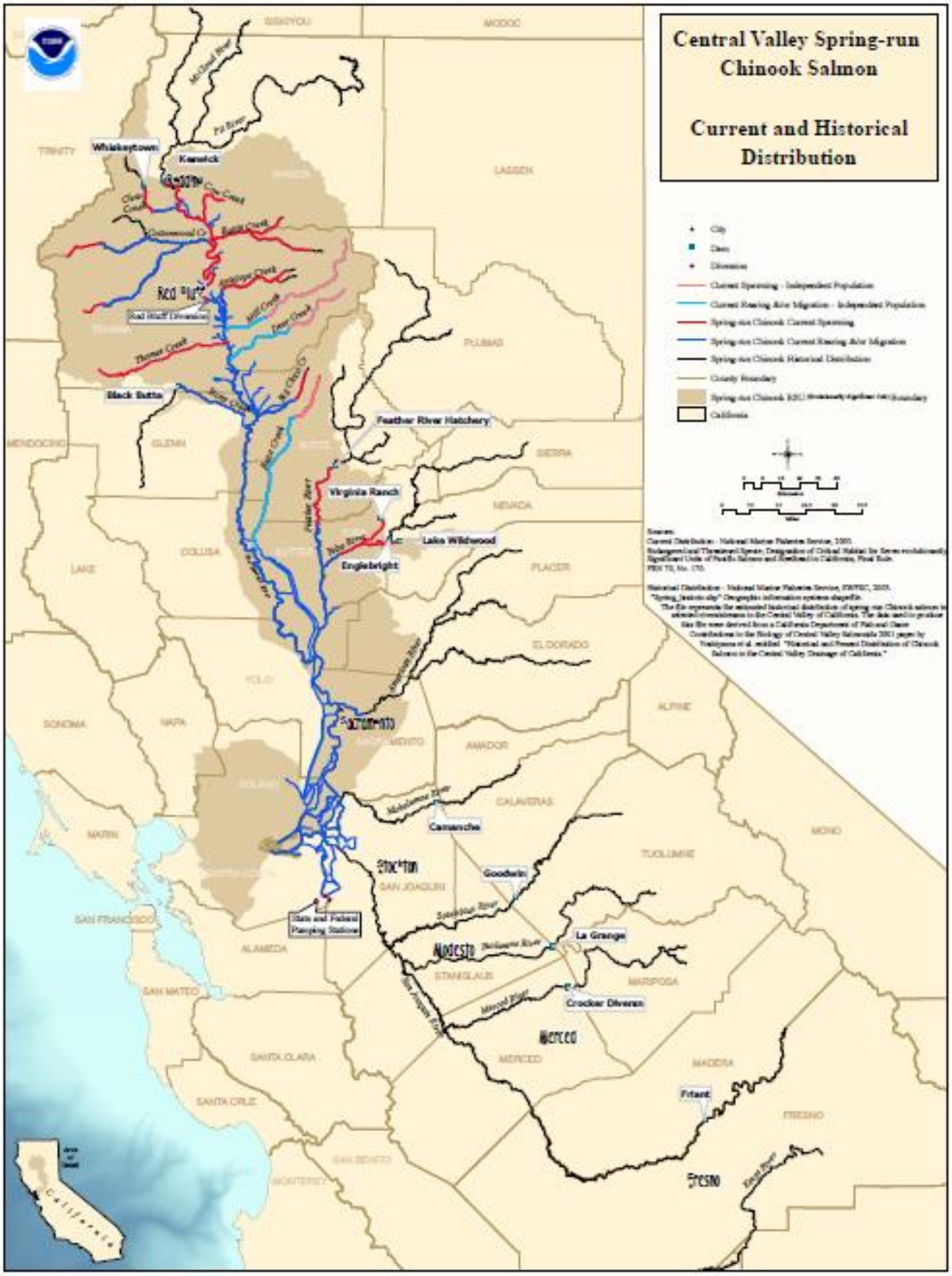
Spawning occurs in gravel substrate in relatively fast-moving, moderately shallow riffles or along banks with relatively high water, which promotes higher oxygen levels and reduced deposition of fines. Adult spawning conditions, incubation, and emergence from gravel is dependent on cold water temperatures (Myrick and Cech 2004). Fry emerge from gravels from November to March (Williams 2006). Post-emergent fry inhabit calm, shallow waters with fine substrates; fry depend on fallen trees, undercut banks, and overhanging riparian vegetation for refuge (Healey 1991).

Juvenile spring-run Chinook salmon can have highly variable outmigration timing based on various environmental factors (National Marine Fisheries Service 2009). Some juveniles begin outmigrating soon after emergence from gravel, whereas others oversummer and outmigrate as yearlings with the onset of intense fall storms (California Department of Fish and Game 1998). The outmigration period for spring-run Chinook salmon can extend from November to early May (National Marine Fisheries Service 2009:94) or June (California Department of Fish and Game 1998:III-9), with residency in the Delta probably lessening as the season progresses into the late spring months (California Department of Fish and Game 1998:III-9). Peak movement of yearling spring-run Chinook salmon occurs October–December (Goertler et al. 2020:3).

Juveniles prefer stream margin habitats with enough depth and velocities to provide suitable cover and foraging opportunities during rearing and downstream movement. Off-channel areas and floodplains can provide important rearing habitat. A greater availability of prey and favorable rearing conditions in floodplains increases juvenile growth rates compared with conditions in the mainstem Sacramento River, which can lead to improved survival rates during both their migration through the Delta and later in the marine environment (Sommer et al. 2001).

2.4.3 Distribution and Abundance

Spring-run Chinook salmon were historically the dominant run of salmon 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 (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 2-8). 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).



Source: National Marine Fisheries Service 2014:32.

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.4-1. Current and Historical Spring-Run Chinook Salmon Distribution

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). 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 has displayed broad fluctuations in adult abundance. Estimates of spring-run Chinook salmon in the Sacramento River and its tributaries have ranged from 1,105 in 2017 to 25,890 in 1982. This estimate does not include in-river or hatchery spawners in the lower Yuba and Feather Rivers because CDFW's GrandTab does not distinguish between fall-run Chinook salmon and spring-run Chinook salmon in these rivers prior to 2005.[8]

Since 1995, spring-run Chinook salmon annual run size estimates typically have been dominated by Butte Creek returns. Of the three tributaries producing naturally spawned spring-run Chinook salmon (Mill, Deer, and Butte Creeks), Butte Creek has produced an average of two-thirds of the total production over the past 10 years (Azat 2021). During recent years, spring-run Chinook salmon escapement estimates (excluding in-river spawners in the Yuba and Feather Rivers) have ranged from 23,810 in 2013 to 1,591 in 2017 throughout the tributaries to the Sacramento River surveyed.

Recent spring-run Chinook salmon population estimates have fluctuated but generally remain low. In-river escapement was estimated to range between 1,059 and 16,189 fish during 2016–2020, with 1,688 estimated in 2020 (Figure 2.4-2). During these years, escapement to hatcheries ranged from 532 (2017) to 3,867 (2019) (Figure 2.4-2). The 2022 escapement estimates were 5,132 for in-river spring-run Chinook salmon and 1,772 for hatchery spring-run Chinook salmon (Figure 2.4-2). In addition, fish monitoring is conducted throughout the year at the TFCF and the Skinner Fish Facility. Based on length-at-date criteria, during water year 2017, 26,551 wild (non-fin-clipped) juvenile spring-run and 963 hatchery (fin-clipped) spring-run Chinook salmon were estimated to have been salvaged at the TFCF and Skinner Fish Facility, and 9,487 wild juvenile spring-run and 1,010 hatchery spring-run were estimated to have been salvaged during water year 2018. Note, however, that length-at-date criteria for spring-run Chinook salmon are particularly prone to error because of the high overlap in lengths with the more abundant fall-run Chinook salmon, suggesting that actual spring-run entrainment is considerably lower than length-at-date criteria (Harvey et al. 2014). Recent data at the south Delta salvage facilities illustrate the extent of the errors: only two of 620 juvenile Chinook salmon examined in 2019, three of 750 juvenile Chinook salmon examined in 2020, and one of 99 juvenile Chinook salmon examined in 2021 that were classified as spring-run based on length-at-date criteria were confirmed to be genetic spring-run (the remainder were nearly all

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[8] The Feather River Hatchery implemented a methodology change in 2005 for distinguishing spring-run from fall-run. Fish arriving prior to the spring-run spawning period were tagged and returned to the river. The spring-run escapement was the number of these tagged fish that subsequently returned to the hatchery during the spring-run spawning period.

genetic fall-run; Reece pers. comm.).[9] No spring-run Chinook salmon have been collected in CCWD's Fish Monitoring Program at the Rock Slough Intake since 2008. The most recent (2020) 5-year viability assessment by NMFS concluded that there was high extinction risk for spring-run Chinook salmon populations from Mill Creek, Deer Creek, Battle Creek, Clear Creek, and the Feather River Hatchery (Johnson et al. 2023:172). These populations were all assessed to have had moderate risk of extinction at the time of the previous viability assessment in 2015 and high risk of extinction at the time of the prior (2010) assessment, except the Feather River Hatchery population, which was assessed to have high risk of extinction in 2010, 2015, and 2020. The Butte Creek population was concluded to have low risk of extinction in the 2010, 2015, and 2020 assessments (Johnson et al. 2023:172).

Spring-run Chinook salmon are being reintroduced to the San Joaquin River as part of restoration efforts. These fish originate from Feather River broodstock (San Joaquin River Restoration Program 2018:2-3). In 2013, NMFS designated the (Central Valley) spring-run Chinook salmon reintroduced to the San Joaquin River as an experimental nonessential population in accordance with Section 10(j) of the ESA (78 FR 79622). This allows for the release of threatened California spring-run Chinook salmon outside their current range. The population is considered an experimental, nonessential population (meaning that if the population does not survive, it will not threaten the whole species' population) because the population is geographically separate from the protected population of the same species (National Oceanic and Atmospheric Administration Fisheries 2020b). NMFS accounts for south Delta salvage of these reintroduced juveniles (adipose clipped and coded wire tagged) so that the reintroduction does not impose more than *de minimis* effects on water users (e.g., Strange 2020), although any naturally produced juveniles are not part of this accounting. Monitoring occurs for juvenile spring-run Chinook salmon released into the San Joaquin River because all fish are marked with adipose clips and coded wire tags (National Marine Fisheries Service 2022:4), allowing detection in monitoring programs such as the south Delta export salvage facilities. Adults are monitored using techniques such as video monitoring, traps, carcass surveys, and redd surveys (National Marine Fisheries Service 2022:7). 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 Feather River hatchery (fall- or spring-run) or spring-run Chinook salmon produced in the Sacramento River Basin (National Marine Fisheries Service 2019:7).

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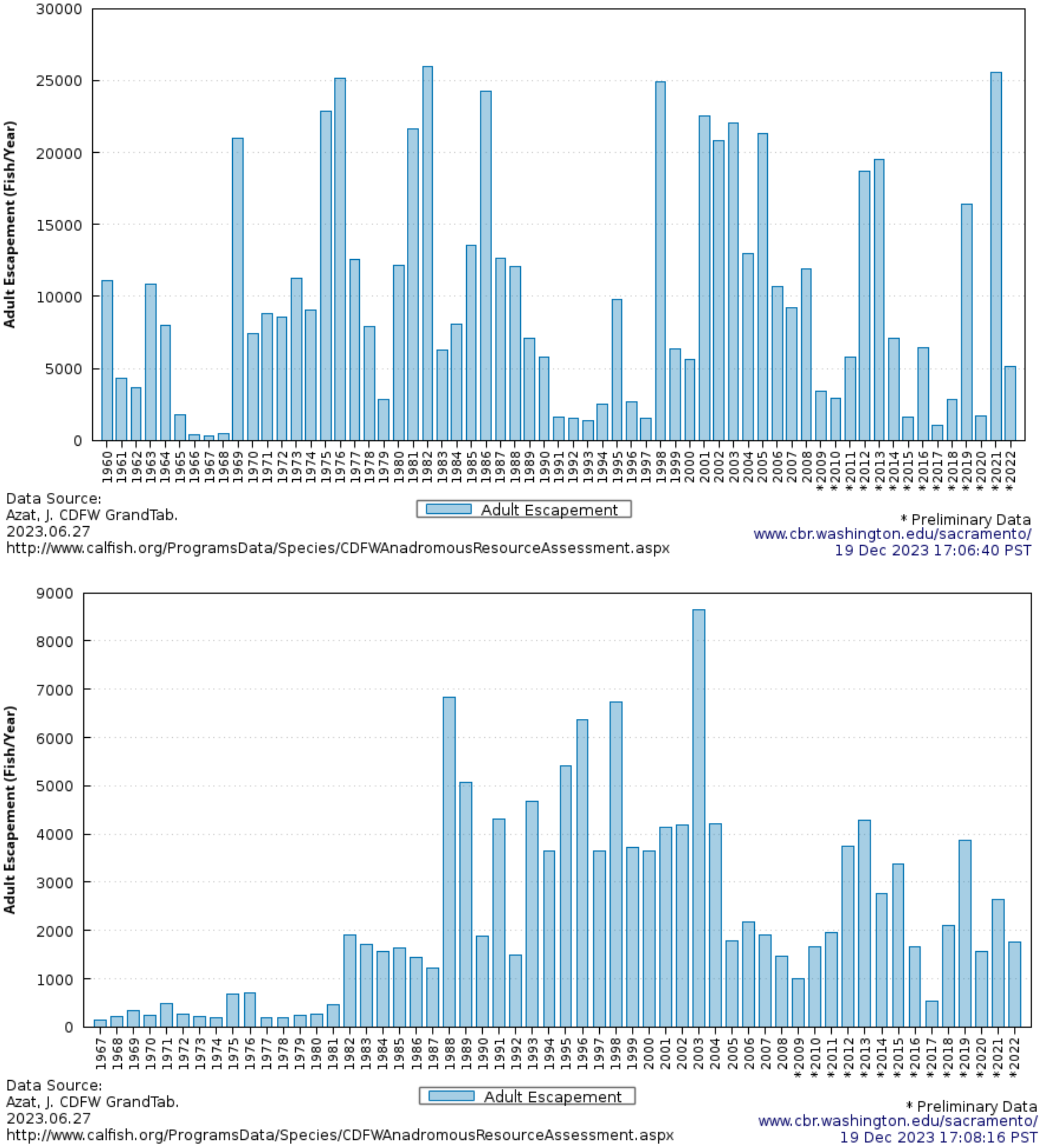
[9] To address such errors, DWR is developing a Chinook salmon run identification program to identify spring-run individuals at key monitoring locations throughout their known range within the Central Valley (Boro et al. 2023:1).



Source: Columbia Basin Research, University of Washington 2023.

Note: Vertical axis scale differs between upper and lower panel. See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.4-2. Spring-Run Chinook Salmon Adult In-River (Upper) and Hatchery (Lower) Annual Escapement in the Central Valley, 1970–2022

2.4.4 Species Threats

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 (Section 2.3, *Chinook Salmon—Winter-Run*). Among the most important stressors are agricultural diversions, diversion dams, andor weirs on Deer, Mill, Antelope, and Butte Creeks impeding or blocking access to upstream spawning habitat, as well as entrainment in Antelope Creek resulting from terminal diversions and loss of channel connectivity (National Marine Fisheries Service 2014:44–45). In Deer, Mill, and Antelope Creeks, losses of suitable spawning gravel, the development of deep channels and levees, pollutants and siltation from urban development, mining, and water diversions are also stressors on spring-run Chinook salmon Central Valley (National Marine Fisheries Service 2014).

The degradation and simplification of aquatic habitat in the Central Valley have reduced the resiliency of spring-run Chinook salmon to respond to additional stressors such as an extended drought, ocean harvest, and poor ocean conditions. Levee construction and maintenance projects have simplified riverine habitat and have disconnected rivers from the floodplain (National Marine Fisheries Service 2016b).Spring-run Chinook salmon 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 mortality to juvenile salmonids when they are entrained toward the export facilities (Larry Walker Associates 2010).

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 (Grimaldo et al. 2009). When Sacramento River Basin-origin fish route into the interior Delta via Georgiana Slough or the Delta Cross Channel (DCC) and enter the south Delta, entrainment from reverse flows in Old and Middle River 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).

Flow in the south Delta tends to be more complex than in the north Delta because of the influence of radial gate operations at the head of Clifton Court Forebay and the influence of exports on OMR dynamics, as described above. This is further complicated by the presence of temporary barriers,[10] lower inflow from the San Joaquin River, and greater tidal excursion. Highly channelized levee characteristics maintained for water conveyance diminish the potential for the Delta to function as rearing habitat for juvenile salmonids.

As discussed for winter-run Chinook salmon, juveniles have access to floodplain habitat in the Yolo Bypass only during moderate- to high water years, and the quantity of floodplain available for rearing during drought years is currently limited; however, the notching of Fremont Weir as part of the Yolo Bypass Salmonid Habitat Restoration and Fish Passage Implementation Plan will provide access to floodplain habitat for juvenile salmon over a longer period (California Department of Water Resources and Bureau of Reclamation 2012:3). Juvenile spring-run Chinook salmon may also

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[10] The South Delta Temporary Barriers Project located at the Central Valley Project's Jones Pumping Plant and State Water Project's Banks Pumping Plant in the South Delta.

rear in the Sutter Bypass floodplain, with Butte Creek populations able to access the Sutter Bypass because of its connection to lower Butte Creek, whereas other populations and other races of Chinook salmon are only able to access the floodplain during large Sacramento River flooding events (Cordoleani et al. 2020).

Recent work by the Collaborative Adaptive Management Team's (CAMT) Salmonid Scoping Team (SST) suggests that high correlations between inflows and exports make it difficult to evaluate their effects on salmon survival independently using statistical methods (Buchanan et al. 2018). There are very few observations of salmon survival at high export rates, which makes it difficult to determine if there is a relationship, but most acoustic tagging studies show support for a positive relationship between flow and survival (Perry et al. 2010, 2018; Michel et al. 2012). A key conclusion of the SST (2017:ES-5–ES-7) is that water export operations contribute to salmonid mortality in the Delta via direct mortality at the facilities, but direct mortality does not account for the majority of the mortality experienced in the Delta; the mechanism and magnitude of indirect effects of water project operations on Delta mortality outside the facilities is uncertain. Temporary barriers are installed by DWR in the south Delta for the purpose of stabilizing and increasing water surface elevations to facilitate agriculture irrigation. A temporary barrier at the Head of Old River has previously been installed to reduce movement of migrating salmonids into Old River, which would include the experimentally reintroduced spring-run Chinook salmon as part of the San Joaquin River Restoration Program. Conceptual models identified by the CAMT's SST predict that survival to Chipps Island will be higher with the barriers in place by forcing salmon to avoid the interior Delta. Changes in flows resulting from the barrier installation are also expected to benefit salmonid route selection and migration rates, although localized predation around the barriers themselves is expected to increase. Recent investigations of other juvenile salmonids have provided some support that the barrier may positively affect survival (steelhead; Buchanan et al. 2021), whereas another study did not find a statistically significant relationship between survival from the Head of Old River to Chipps Island and barrier presence (fall-run Chinook salmon; Buchanan and Skalski 2020).

Results of Chinook salmon survival and migration studies in the Sacramento and San Joaquin Rivers and Delta suggest that relationships between river flow and migration rates are more complicated than in the Pacific Northwest, where flow is more unidirectional (Zabel et al. 1998; Smith et al. 2002). Higher survival of acoustically tagged wild-origin spring-run Chinook salmon smolts from Mill Creek through the Sacramento River was observed in a wet year (2017) compared to historic drought conditions in 2015 (Notch et al. 2020). Cordoleani et al. (2018, 2019) found higher survival for wild-origin smolts from Butte Creek to the Golden Gate Bridge (spring-run and fall-run) in 2016 than 2015 correlated with greater flow in 2016 than 2015, and they found higher survival in 2017 than the prior two years correlated with greater flow in 2017 than those years. In the 2019 study, Chinook salmon smolts were tracked from the Sutter Bypass to the Golden Gate Bridge, Cordoleani et al. (2019:1) summarized their results to note that release date and Delta flow were significantly correlated with survival rates and that the results were largely driven by 2017 data, for which fish were released a month later than those in 2015 and 2016, and Delta flow and smolt survival were significantly higher than in the previous 2 years. They also noted that more tagging years including measurements of more potentially important environmental factors (such as turbidity) are required to robustly identify the influence of various factors on Butte Creek spring-run Chinook outmigrant smolt survival (Cordoleani et al. 2019:1).

As previously described in the winter-run Chinook salmon assessment above, routing down Georgiana Slough has also been shown to increase when unidirectional flow gives way to tidal influences and flow becomes more bidirectional, particularly below 20,000 cfs at Freeport (Perry et al. 2018). There is a positive correlation between through-Delta survival of large acoustically tagged late fall–run Chinook salmon and Sacramento River flow entering the Delta (Perry et al. 2018).

Historically, wherever spring-run Chinook salmon and fall-run Chinook salmon populations overlapped, they were naturally temporally segregated and genetic integrity was maintained. However, because of difficulties associated with holding adults over the summer in the Feather River Hatchery, fish were left in the river until spawning, which presumably led to mixing with fall-run Chinook salmon in the hatchery (Williams 2006:33). 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).Climate change may pose similar threats to spring-run Chinook salmon as were described for winter-run Chinook salmon, with increasingly high water temperatures and changes to ocean conditions being limiting factors. Like winter-run Chinook salmon, spring-run Chinook salmon are particularly vulnerable to these limiting factors because their life history is adapted to streams with snowmelt runoff, with relatively dependable, sustained high flows that allow fish to ascend to high enough elevations where water temperatures remain tolerably cool through the summer. Snowmelt runoff is relatively more important in the San Joaquin River and its major tributaries, where historically spring-run Chinook salmon were more abundant. Recoveries of coded wire tags and genetic samples suggest that spring-run Chinook salmon have a more northerly ocean distribution and mature later than winter-run Chinook salmon (Satterthwaite et al. 2018). Therefore, climate-induced changes in ocean prey distributions that limit access to coastal prey may disproportionately affect spring-run Chinook salmon that rely on marine resources to a greater degree in order to mature.

General life stage timing for spring-run Chinook salmon is summarized in Tables 2.4-1 and 2.4-2, with a quantitative summary for juveniles in Tables 2.4-3 and 2.4-4. Appendix 12A, Attachment 12A.1, *Juvenile Salmonid Monitoring, Sampling, and Salvage Timing Summary*, of the Final EIR, provides additional summary information from the SacPAS database (California Department of Water Resources 2023).

Table 2.4-1. 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**	
Sac. 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
Sac. 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

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

Sources: National Marine Fisheries Service 2019:83. Note: The patterns are generalized and reflect sampling periods, which can vary (e.g., Knights Landing sampling with rotary screw traps has not always covered August and September; see annual summaries in California Department of Water Resources 2023: Appendix 12A, Attachment 12A.1). 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 2.4-2. Generalized Temporal Occurrence of Spring-Run Chinook Salmon by Life Stage in the Delta

Life Stage	**Month**											
	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

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

Note: Table reflects monitoring based on length-at-date classification of juvenile spring-run Chinook Salmon. See Tables 2.4-3 and 2.4-4 for quantitative summary. * The data in this category reflects juveniles entrained into the salvage facilities. See also annual summaries in California Department of Water Resources 2023: Appendix 12A, Attachment 12A.1, *Juvenile Salmonid Monitoring, Sampling, and Salvage Timing Summary from SacPAS*).

Source: National Marine Fisheries Service 2019:84.

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Table 2.4-3. 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% (588)
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)	3.8% (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 Trawl at Sherwood Harbor	1/3/2000–7/29/2022	Trawl tows	4.7% (3,42)	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 Trawl at Chipps Island	1/2/2000–7/29/2022	Trawl tows	0% (4,4425)	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% (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% (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 2.4-4. 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% (588)
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% (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 Trawl at Sherwood Harbor	1/3/2000–7/29/2022	Trawl tows	0% (3,402)	1.1% (3,273)	4.9% (3,524)	16.9% (3,52)	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 Trawl 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% (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.

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2.5 White Sturgeon

2.5.1 Legal Status

White sturgeon (*Acipenser transmontanus*) 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). Overall, information on trends in adults and juveniles suggests that numbers are declining (Moyle 2002; Moyle et al. 2015:4–5). As of November 29, 2023, the species has been proposed for listing as threatened under the CESA (California Department of Fish and Wildlife 2023a).

2.5.2 Life History and General Ecology

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 December to July, but typically February to early June (McCabe and Tracy 1994; Schaffter 1997; California Department of Fish and Wildlife 2023a). Miller et al. (2020) detected the greatest number of adult white sturgeon in their middle Sacramento River spawning reach during February through April, although individuals were present from October to June. After absorbing yolk sacs and initiating feeding, Young-of-the-year 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 are iteroparous; a small proportion of adults spawn in any given year. Reproduction occurs episodically, when spring and summer river flows are high enough to support incubation and early rearing. In the San Francisco Bay and Delta, females generally mature reproductively between ages 12 and 16 (as early as 10 and as late as 19), while males mature generally between 10 and 12 years of age and appear to spawn more frequently than females. Males may spawn every 1 to 2 years, while females can only reproduce every 2 to 3 years but typically wait at least 2 to 4 years between reproductive events in favorable conditions (California Department of Fish and Wildlife 2023a).

Eggs are negatively buoyant and become adhesive upon fertilization; embryonic development is rapid and temperature-dependent, ranging from 3 to 13 days in the white sturgeon population. Optimal egg incubation occurs between 14°C and 17°C; mortality is nearly complete at temperatures less than 8°C and greater than 20°C (California Department of Fish and Wildlife 2023a). Among California white sturgeon, at temperatures between 14°C and 17°C, the yolk sac is completely absorbed approximately 20 to 23 days post-fertilization (Wang 2010). Larvae are photonegative upon hatching and swim near the bottom of rivers; the presence of physical cover in well-lit mesocosms decreased predation on white sturgeon larvae less than 17 mm TL (California Department of Fish and Wildlife 2023a).

Recruitment of juvenile white sturgeon is positively correlated with high river flows and Delta outflow during spring and early summer months (California Department of Fish and Wildlife 2023a). The mechanism underlying the relationship between high river flows and California white sturgeon recruitment has been attributed to improved survival and transport of larval sturgeon into suitable rearing areas, increases in the number of females spawning during high flow periods, or both, and it is also possible that high river flows improve spawning habitat by cleaning fine sediment out of gravel and cobble spawning substrates (Hildebrand et al. 2016; California Department of Fish and Wildlife 2023a).

Juvenile sturgeon actively swim downstream toward the estuary, suggesting that their capacity to osmoregulate in brackish environments develops as larvae mature into juvenile fish. In the Central Valley, California white sturgeon spawning has been detected during wet and dry years in both the San Joaquin River and the Sacramento River, indicating that adults will attempt to spawn even when flows are low (Jackson et al. 2016). The fact that juvenile recruitment appears to be successful only in years when elevated river flows occur during larval dispersal and early juvenile rearing suggests that flows during the spring and early summer are essential for sustaining populations (California Department of Fish and Wildlife 2023a).

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 San Francisco Bay and Napa and Petaluma Rivers and Sonoma Creek in the north (San Pablo 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.

2.5.3 Distribution and Abundance

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). The current and historical distribution is shown in Figure 2.5-1. 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). 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 through 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). Young-of-the-year 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 in 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, 2012; California Department of Fish and Wildlife 2013, 2014) documented that anglers upstream of State Route (SR) 140 annually caught between 8 and 25 mature white sturgeon between 2007 and 2013. Below SR 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 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. 2017). From 1995 through 2019, white sturgeon larvae were collected in the 20 mm Survey in 89% of wet, 67% of above normal, and 40% of below normal water year types (Table 2.5-1). No white sturgeon larvae were collected in any critical or dry water years (California Department of Fish and Wildlife 2020a), consistent with previous observations from Stevens and Miller (1970).

Table 2.5-1. Annual Number of White Sturgeon Larvae Collected in the 20-mm Survey, 1995–2019

Year	Water Year Type	Number of White Sturgeon Larvae Collected
1995	W	4
1996	W	1
1997	W	0
1998	W	81
1999	W	7
2000	AN	16
2001	D	0
2002	D	0
2003	AN	2
2004	BN	0
2005	AN	0
2006	W	15
2007	D	0
2008	C	0
2009	D	0
2010	BN	0
2011	W	8
2012	BN	0
2013	D	0
2014	C	0
2015	C	0
2016	BN	137
2017	W	73
2018	BN	6
2019	W	34

Source: California Department of Fish and Wildlife 2020a.

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. Currently almost all sturgeon found in the Klamath River basin above the estuary are green sturgeon (Moyle 2002).

Source: California Department of Fish and Wildlife 2023a.

Note: See Chapter 10, *Description of Figures*, for a description of this figure.

Figure 2.5-1. Current and Historical White Sturgeon Distribution

2.5.4 Species Threats

Numerous factors likely affect the white sturgeon population in the Delta. Survival during early life history stages may be adversely affected by insufficient flows, lack of rearing habitat, predation, warm water temperatures, decreased dissolved oxygen, 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%) 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%) (Blackburn et al. 2019). Like green sturgeon, there have historically been substantial passage problems for white sturgeon such as 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). 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 (i.e., selenium, mercury, cadmium, arsenic, and copper) at levels known to impair fish health and likely negative 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. 2019).

2.6 California Tiger Salamander

2.6.1 Geographic Distribution and Status

The Central California DPS of California tiger salamander is listed as threatened (69 FR 47212) under the federal ESA on August 4, 2004 and listed as threatened under the CESA in 2020 (Cal. Code Regs. tit. 14, § 670.5, subd. (b)(3)(G), California Department of Fish and Wildlife 2020b:39). A recovery plan for the Central California DPS of the California tiger salamander was published June 6, 2017 (U.S. Fish and Wildlife Service 2017a:I-2, I-6–I-7). The most recent 5-Year Review for the species was published on August 10, 2023 (U.S. Fish and Wildlife Service 2023). Isolated populations in Sonoma and Santa Barbara Counties are listed as endangered under the ESA and threatened under the CESA.

California tiger salamander historically occurred throughout the Central Valley and surrounding foothills, from Yolo County south to Tulare County, and in the south coast ranges from north of Monterey Bay to San Luis Obispo County, although many of the populations in the Central Valley are now extirpated. Currently, the Central California DPS of this species is distributed along the foothills of the Central Valley and Inner Coast Range from Sacramento and Yolo counties in the north, to San

Luis Obispo, Kern, and Tulare counties in the south (U.S. Fish and Wildlife Service 2017a:I-2, I-6–I-7).

California tiger salamander has a potential to occur in the study area generally in areas west of the Yolo Basin but including the Tule Ranch Unit of the Yolo Basin Wildlife Area, which includes western Yolo County and Solano County, and portions of eastern Contra Costa and Alameda Counties, and western portions of San Joaquin County.

There are numerous extant California tiger salamander CNDDB occurrences west of the Clifton Court Forebay area and several occurrences in the Jepson Prairie portion of the study area (California Department of Fish and Wildlife 2020c).

2.6.2 Life History and Habitat Requirements

California tiger salamander is found in annual grassland, vernal pool complexes, open mixed woodland and oak savanna communities in lowland and foothill regions of central California where suitable aquatic sites, such as vernal pools, seasonal ponds or constructed ponds, are available for breeding (U.S. Fish and Wildlife Service 2009). The species is typically found at elevations from sea level to 2,000 feet, although the known elevational range extends up to 3,940 feet (U.S. Fish and Wildlife Service 2017a:I-2).

The suitability of California tiger salamander habitat is proportional to the abundance of upland refuge sites near aquatic breeding sites. Adult California tiger salamanders are terrestrial and spend much of the year in the underground burrows of small mammals, such as California ground squirrels (*Spermophilus beecheyi*) (Loredo et al. 1996:283; Trenham 2001:343–344) and Botta's pocket gopher (*Thomomys bottae*) (Jennings 1996:194). Active rodent burrow systems are considered an important component of California tiger salamander upland habitat (Loredo et al. 1996:283), as inactive burrow systems begin to deteriorate and collapse over time. Therefore, active ground-burrowing rodent populations are likely needed to sustain California tiger salamander populations. California tiger salamander is known to move up to 1.3 miles into upland habitat from aquatic habitat (U.S Fish and Wildlife Service 2017a:I-4; Orloff 2007:26).

Historically, vernal pools and other seasonal rain pools were the primary breeding habitat of California tiger salamander (Barry and Shaffer 1994:159); however, the species is now known to also successfully reproduce in ephemeral and permanent man-made ponds (U.S. Fish and Wildlife Service 2017a:I-5). In the East Bay Regional Park District in Contra Costa and Alameda Counties, California tiger salamanders breed almost exclusively in seasonal and perennial stock ponds (Bobzien and DiDonato 2007:7). The presence of predatory fish and bullfrogs (*Rana catesbeiana*) can affect the habitat suitability of perennial ponds, making them less suitable than ephemeral ponds. Barry and Shaffer (1994:163) note that annual draining can prevent predatory species from establishing. The species is not known to breed in streams or rivers; however, breeding has been documented in ditches that contain seasonal wetland habitat and in slow-moving swales and creeks near other suitable breeding habitat (U.S. Fish and Wildlife Service 2017a:I-5).

Adult California tiger salamanders migrate from upland refuge sites to breeding pools during relatively warm late fall and early rains (Thomson et al. 2016), usually from November through April. Breeding generally occurs from December through March (Stebbins 2003). Eggs are laid individually or in clumps on submerged vegetation and debris in shallow water and generally hatch in 10 to 28 days (Jennings and Hayes 1994:12; U.S. Fish and Wildlife Service 2017a:I-3). Development through metamorphosis requires 3 to 6 months, beginning late spring or early summer. Post-metamorphic juveniles disperse from breeding sites at night during the late spring or early summer to upland burrows or soil crevices (U.S. Fish and Wildlife Service 2017a:I-3).

The proximity of refuge sites to aquatic breeding sites also affects the suitability of salamander habitat. Based on capture data from a single-season study at Olcott Lake in Jepson Prairie Preserve (Solano County), Shaffer and Trenham (2005:1163) estimated that 95% of adult and subadult tiger salamanders occurred within approximately 0.4 mile of the breeding pond. Their model also suggests that 85% of subadults were concentrated between 0.1 and 0.4 mile from the pond.

2.6.3 Species Threats

Residential and commercial development, road construction, and agricultural projects are considered the most significant threat to California tiger salamander (U.S. Fish and Wildlife Service 2017a:I-2, I-6–I-7). Approximately 75% of the species' historical natural habitat has been lost. The species has been eliminated from 55% to 58% of historical breeding sites. Holland (1998) indicated that about 75% of the historical vernal-pool breeding habitat has been lost, although some question the reliability of this estimate.

Development, agriculture, energy (e.g., Mariposa Energy), and water projects (e.g., CVP, SWP, South Bay Aqueduct), and associated roads, lead to habitat fragmentation in the study area. Fragmentation of habitat leads to increasingly isolated populations with no mechanism for inter-pond dispersal, thus reducing the long-term viability of local populations. Additionally, roads often bisect habitat, creating a barrier to dispersal and leading to direct mortality of migrating salamanders (Twitty 1941:1; Barry and Shaffer 1994:162–163; Jochimsen et al. 2004:19–21). Construction activities such as grading or deep ripping also contribute to direct mortality (U.S. Fish and Wildlife Service 2017a:I-2, I-6–I-7). Adjacent to the Bethany Reservoir are the Bethany and Bethany West Conservation Easements, which protect grassland habitat in the action area from further development and fragmentation. These easements were purchased by DWR and are managed by CDFW to offset the effects of the South Bay Aqueduct Improvement and Enlargement Project on California tiger salamander and California red-legged frog.

Nonnative, invasive species, such as bullfrog, mosquitofish (*Gambusia affinis*), and various centrarchids that live in perennial ponds (e.g., stock ponds) are considered to have negatively affected California tiger salamander populations by preying on larval salamanders (Fisher and Shaffer 1996; U.S. Fish and Wildlife Service 2017a:I-2, I-6–I-7). Hybridization with the barred tiger salamander (*Ambystoma tigrinum mavortium*) is also a threat to this species (U.S. Fish and Wildlife Service 2023:5). It has been suggested that nonnative alleles may be present in California tiger salamander populations found in the Altamont Hills near the study area, therefore, hybridization may be a threat in this region (U.S. Fish and Wildlife Service 2017a:I-2, I-6–I-7).

Pesticides, hydrocarbons, and other pollutants are all thought to negatively affect breeding habitat. Other sources of chemical pollution that may negatively affect California tiger salamander include the application of chemicals from agricultural production, landscape maintenance, and rodent and vector control programs (U.S. Fish and Wildlife Service 2017a). California ground squirrel and pocket gopher control operations may have the indirect effect of reducing the availability of upland burrows for use by California tiger salamanders (Loredo-Prendeville et al. 1994:74). Death by entombment could also result when burrow entrances are crushed during rodent control activities.

2.6.4 Suitable Habitat Definition

As described above in Section 2.6.2, *Life History and Habitat Requirements*, suitable habitat for California tiger salamander includes aquatic habitat consisting of vernal pools, other seasonal pools, and ponds that inundate for at least 3 months and upland habitat consisting of adjacent annual grassland, including alkali grasslands, with small mammal burrows for refugia within at least 1.24 miles of aquatic habitat. This definition is consistent with the USFWS's 2003 *Interim Guidance on Site Assessment and Field Surveys for Determining Presence or a Negative Finding of the California Tiger Salamander*. The areas of suitable habitat in the study area are limited to those areas described in Section 2.6.5, *Habitat Suitability Model*.

2.6.5 Habitat Suitability Model

The habitat model for California tiger salamander includes both aquatic and upland habitats. The modeled aquatic habitat relies on both delineation data that were collected for a smaller portion of the study area, in what is called the delineation study area, and suitable habitats found in the Great Valley Ecoregion 2018 Vegetation Dataset, the Delta Vegetation and Land Use Update, and the Great Valley Vernal Pool Data.

The modeled upland habitat is limited to areas within 1.24 miles of suitable aquatic habitat, based on USFWS' 2003 *Interim Guidance on Site Assessment and Field Surveys for Determining Presence or a Negative Finding of the California Tiger Salamander* (U.S. Fish and Wildlife Service 2003:4). For areas along the western edge of the study area, upland habitat was also identified by reviewing aerial photographs for aquatic habitat outside of, but within 1.24 miles of, the study area. The extent of modeled habitat in the study area is depicted in Chapter 4, *Analysis of Take and Effects*, Figures 4.6-1 through 4.6-16.

Based on the known range and occurrences described in the section above, the California tiger salamander habitat model is constrained to the following portions of the study area.

- West of the Yolo Basin but including the Tule Ranch Unit of the CDFW Yolo Basin Wildlife Area.
- Within the Contra Costa, Alameda, and San Joaquin Counties portions of the study area starting near Antioch, then extending south on SR 4 to Balfour Road, then east on Balfour Road to Byron Highway, then south Byron Highway to SR 4, then east to the western bank of Old River, then south along the western bank to Old River's confluence with Italian Slough, then continue south along the western bank of Italian Slough to where Italian Slough turns to the west at which point the geographic limits cross Italian slough and continue south along the western edge of Clifton Court Forebay and the start of the California Aqueduct until Byron Highway at which point the limits continue southeast on Bryon Highway to I-205 and then west on I-205 to the edge of the study area.
- In Sacramento County south of the Cosumnes River and east of Interstate (I-) 5.

The rationale for including each of the above locations in the habitat model is provided below.

- There is a cluster of extant occurrences at Jepson Prairie in Solano County, which increases the potential for occurrence on suitable habitat in the Cache Slough region and Yolo Basin.
- The areas within the study area surrounding Clifton Court Forebay are considered potentially suitable and were included in the species model because there is suitable grassland/vernal pool

habitat within the accepted movements distance (1.24 miles) of known, extant occurrences west of Byron Highway and Clifton Court Forebay (California Department of Fish and Wildlife 2020c).

- In San Joaquin County, there is a CNDDB record just south of SR 120 near Manteca in the study area (California Department of Fish and Wildlife 2020c).
- The location in Sacramento County south of the Cosumnes River is included because it is on the edge of the study area and within the range of the species (although there are no records in this portion of the legal Delta).

Information on species habitat model development is included in Appendix 4B, *Terrestrial Take Analysis Methods*, Section 4B.5.1.1, *Suitable Habitat Models*.

2.6.5.1 Aquatic Breeding Habitat Model

Inside the Delineation Study Area

Modeled habitat includes the following types from the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021).

- Vernal pool complex
  - Vernal pool

Outside the Delineation Study Area

Modeled habitat includes the following types from the Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), and the Great Valley Vernal Pool Data (Witham et al. 2014).

- Vernal pool complex
  - *Distichlis spicata*
  - California annual herb/grass group
  - Californian mixed annual/perennial freshwater vernal pool/swale bottomland
  - Mediterranean California naturalized annual and perennial grassland

2.6.5.2 Upland Habitat Model

Inside the Delineation Study Area

Modeled habitat includes the following types from the Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), and the Great Valley Vernal Pool Data (Witham et al. 2014).

- Vernal pool complex

  - All types
- Grassland
  - All types
- Alkaline seasonal wetland complex
  - *Distichlis spicata*

Outside the Delineation Study Area

Modeled habitat includes the following types from the Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), and the Great Valley Vernal Pool Data (Witham et al. 2014).

- Vernal pool complex
  - *Allenrolfea occidentalis*
  - *Frankenia salina*
  - *Suaeda moquinii*
  - Alkaline wetland
- Grassland
  - All types
- Alkaline seasonal wetland complex
  - *Distichlis spicata*

2.7 Giant Garter Snake

2.7.1 Geographic Distribution and Status

Giant garter snake was listed as threatened under the ESA on October 20, 1993 (58 FR 67046–67053) and listed as rare in 1971 and reclassified as threatened in 1985 under the CESA (Cal. Code Regs. tit. 14, § 670.5, subd. (b)(4)(E), California Department of Fish and Wildlife 2019). The *Draft Recovery Plan for the Giant Garter Snake* was completed in 1999, the most recent 5-year review was completed in 2020, and a revised recovery plan was released in 2015 (U.S. Fish and Wildlife Service 2020b:1-3). The final *Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*)* was published in 2017 (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8).

Giant garter snake historically occurred throughout the Central Valley of California. Its current range has been reduced to fragmented populations from Glenn County to the edge of the Delta, and south from Merced to Fresno Counties. The current known distribution of giant garter snakes is variable and extends from near Chico in Butte County, south to the Mendota Wildlife Area in Fresno County (U.S. Fish and Wildlife Service 2020b:4). There are nine recognized distinct populations of giant garter snake, which correspond to recovery units identified in the 2017 recovery plan: the Butte, Colusa, American, Yolo, Cosumnes-Mokelumne, Delta, San Joaquin, and Tulare Basins (U.S. Fish and Wildlife Service 2017b:iii).

The entire study area falls within the range of giant garter snake where suitable habitat exists. The study area overlaps with the Delta Basin, Yolo Basin, and Cosumnes-Mokelumne Basin Recovery Units identified in the recovery plan (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8). Each of these basins includes an extant population of giant garter snake. Although giant garter snake may have occupied much of the Sacramento–San Joaquin Delta at one time, longstanding wetland reclamation projects for intense agricultural applications has eliminated most suitable habitat (Hansen 1986:14, 16–17) and prevented the reestablishment of viable giant garter snake breeding populations in the Delta, other than the three populations noted above. However, several recent occurrences of giant garter snake in the vicinity of Sherman and Twitchell Islands, Jersey Island, and Webb Tract (California Department of Fish and Wildlife 2020c) have USFWS and CDFW giant garter snake experts asking if these recent sightings in the central Delta may represent an extant population that lives in emergent vegetation along river edges.

There are numerous CNDDB occurrences for giant garter snake throughout the study area north of SR 4 (California Department of Fish and Wildlife 2020c). Giant garter snake was observed at the Shin Kee Tract wetland site adjacent to the White Slough Wildlife Area south of SR 12 (Zentner Planning and Ecology 2019). As of 2020, new CNDDB occurrences were reported from west of the Sacramento River Deep Water Ship Channel on Liberty Island , and in 2022, giant garter snake was reported on the south shore of the San Joaquin River, near Antioch Point (California Department of Fish and Wildlife 2024) and a snake was found at West False River Drought Barrier on Jersey Island (Wunderlich pers. comm.).

2.7.2 Life History and Habitat Requirements

Giant garter snake is a highly aquatic, diurnal snake, relying on the presence of water throughout the summer months, and associated upland with burrows, crevices, or riprap for use as refugia during the late fall through early spring. Suitable aquatic habitat consists of marshes, sloughs, rice fields, and other water bodies, including lacustrine and riverine habitats, with emergent vegetation for basking and camouflage and a suitable prey base of fish and amphibians. Due to loss or degradation of natural wetland habitat (Wylie et al. 1997:2), giant garter snakes now commonly use highly modified and degraded habitats including cultivated farmlands and water conveyance infrastructure (Hansen and Brode 1980:10; Brode 1988:27; U.S. Fish and Wildlife Service 2017b:iii, I-2). Giant garter snakes generally are not present in larger rivers and wetlands with sand, gravel, or rock substrates. In addition, the major rivers within the species' range have been highly channelized, removing oxbows and backwater areas that probably at one time provided suitable habitat. Riparian woodlands do not generally provide suitable habitat because most have excessive shade, lack of basking sites, and absence of prey populations. Giant garter snakes are also absent from most permanent waters that support established populations of predatory game fishes and from most sites that undergo routine dredging, mechanical or chemical weed control, or compaction of bank soils (Brode 1988:25–28; U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8).

Terrestrial habitat adjacent to suitable aquatic habitat is also an important resource for giant garter snake (Halstead et al. 2015:633). Terrestrial habitat serves two purposes for giant garter snake. Near aquatic habitat, upland can be used for thermoregulation and summer shelter in nearby burrows; further away from aquatic habitat, and above the high winter waters, the upland can provide refugia for brumation (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8). During the colder winter months (generally October 1 to April 1), giant garter snakes over-winter in upland areas that provide sufficient cover, which are usually mammal burrows and include human-made features such as riprap (U.S. Fish and Wildlife Service 2017b:I-3). They may over-winter as far as 650 to 820 feet from the edge of aquatic habitat (U.S. Fish and Wildlife Service 2017b:I-3). A study by Halstead et al. (2015:638) found that giant garter snakes spend more than half of their time in terrestrial habitat during the summer (U.S. Fish and Wildlife Service 2020b:17). Halstead et al. (2015) found the average snake to be within 98 feet of water with 95% of observations within 33 feet of water with their model predicting that some individuals could be as far as 571 feet from water (U.S. Fish and Wildlife Service 2020b:17; Halstead et al. 2015:639).

Giant garter snakes emerge from winter brumation in early March or April, depending upon weather, and remain active through late September or early October. Breeding occurs from shortly after emergence until as late as May, with females giving birth from July through September.

2.7.3 Species Threats

Habitat loss and fragmentation from development, flood control (e.g., Central Valley Flood Protection Plan), water projects (e.g., CVP and SWP), agriculture actions and changes in agricultural and land management practices (including changes to rice production methods), and predation from introduced and native species are the main causes for the decline of giant garter snake (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8). Conversion of Central Valley wetlands for agriculture and urban uses has resulted in the loss of as much as 95% of historical habitat for giant garter snake (Wylie et al. 1997:2). In areas where giant garter snake has adapted to agriculture, maintenance activities such as vegetation and rodent control, bankside grading or dredging, and changes to water use regimes threaten their survival (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8; Wylie et al. 2004:9). In developed areas, vehicular mortality continues to be a threat, but this is not considered to be significant.

Giant garter snakes are predated upon by nonnative, invasive species such as bullfrogs (Dickert 2003). In areas near urban development, giant garter snakes may also fall prey to domestic or feral cats. In permanent waterways, introduced predatory game fishes, such as bass (*Micropterus* spp.), sunfish (*Lepomis* spp.), and channel catfish (*Ictalurus* spp.), prey on giant garter snakes and compete with them for smaller prey. While predation continues to be a threat, it is not considered to be significant (U.S. Fish and Wildlife Service 2017b:I-12).

2.7.4 Suitable Habitat Definition

Suitable habitat is described by USFWS in the 2017 final recovery plan (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8), including the following components, which are quoted verbatim.

> **Aquatic Component.** The giant garter snake has been recognized as requiring aquatic habitat since it was first described and has been consistently observed and captured in association with aquatic habitats since accounts of the snake were first published (Fitch 1940) (; G. Hansen and Brode 1980). The aquatic component of the giant garter snake habitat has been regarded as necessary for the survival of the snake, and researchers acknowledge the following qualitative attributes of ideal aquatic habitat for the giant garter snake (G. Hansen 1986; G. Hansen and Brode 1980; Wylie *et al.* 1995; Dickert 2002; E. Hansen 2002):
>
> 1. Water present from March through November.

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

> **Upland Component.** Although the giant garter snake is predominately an aquatic species, incidental observations and radio telemetry studies have shown that the snake can be found in upland areas near the aquatic habitat component during the active spring and summer seasons (G. Hansen 1986, 1988; Brode and G. Hansen 1992; E. Hansen 2002; Dickert 2003; Wylie and Cassaza 2000, 2001; Wylie et al. 1995, 1997a, 2002a, 2003a, 2004, 2005). Upland habitat (land that is not typically inundated during the active season and is adjacent to the aquatic habitat of the giant garter snake) is used for basking to regulate body temperature, for cover, and as a retreat into mammal burrows and crevices in the soil during ecdysis (shedding of skin) or to avoid predation (G. Hansen and Brode 1993; Wylie *et al.* 2003). Giant garter snakes have been observed using burrows for refuge in the summer as much as 50 meters (164 feet) away from the marsh edge (Wylie *et al.* 1997). Important qualities of upland habitat have been found by researchers (E. Hansen 2003; Wylie *et al.* 2003). Important qualities of upland habitat have been found by researchers (E. Hansen 2003; Wylie *et al.* 2003) to include:

1. Availability of bankside vegetative cover, typically tule (*Scirpus* sp.) or cattail (*Typha* sp.), for screening from predators.
2. Availability of more permanent shelter, such as bankside cracks or crevices, holes, or small mammal burrows.
3. Free of poor grazing management practices (such as overgrazed areas).

> *Upland Winter Refugia Component.* During the colder winter months, giant garter snakes spend their time in a lethargic state. During this period, giant garter snakes over-winter in locations such as mammal burrows along canal banks and marsh locations, or riprap along a railroad grade near a marsh or roads (Wylie *et al.* 1997; Wylie *et al.* 2002). Giant garter snakes typically do not over-winter where flooding occurs in channels with rapidly moving water, such as the Sutter Bypass (B. Halstead, USGS, pers. comm. 2011). Over-wintering snakes use burrows as far as 200 to 250 meters (656 to 820 feet) from the edge of summer aquatic habitat (G. Hansen 1988; Wylie *et al.* 1997; P. Coates, pers. comm. 2010). (U.S. Fish and Wildlife Service 2017b:iii, 11–13, v-2, 8)

This ends the verbatim description of suitable habitat from the U.S. Fish and Wildlife Service 2017 recovery plan. The exact text from the recovery plan was included here because of the specificity, level of detail, and supporting documentation.

2.7.5 Habitat Suitability Model

The habitat model for giant garter snake includes both aquatic and upland habitats. The modeled aquatic habitat relies on both delineation data that were collected for a smaller portion of the study area, in what is called the delineation study area, and suitable habitats found in the data sets outside the delineation study area.

The modeled upland habitat includes suitable habitat within 200 feet of modeled aquatic habitat. The 200-foot buffer stems from previous USFWS guidance on upland habitat when considering impacts on the species. Though the species is known to utilize uplands further than 200 feet, as discussed in Section 2.7.2, *Life History and Habitat Requirements*, the model is intended to capture where the majority of actively used upland habitat occurs.

For the tidal perennial aquatic features of the model, modeled habitat only extends 20 feet into tidal perennial aquatic polygons from the edges of adjacent land. In tidal perennial aquatic features (e.g., the Sacramento and San Joaquin Rivers and tidal zones in the central Delta), giant garter snakes are likely limited to shallow, near-shore habitats providing emergent vegetation and vegetative cover. Accordingly, tidal perennial aquatic features are buffered internally by 20 feet to capture the near-shore habitat and exclude the relatively deep-water areas that are considered unsuitable. The Clifton Court Forebay and Discovery Bay are excluded from modeled tidal perennial aquatic features, as the aquatic habitat is not suitable for giant garter snake. The extent of modeled habitat in the study area is depicted in Chapter 4, Figure 4.7-1. Information on species habitat model development is included in Appendix 4B, Section 4B.5.1.1.

2.7.5.1 Aquatic Habitat Model

Inside the Delineation Study Area

Modeled habitat includes the following types from the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, 2021).

- Tidal freshwater emergent wetland
    - Freshwater emergent wetland
- Nontidal freshwater perennial emergent wetland
    - Freshwater emergent wetland
- Agricultural
    - Agricultural ditch
- Nontidal perennial aquatic
    - Depression
    - Natural channel
- Tidal perennial aquatic
    - Tidal channel
    - Natural channel

Modeled habitat also includes the following types from the 2018 Statewide Crop Mapping (Land IQ and California Department of Water Resources 2021).

- Agricultural
    - Rice

  - Wild rice

Outside the Delineation Study Area

Modeled habitat includes the following types from the 2018 Statewide Crop Mapping (Land IQ and California Department of Water Resources 2021).

- Agricultural
  - Rice
  - Wild rice

Modeled habitat includes the following type from the National Hydrography Dataset (U.S. Geological Survey 2020).

- Agricultural
  - Ditches

Modeled habitat also includes the following types from the Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018) and the Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019).

- Tidal perennial aquatic
  - All types
- Tidal freshwater emergent wetland
  - All types
- Nontidal freshwater emergent wetland
  - All types
- Nontidal perennial aquatic
  - All types

2.7.5.2 Upland Habitat

Modeled habitat includes the following type from the Draft San Joaquin County 2017 Land Use Survey (California Department of Water Resources 2020b), the Sacramento County 2015 Land Use Survey (California Department of Water Resources 2016), and the Delta 2017 Land Use Survey (Land IQ 2019).

- Agricultural
  - Upland herbaceous

Modeled habitat includes the following type from the 2018 Statewide Crop Mapping (Land IQ and California Department of Water Resources 2021).

- Agricultural
  - Mixed pasture

Modeled habitat includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018) and the Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019).

- Alkaline seasonal wetland complex
    - *Allenrolfea occidentalis*
    - *Distichlis spicata*
    - *Frankenia salina*
    - *Juncus arcticus (var. balticus, mexicanus)*
    - Western North American disturbed alkaline marsh and meadow
- Grassland
    - All types
- Valley/foothill riparian
    - All types

Modeled habitat includes the following types from the DCP vernal pool complex (Witham et al. 2014; Chico State Research Foundation, Geographical Information Center 2019).

- Vernal pool complex
    - All types

2.8 Swainson's Hawk

The Swainson's hawk (*Buteo swainsoni*) is listed as a threatened species under the CESA (Cal. Code Regs. tit. 14, § 670.5, subd. (b)(5)(A)., California Fish and Game Code, Sections 2050 *et seq*.). The Swainson's hawk has no federal regulatory status. However, the species is included on the USFWS list of Birds of Conservation Concern for Region 1 (California Department of Fish and Wildlife 2020b:54).

2.8.1 Geographic Distribution and Status

Swainson's hawks nest in the grassland plains and agricultural regions of western North America from southern Canada (and possibly in the northern provinces and territories and Alaska) to northern Mexico (California Department of Fish and Wildlife 2016:5). Other than a few documented small wintering populations in the United States (Herzog 1996:877; Bechard et al. 2020), most populations winter primarily in the pampas of Argentina. The Central Valley population, however, winters mainly between Mexico and central South America (Airola et al. 2019:237).

The 2007 statewide population estimate for California was 2,081 breeding pairs (Anderson et al. 2007:2). Nearly 95% of nesting Swainson's hawks in California are found in the Central Valley (Anderson et al. 2007:3). Over 60% of the statewide population occurred within Yolo, Sacramento, Solano, and San Joaquin counties (Anderson et al. 2007:4). In 2018, the statewide population estimate for California was 18,810 breeding pairs, with over 85% of the statewide population occurring within 18.6 miles (30 kilometers) of the Central Valley (Furnas et al. 2022:6).

Swainson's hawk was observed perching, soaring, foraging, carrying nesting material, or nesting throughout the study area on numerous occasions during field surveys conducted in and around water conveyance facility footprints (California Department of Water Resources 2011). There are numerous (greater than 400) nesting records for Swainson's hawk throughout the study area where riparian and isolated trees are present (California Department of Fish and Wildlife 2024, 2020c).

2.8.2 Life History and Habitat Requirements

Swainson's hawks arrive on their breeding grounds in the Central Valley between March and April, and begin nest-building and egg-laying shortly after arrival (California Department of Fish and Wildlife 2016:5–6). Incubation of eggs lasts approximately 35 days and most young fledge approximately 6 weeks after hatching (typically by early July; California Department of Fish and Wildlife 2016:5–6). Post-breeding foraging flocks of up to 100 birds, often congregate on recently mowed or disked fields such as alfalfa or other row crops (California Department of Fish and Wildlife 2016:9) Migration back to the wintering grounds begins mid-August and most individuals leave California by October (California Department of Fish and Wildlife 2016:5–6).

In the Central Valley, nests are constructed in riparian woodlands, isolated trees, trees along roadsides, bordering fields, along the edges of remnant oak woodlands, and in small groves (Estep 2008:4-5). The majority of known nests in the Central Valley occur along narrow stringers of remnant riparian forest (Estep 2008:4-5; Estep 1984:20–21; Schlorff and Bloom 1984:827, 832; Bechard et al. 2020). Nests are usually constructed as high as possible in the tree, which provides good visibility and nest protection (Estep 2008:4-5). Swainson's hawks most commonly nest in large native trees such as valley oak (*Quercus lobata*), Fremont cottonwood (*Populus fremontii*), Hinds' walnut (*Juglans hindsii*), and willows (*Salix* spp.), and in nonnative trees, such as eucalyptus (*Eucalyptus* spp.) (Estep 2007:33, 2008:6-15). Nesting pairs will often use the same nesting territories and nesting trees year after year (Estep 2008:4-5). Many nest sites in the Central Valley have been occupied annually since 1979 and banding studies have shown a high degree of both nest and mate fidelity (Estep 2008:4-5).

Swainson's hawk historically foraged in open grasslands and prairies; however, with substantial conversion of grasslands for farming practices, Swainson's hawks have shifted their foraging to include agricultural lands that provide large rodent prey populations amid low, open vegetation (California Department of Fish and Wildlife 2016:5, 7). Foraging habitat value is a function of patch size, the ability to access prey (vegetation cover), and prey abundance (Estep 2008:4-7, 2009:2). In the Central Valley, land use or specific crop type and management practices determine the foraging value of a field at any given time. Important land cover or agricultural crops for foraging are alfalfa and other hay, disked fields, fallow fields, dryland pasture, and perennial grassland (Estep 1989:33; Babcock 1995:197; Woodbridge 1998:9–10). Central Valley Swainson's hawk preys on small mammals, birds, toads, crayfish, and insects. The primary prey species during the breeding season are California voles (*Microtus* spp.), pocket gophers (*Thomomys bottae*), and deer mice (*Peromyscus maniculatus*) (Estep 1989:19–20). Data on minimum foraging patch size are largely anecdotal but are in the range of between 5 and 25 acres (2 and 10 hectares) (Estep and Teresa 1992:778; California Department of Fish and Game 1994).

Data on minimum foraging patch size are largely anecdotal but are in the range of between 5 and 25 acres (2 and 10 hectares) (Estep and Teresa 1992:778; California Department of Fish and Game 1994). Although Swainson's hawks have been observed foraging in habitat patches smaller than 40 acres (16 hectares), 40-acre fields are more likely to be seen by Swainson's hawks and more likely to provide higher density prey (Stillwater Sciences 2014). In the Central Valley, land use or specific crop type and management practices determine the foraging value of a field at any given time. Important land cover or agricultural crops for foraging are alfalfa and other hay, grain and row crops, fallow fields, dryland pasture, and annual grasslands (Estep 1989:7–17; Babcock 1995:197; Woodbridge 1998:9–10). The matrix of these cover types across a large area creates a dynamic foraging landscape as temporal changes in vegetation results in changing foraging patterns and foraging ranges.

Home ranges are highly variable depending on landcover type and fluctuate throughout the breeding season with changes in vegetation structure from growth and harvesting of crops, and annually from crop rotation (Estep 1989:24; Woodbridge 1991:40–41; Babcock 1995:196). High-value crop types such as alfalfa, fallow fields, and pastures allow for smaller home ranges, whereas larger home ranges are associated with landcover with reduced prey availability, such as vineyards and orchards, or reduced prey abundance such as flooded fields (Estep 1989:30; Woodbridge 1991:40–41; Babcock 1995:197). Although Swainson's hawk have been recorded foraging up to 18 miles from a nest site, traveling more than 3 to 5 miles from a nest site to find high-value foraging sites may reduce reproductive success (Estep 1989:23, 40, 2008:4-8; England et al. 1995:185).

Swainson's hawks are highly responsive to farming and management activities that expose and concentrate prey, such as cultivating, harvesting, and discing (Estep 1989:23). During these activities, particularly late in the season, Swainson's hawks will hunt behind tractors searching for exposed prey. (California Department of Fish and Game 1994:6; Estep 1989:23). Other activities, such as flood irrigation, also expose prey and attract foraging Swainson's hawks (Estep 1989:23).

2.8.3 Species Threats

Swainson's hawk faces different threats in different portions of their range. In California, causes of population decline are thought to be loss of nesting habitat (Schlorff and Bloom 1984:827, 832) and loss and fragmentation of foraging habitat to urban development and to conversion to unsuitable agriculture, such as orchards and vineyards (California Department of Fish and Wildlife 2016). Conversion from compatible to incompatible crop patterns also reduces available foraging habitat and influences the distribution of nesting Swainson's hawks. Large regions of the Central Valley have been converted to rice, vineyards, orchards, cotton, and other incompatible crop types that support few Swainson's hawks. The conversion of suitable agricultural landscapes (e.g., annually rotated irrigated cropland, hayfields, and pasturelands) to vineyards and other unsuitable cover types continues to reduce available foraging habitat on a local and regional basis (California Department of Fish and Wildlife 2016). Spring and summer inundation of agricultural lands or other habitats also reduces available foraging habitat but can reveal prey.

Nestlings are vulnerable to starvation, fratricide (the larger nestling killing the smaller nestling in times of food stress), and predation from crows, ravens, and other raptors. Natural population cycles of voles in central California may be a factor in reproductive success when vole population crashes suppress Swainson's hawk reproduction or lead to increased nestling starvation rates. Insecticides and rodenticides may contribute to food scarcity by reducing prey abundance.

Loss of riparian and other nesting habitat continues throughout the Central Valley from levee projects, agricultural practices, and local development along watercourses. A related issue is the loss and lack of regeneration of valley oak and other native trees, an ongoing problem in areas that have continued to support remnant valley oaks and oak groves.

Adult Swainson's hawks are rarely killed by natural predators or competitors, but collisions with moving vehicles and illegal shooting and trapping have been identified as significant sources of mortality (England et al. 1997). Well-documented mass poisonings of hundreds or thousands of Swainson's hawks wintering in Argentina (Woodbridge et al. 1995:203; Goldstein et al. 1996:106–107) have led to that country's ban of an insecticide (monocrotophos) used on alfalfa and sunflower fields to control grasshopper populations. Levels of dichlorodiphenyldichloroethylene (DDE) (a toxic degradation product of dichlorodiphenyltrichloroethane [DDT], a pesticide used extensively until 1972 when it was banned in the United States) in Swainson's hawks from the Central Valley may have been high enough to negatively affect reproductive success during the decades when DDT was used extensively in the United States. However, levels of DDE measured in eggs collected in 1982 and 1983 were not considered high enough to indicate a health threat (Risebrough et al. 1989:63).

2.8.4 Suitable Habitat Definition

Swainson's hawk suitable nesting habitat includes mature trees (20 feet or greater) in riparian systems as well as in single, isolated and roadside trees (California Department of Fish and Game 1994). Nest sites are generally adjacent to or within easy flying distance to alfalfa or hay fields or other habitats or agricultural crops that provide abundant prey sources. The tree types listed below are known to be preferred by Swainson's hawk (California Department of Fish and Game 1994).

- Valley oaks (*Quercus lobata*)
- Fremont's cottonwood (*Populus fremontii*)
- Willows (*Salix* spp.)
- Sycamores (*Platanus* spp.)
- Walnuts (*Juglans* spp.)
- *Eucalyptus* spp.
- *Alnus* spp.

In addition, nest trees occupied by Swainson's hawks within the last 5 years are considered suitable nesting habitat (California Department of Fish and Game 1994). See Section 2.8.2, *Life History and Habitat Requirements*, and Section 2.8.5, *Nesting and Foraging Habitat Suitability Model*, for a complete description of nesting and foraging types.

2.8.5 Nesting and Foraging Habitat Suitability Model

As described in Section 2.8.2, Swainson's hawk nesting habitat includes valley/foothill riparian vegetation types with valley oak or cottonwood-dominated riparian forests considered optimal nesting habitat for this species. Swainson's hawks also nest in a variety of other native and nonnative isolated trees (e.g., Oregon ash (*Fraxinus latifolia*), box elder (*Acer negundo*), white alder

(*Alnus rhombifolia*), and *Eucalyptus* spp.) such as those on roadsides, in windbreaks, and around rural residences. Individual or small clumps of isolated trees are not mapped in the Delta and therefore this type of nesting habitat is not captured by the model. This underestimation of non-riparian habitat in the model is offset by the overestimation of potential riparian habitat. Riparian habitats are overestimated because not all riparian habitat is suitable for Swainson's hawk nesting, but all riparian habitat is considered suitable in the model.

Foraging habitat is also described in Section 2.8.2, and includes grasslands, managed and natural seasonal wetland types, and agricultural types such as irrigated pastures and hays and seasonally rotated croplands. The grain and hay, field, truck, nursery and berry crop types listed below are seasonally rotated and, therefore, the value of individual fields for foraging changes each year. These crop types are not differentiated based on their seasonal value in the model and are instead combined into a category of seasonally rotated croplands. As a result, this model overestimates the extent of available agricultural foraging habitat in any given year as suitable, seasonally rotated crops are exchanged with non-suitable crop types. To maintain consistency with CDFW guidance (California Department of Fish and Game 1994:12–13), a minimum foraging patch size of 5 acres is used. The extent of modeled habitat in the study area is depicted in Chapter 4, Figure 4.8-1.

This model maps the distribution of suitable Swainson's hawk nesting and foraging habitat throughout the study area, which overlaps with the year-round range for the species (Battistone 2011). Information on species habitat model development is included in Appendix 4B, Section 4B.5.1.1.

2.8.5.1 Nesting

Modeled nesting habitat includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019) and the Great Valley Ecoregion 2018 Vegetation dataset (Chico State Research Foundation, Geographical Information Center 2018).

- Valley/foothill riparian
    - *Alnus rhombifolia*
    - *Fraxinus latifolia*
    - *Acer negundo*
    - *Juglans hindsii* and hybrids
    - *Populus fremontii*
    - *Salix gooddingii*
    - *Quercus agrifolia*
    - *Quercus wislizeni* (tree)
    - *Quercus lobata*
    - Southwestern North American riparian evergreen and deciduous woodland
    - California broadleaf forest and woodland

  - Vancouverian riparian deciduous forest
  - *Eucalyptus* spp.—*Ailanthus altissima–Robinia pseudoacacia*
  - Introduced North American Mediterranean woodland and forest
  - *Salix exigua*
  - *Salix laevigata*
  - *Salix lasiolepis*
  - *Platanus racemosa* alliance

Modeled nesting habitat also includes the following types from the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021).

- Forested wetland

2.8.5.2 Foraging

Modeled foraging habitat includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), Great Valley Ecoregion 2018 Vegetation dataset (Chico State Research Foundation, Geographical Information Center 2018), DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021), and DCP Vernal Pool Complex (Witham et al. 2014; Chico State Research Foundation, Geographical Information Center 2019; California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021) layers with a patch size of at least 5 acres (California Department of Fish and Game 1994:12–13).

- Alkaline seasonal wetland
  - All types
- Grassland
  - All types
- Other seasonal wetland
  - All types
- Vernal pool complex
  - All types

Modeled foraging habitat also includes the following landcover types from the 2018 Statewide Crop Mapping (Land IQ and California Department of Water Resources 2021) layer and the Delta, Sacramento County, and San Joaquin County, Land Use Survey (Land IQ 2019; California Department of Water Resources 2016, 2020) layers with a patch size of at least 5 acres (California Department of Fish and Game 1994:12–13).

- Agricultural
    - Wheat
    - Miscellaneous grain and hay
    - Safflower
    - Miscellaneous field crops
    - Corn, sorghum, and sudan
    - Beans (dry)
    - Sunflowers
    - Alfalfa and alfalfa mixtures
    - Miscellaneous grasses
    - Mixed pasture
    - Miscellaneous truck crops
    - Young perennials
    - Carrots
    - Cole crops
    - Melons, squash, and cucumbers
    - Onions and garlic
    - Bush berries
    - Strawberries
    - Tomatoes
    - Peppers
    - Fallow
    - Unclassified fallow
    - Upland herbaceous

2.8.5.3 Habitat Value Categories

Most of the Delta consists of agricultural land and most is considered to have some value as foraging habitat for Swainson's hawk. However, the value of crop types differs widely due to their growth and structure, which influences accessibility by foraging hawks, and in prey abundance, which influences the availability of prey. Because of the dynamic nature of the agricultural landscape and the variability of crop patterns and conditions seasonally and annually, only a proportion of the agricultural landscape is suitable or available for foraging in any given season or year. Lands that are fallowed in any given season or year will be assigned the same habitat value as that assigned to the dominant crop type on those lands.

Sufficient information is available on the growth and structure of different agricultural crops and the prey abundance and use of different crop types to generally categorize crops based on their value as foraging habitat. Table 2.8-1 categorizes modeled land cover types according to three relative value classes: very high, high, medium, and low. These value classes correspond to the mitigation requirement for the Swainson's hawk with regard to maintaining medium- to very high–value habitat types on protected mitigation lands.

Table 2.8-1. Swainson's Hawk Foraging Habitat Value Classes

Habitat Value Class	Habitat	Rationale for Assignment of Value Class	Information Sources
Very high value	Alfalfa and alfalfa mixtures	Alfalfa has the highest value because it is semiperennial (up to 5 years before rotation), which increases prey abundance; has a relatively low profile such that prey are accessible season-long; and has a management regime (mowing and irrigation) which further increases prey accessibility.	Estep 1989, 2009; Swolgaard et al. 2008
High value	Mixed pasture, miscellaneous grasses, upland herbaceous	These pasture types provide a relatively consistent vegetation structure and rodent prey populations. There is less seasonal variability with respect to prey abundance and accessibility compared with grain and vegetable crops, but they lack the management practices that enhance prey accessibility found in alfalfa.	Estep 1989, 2009; Swolgaard et al. 2008
Medium value	Grasslands, managed wetlands, alkaline seasonal wetlands, vernal pool complex, tomatoes, wheat, miscellaneous grain and hay	Certain row crops, such as beets and tomatoes, have a relatively high value because they support large rodent prey populations, are accessible season-long because of their relatively low vegetation profile, and they are harvested prior to migration, when an abundance of prey becomes available. Most grain crops provide value during and following harvesting, when prey become accessible. Grasslands are generally available season-long but provide lower prey abundance compared with higher value agricultural habitats, don't provide a peak period of high-value abundance and accessibility like some agricultural crops (e.g., tomatoes), and in some cases grass height reduces prey accessibility during a portion of the breeding season.	Estep 1989, 2009; Swolgaard et al. 2008
Low value	Cole crops, corn, sorghum, and sudan, dry beans, field crops, young perennials, miscellaneous truck crops, carrots, melons, squash, cucumbers, onions and garlic, peppers, miscellaneous field	These agriculture types are suitable for a portion of the breeding season depending on their structure and planting/harvesting regime. In general, they produce less prey abundance and less prey availability than the other agriculture types listed above.	Estep 1989, 2008; Swolgaard et al. 2008

Habitat Value Class	Habitat	Rationale for Assignment of Value Class	Information Sources
	crops, safflower, sunflower		

2.9 Tricolored Blackbird

2.9.1 Geographic Distribution and Status

Tricolored blackbird was listed as threatened by the California Fish and Game Commission pursuant to CESA (Cal. Code Regs. tit. 14, § 670.5 (b)(5)(H), Fish and Game Code, Sections 2050 *et seq.)* on April 19, 2018, and is designated as a California Species of Special Concern (California Department of Fish and Wildlife 2020b). Tricolored blackbird has no federal regulatory status; however, the species is protected under the federal Migratory Bird Treaty Act and is designated as a Bird of Conservation Concern by USFWS (U.S. Fish and Wildlife Service 2019a:52). A petition was submitted to USFWS in 2004 and was denied in 2006, based on insufficient scientific evidence to warrant listing the species under the federal ESA. Another petition was submitted to USFWS on February 3, 2015, to list tricolored blackbird under the ESA; on September 18, 2015, USFWS issued a 90-day finding of the petition stating that listing may be warranted and requested more information (80 FR 56423–56432). In 2019, the USFWS issued a 12-month finding on the petition that listing the tricolored blackbird as endangered or threatened is not warranted (84 FR 41694–41699).

The tricolored blackbird is a colonial nesting passerine bird that is largely restricted to California. The species forms some of the largest colonies of any North American passerine bird, which may contain tens of thousands of breeding pairs (Beedy et al. 2020). Most of the California breeding population of tricolored blackbird occurs in the Central Valley (California Department of Fish and Wildlife 2018:40; Beedy et al. 2020). Statewide surveys conducted in 2017 documented 51% of breeding birds in Merced and Kern Counties (Meese 2017:11). While the geographic extent of the tricolored blackbird's range has been largely unchanged since the 1930s (Neff 1937:61–81; DeHaven et al. 1975:168–171, 178–179; Beedy et al. 1991:1; Beedy 2008:437–439; Hamilton 1998:225; California Department of Fish and Wildlife 2018:40; Beedy et al. 2020), substantial annual variation in centers of breeding abundance have been regularly documented over the past 70 years, particularly between the Sacramento and San Joaquin Valleys (California Department of Fish and Wildlife 2018:59). These shifts in abundance are indicative of the tricolored blackbird's ability to adapt to variation in food supply and nesting substrate (California Department of Fish and Wildlife 2018:59). Wintering tricolored blackbirds often form huge, mixed species flocks that forage across the landscape. The Delta and central coast are recognized as major wintering areas for tricolored blackbirds (Beedy 2008:439; California Department of Fish and Wildlife 2018:14).

Historical population sizes of tricolored blackbird are unknown, but by the mid-1930s, following the removal of most major wetland areas in the state, populations still likely exceeded 1.1 million adult birds (Hamilton 1998:225). In the first systematically conducted range-wide surveys, Neff (1937) documented 252 colonies of tricolored blackbirds in 26 California counties, including over 700,000 adults in eight Central Valley counties. Surveys conducted in the 1960s and 1970s indicated that range-wide populations declined by more than 50% during the 30- to 35-year period following Neff's surveys in the 1930s (Orians 1961:287, 297, 310; Payne 1969:43–45; DeHaven et al. 1975:168, 176).

In the 1990s and 2000s, USFWS, CDFW, and California Audubon have cosponsored systematic tricolored blackbird surveys throughout California. Surveys during the 1990s (Hamilton et al. 1995:5–6; Hamilton 2000:5–6; Beedy et al. 2020) confirmed the significant declining trend in California populations since the 1930s, with a particularly rapid decline noted after 1994. A population low of 94,269 adult birds was documented during the 1999 survey. Statewide surveys conducted during the 2000s indicate some recovery from the 1999 low; however, the population increases have primarily been limited to the San Joaquin Valley and the Tulare Basin (Kyle and Kelsey 2011:5). A total of 145,135 adults were counted during the 2014 statewide survey, representing a population decline of about 44% from the previous statewide count of 258,000 birds in 2011 and least productive breeding season ever recorded during the statewide surveys (Meese 2014:6, 9). The statewide survey conducted in 2017 estimated 177,700 adults, a 22% increase from 2014, but a 55% population decline since 2008, despite an increase in number of sites surveyed (Meese 2017:10, 14, 18). A total of 218,000 adult birds were counted during the 2022 statewide survey, representing a continuation of modest increase in the statewide population (Colibri Ecological Consulting 2022:ii, 19). The estimate represents a 23% increase from the 2017 statewide population estimate and a 50% increase over the population low in 2014. However, the increasing population trend is still only 55% of the adult birds estimated during the 2008 statewide survey (Colibri Ecological Consulting 2022:ii, 19).

These recent increases are in part due to more intensive and focused survey efforts (i.e., more locations surveyed than in any previous statewide survey), potentially an increase in fecundity and recruitment a decrease in mortality (Colibri Ecological Consulting 2022), and reductions in human-caused mortality resulting from the silage buy-out programs at dairies in the San Joaquin Valley and Riverside and San Benito Counties (Beedy et al. 2020). The total proportion of breeding birds from the 10 largest colonies has decreased from 77.5% and 81% in years in 2008 and 2011, respectively, to 64% and 55% in 2014 and 2017, respectively (Meese 2017:10, 14, 18). This reflects a downward trend in the size of the largest colonies.

Survey data also indicate that populations continue to be the highest in the San Joaquin Valley, where the species was formerly common. In the San Joaquin Valley, the tricolored blackbird population had dropped 78% by 2014 (Meese 2014:6, 9), but increased by 38% in 2017 (340,730 in 2008, 73,482 in 2014, and 118,049 in 2017) (Meese 2017:10, 14, 18). In 2022, the tricolored blackbird population in the San Joaquin Valley were 63% lower than those in 2017 (Colibri Ecological Consulting 2022:ii). In contrast to the San Joaquin Valley, the Sacramento Valley showed a 208% increase over the same period, suggesting a potential northward shift in the species distribution between the 2017 and 2022 breeding seasons (Colibri Ecological Consulting 2022:ii).

Surveys in 2018 of 67 sites in Fresno, Kings, Madera, Merced, Tulare, and Kern Counties located 15 breeding colonies in Merced, Tulare, and Kern Counties, of which the 12 nests in silage fields had an estimated total of 92,100 to 126,500 adult tricolored blackbirds (Colibri Ecological Consulting 2018:1). In 2022, San Joaquin Valley tricolored blackbird colonies were concentrated in Merced, Tulare, and Kern Counties (Colibri Ecological Consulting 2022:10). Breeding bird populations in 2014 in counties along the central coast were less than 10% of those seen in 2008 (7,514 in 2008, 1,562 in 2014) (Meese 2014:6, 9); however, in 2017 the population was estimated to be more than double that seen in 2008 (18,336 in 2017) due to the identification of three new colonies (Meese 2017:10, 14, 18). The number of tricolored blackbirds observed in the Central Coast bioregion declined slightly from 2017 (Colibri Ecological Consulting 2022:14, 15).

Based upon recent survey results, the tricolored blackbird appears to be an uncommon breeder in the Delta (Meese pers. comm.[a]) Historical nesting activity was generally restricted to the northern and southern ends of the Delta. There are 11 sites where breeding occurred at least once between 2005 and 2020 that are within the study area. Most of these breeding records are single year occurrences but tricolored blackbirds do sometimes nest at the same site, sometimes in sequential years and sometimes with one or more years in between. These nesting sites range from having as few as 3 to as many as 2,000 breeding adults per site (Meese pers. comm.[a].; California Department of Fish and Wildlife 2020c). The Delta is recognized as an important wintering area for tricolored blackbirds (Hamilton 2004; Beedy 2008:438). Tricolored blackbirds are nomadic during the nonbreeding season; therefore, roost site locations vary annually (California Department of Fish and Wildlife 2018:16). The species roosts and forages in large, mixed, wintering flocks with other blackbird species throughout the study area including Sherman Island (eBird 2021; Tricolored Blackbird Portal 2021). In 2024, eBird.org also reports wintering tricolored blackbird at Cosumnes River Preserve, a feed lot southwest of Clifton Court Forebay, Bradford Island, Bethal Island, Jersey Island, and Sherman Island (eBird 2024b).

2.9.2 Life History and Habitat Requirements

Tricolored blackbirds nest colonially, enabling them to synchronize their timing of nest building and egg laying (Beedy et al. 2020). Tricolored blackbird typically nests in areas with open accessible water, a nesting substrate that is protected from ground predators (e.g., vegetation that is flooded, thorny, or spiny), and suitable foraging habitat (e.g., pastures, dry seasonal pools, agricultural fields such as alfalfa and sunflower) that provides abundant insect prey (Hamilton et al. 1995:25; California Department of Fish and Wildlife 2018:27; Beedy et al. 2020). Open water within 1,640 feet (500 meters) of nesting substrate is a requirement for colony settlement (Hamilton 2004). Breeding colonies have been recorded in freshwater marshes, willows, blackberries, thistles, and nettles, and more recently in triticale and other grain fields in the San Joaquin Valley (California Department of Fish and Wildlife 2018:24–27). Most breeding tricolored blackbirds forage within 5 miles of their colony sites (California Department of Fish and Wildlife 2018:28; U.S. Fish and Wildlife Service 2019b:24). Foraging is typically concentrated in areas that support abundant insect populations, a vital food resource for provisioning nestlings (Beedy 2008:440). Foraging habitat includes grasslands, alkaline seasonal wetlands, vernal pools, pastures and agricultural crops such as alfalfa and rice, which produce a high abundance of insects, in addition to cattle feedlots and dairies, which supply grains for foraging individuals (California Department of Fish and Wildlife 2018:28).

Roosting by tricolored blackbirds during the fall and winter generally occurs in emergent wetlands consisting of cattails (*Typha* spp.) and bulrushes (*Schoenoplectus* spp.; U.S. Fish and Wildlife Service 2019b:12) During the nonbreeding season, tricolored blackbirds often congregate at dairy feedlots to consume grains and other livestock feed, or forage on insects, grains, and other plant material in grasslands and agricultural fields (Beedy et al. 2020).

In the Central Valley, breeding typically occurs between mid-March and mid-August (California Department of Fish and Wildlife 2018:31). Females typically lay 3 to 4 eggs and incubate them for 11 to 14 days (Emlen 1941:216–217; Orians 1961:287, 297, 310); then both parents feed young until they fledge approximately 9 to 14 days after hatching (U.S. Fish and Wildlife Service 2019b:11). The colony itself remains active and in various stages of the breeding cycle for an extended period, which may last more than 90 days, but generally requires a minimum of 50 days for a complete breeding cycle (Beedy et al. 2020). Many tricolored blackbirds reside throughout the year in the Central Valley of California. Individual tricolored blackbirds may occupy and breed at several sites, or renest at the same site during a given breeding season, depending on environmental conditions and their previous nesting success (Hamilton 1998:225; Beedy et al. 2020; Meese 2006:5). In the fall, after the nesting season, large roosts form at managed wildlife refuges and other marshes near abundant food supplies such as cultivated rice (*Oryza sativa*) and water grass (*Echinochloa crus-galli*) (Beedy et al. 2020). During winter, many tricolored blackbirds move from the Sacramento Valley to the Delta, central and southern San Joaquin Valley, and to the dairy farms in coastal areas such as Point Reyes and Monterey County (Beedy and Hamilton 1997:17–19).

2.9.3 Species Threats

The most significant threat to the tricolored blackbird is habitat loss and alteration. The initial conversion from native landscapes to agriculture removed vast wetland areas in the state and caused initial declines in populations. The more recent conversion of suitable agricultural lands to urban areas has permanently removed historical breeding and foraging habitat for this species. In urbanizing areas, habitat fragmentation and proximity to human disturbances has also led to abandonment of large historical colonies (Beedy et al. 2020). In Sacramento County, a historical breeding center of this species, the conversion of grassland and pastures to vineyards expanded from 7,537 acres in 1996, to 13,171 acres in 1998 (DeHaven 2000:11–13), to 16,709 acres in 2003 (U.S. Department of Agriculture 2012). Conversions of pastures and grasslands to vineyards and orchards in Sacramento County and elsewhere in the species' range in the Central Valley have resulted in the recent loss of several large colonies and the elimination of extensive areas of suitable foraging habitat for this species (DeHaven 2000:11–13; Hamilton 2004).

Entire colonies (up to tens of thousands of nests) in cereal crops and silage are often destroyed by harvesting and plowing of agricultural lands (Beedy et al. 2020; Hamilton 2004; Cook and Toft 2005:73–75; Colibri Ecological Consulting 2017:1). While adult birds can fly away, eggs and fledglings cannot. The concentration of a high proportion of the known population in a few breeding colonies increases the risk of major reproductive failures, especially in vulnerable habitats such as active agricultural fields).

Historical accounts documented the destruction of nesting colonies by a diversity of avian, mammalian, and reptilian predators (Beedy et al. 2020). Recently, especially in perennial freshwater marshes of the Central Valley, entire colonies have been lost to black-crowned night-herons (*Nycticorax nycticorax*) and common ravens (*Corvus corax*). Since 2006, cattle egrets (*Bubulcus ibis*) have been observed preying on tricolored blackbird nests in Tulare County. Predation by cattle egrets has become so severe that complete reproductive failure has occurred in at least one large colony for 5 consecutive years (Meese 2011). Some large colonies (up to 100,000 adults) may lose greater than 50% of nests to coyotes (*Canis latrans*), especially in silage fields, but also in freshwater marshes when water is withdrawn (Hamilton et al. 1995:5–6). Thus, water management by humans often has the effect of increasing predator access to active colonies).

Tricolored blackbird colonies are highly sensitive to human disturbances. Proximity to urbanizing areas can cause colonies to be permanently abandoned. Increases in noise, loose pets, and human presence can cause nest abandonment. Even entry into colonies for management or scientific purposes can cause disturbance and should be avoided (Beedy et al. 2020). Various poisons and contaminants have caused mass mortality of tricolored blackbirds. McCabe (1932:49–50) described the strychnine poisoning of 30,000 breeding adults as part of an agricultural experiment. Neff

(1942:46) considered poisoning to regulate numbers of blackbirds preying upon crops (especially rice) to be a major source of mortality. This practice continued until the 1960s, and thousands of tricolored blackbirds and other blackbirds were exterminated to control damage to rice crops in the Central Valley. Beedy and Hayworth (1992:33–46) observed a complete nesting failure of a large colony (about 47,000 breeding adults) at Kesterson Reservoir in Merced County; selenium toxicosis was diagnosed as the primary cause of death. At a colony in Kern County, all eggs sprayed by mosquito abatement oil failed to hatch (Beedy et al. 2020). Other factors that may affect the nesting success of colonies in agricultural areas include herbicide and pesticide applications (Beedy et al. 2020).

2.9.4 Suitable Habitat Definition

Suitable breeding season habitat for tricolored blackbirds consists of previously occupied (within the last fifteen years) colony habitat and active nesting colonies. Suitable breeding foraging habitat is any one of the natural or cultivated types listed in Section 2.9.5.1, *Breeding Habitat*, within 3.1 miles (5 kilometers) of previously occupied (within the last 15 years) or potentially suitable colony habitat. Suitable non-breeding foraging habitat is any one of the natural or cultivated types listed in Section 2.9.5.3, *Non-Breeding Foraging Habitat*, that occurs outside the 3.1-mile buffer around of previously occupied (within the last 15 years) or potentially suitable colony habitat.

2.9.5 Habitat Suitability Model

The tricolored blackbird model consists of three components: previously occupied colony habitat, potential nesting habitat, and foraging (breeding and non-breeding) habitat. Overwintering habitat is not modeled because the available scientific literature does not thoroughly describe tricolored blackbird nonbreeding habitat (California Department of Fish and Wildlife 2018:23–30; U.S. Fish and Wildlife Service 2019b:24) and the loss of overwintering roosting habitat is not likely to result in injury or mortality of individuals. However, emergent vegetation within modeled previously occupied colony habitat and potential nesting habitat could also support overwintering tricolored blackbird night roosts (U.S. Fish and Wildlife Service 2019b:12).

The modeled previously occupied colony habitat and potential nesting habitat rely on both delineation data that was collected for a smaller portion of the study area, in what is called the delineation study area, and suitable habitats found in the datasets outside the delineation study area. Foraging habitat includes grasslands; seasonal wetlands; shrublands; riparian scrub; agricultural lands such as hay, pastures (including alfalfa), and sunflowers; and growing or stored grain crops in agricultural lands (such as livestock feedlots and dairies). During the breeding season, tricolored blackbirds typically forage within 3.1 miles (5 kilometers) of a colony site (California Department of Fish and Wildlife 2018:28; U.S. Fish and Wildlife Service 2019b:24); therefore, the modeled breeding foraging habitat includes suitable landcover types within 3.1 miles of modeled previously occupied colony habitat and potential nesting habitat. Modeled non-breeding foraging habitat includes suitable landcover types outside 3.1 miles (5 kilometers) of modeled previously occupied colony habitat and potential nesting habitat. The extent of modeled habitat in the study area is depicted in Chapter 4, Figure 4.9-1.

The model includes the entire study area because although the tricolored blackbird is an uncommon breeder in the Delta, there is potential habitat and there are records of known colonies distributed throughout the study area, as described in Section 2.9.1, *Geographic Distribution and Status*. Information on species habitat model development is included in Appendix 4B, Section 4B.5.1.1.

2.9.5.1 Breeding Habitat

Previously Occupied Colony Habitat

Previously occupied colony habitat has had active nesting within the past 15 years (2005–2020). Colony locations consist of either a CNDDB breeding colony polygon (California Department of Fish and Wildlife 2020c) or a Tricolored Blackbird Portal Colony location (Meese pers. comm.[b]). Colony locations from the Tricolored Blackbird Portal were buffered by 373 feet to convert a point location into a polygon. A circle with a radius of 373 feet has an area of 10 acres, which is the average area of a tricolored blackbird colony (California Department of Fish and Wildlife 2018:16, 28–29). Previously occupied colony habitat was modeled using the following methods.

- If a colony location consisted of a CNDDB polygon from the past 15 years, previously occupied colony habitat includes suitable nesting land cover types (listed below) within the mapped boundary of the CNDDB polygon.
- If a buffered Tricolored Blackbird Portal Colony point was located within the boundary of a CNDDB polygon from the past 15 years, previously occupied colony habitat includes suitable nesting land cover types (listed below) within the mapped boundary of the CNDDB polygon. If a Tricolored Blackbird Portal Colony point did not overlap with a CNDDB polygon, previously occupied habitat includes the suitable nesting land cover types (listed below) within the 373-foot buffer around the colony point.
- If a CNDDB or Tricolored Blackbird Portal colony occurrence reported nesting in mustard or stinging nettle, previously occupied colony habitat was mapped as including suitable nesting land cover types (listed below) *and* vegetation types within the grassland natural community (as nettle and mustard stands typically occur within the grassland natural community).

Inside the Delineation Study Area

Modeled previously occupied colony habitat includes the following types from the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021).

- Nontidal freshwater perennial emergent wetland
    - Freshwater emergent wetland
- Tidal freshwater emergent wetland
    - Freshwater emergent wetland
- Valley/foothill riparian
    - Scrub shrub wetland

Modeled previously occupied colony habitat also includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), and the Great Valley Ecoregion 2018 Vegetation (Chico State Research Foundation, Geographical Information Center 2018) datasets.

- Valley/foothill riparian
  - Rubus *armeniacus*
  - *Sambucus nigra*
  - *Salix lasiolepis*
  - *Vitis californica*
  - *Salix exigua*
  - *Rosa californica*
  - *Salix gooddingii*
  - Southwestern North American riparian/wash scrub
  - *Salix laevigata*
  - *Salix lucida*
  - Southwestern North American introduced riparian scrub

Outside the Delineation Study Area

Modeled previously occupied colony habitat includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), and the Great Valley Ecoregion 2018 Vegetation dataset (Chico State Research Foundation, Geographical Information Center 2018).

- Nontidal freshwater perennial emergent wetland
  - *Schoenoplectus americanus*
  - Schoenoplectus (*acutus, californicus*)
  - Typha (*angustifolia, domingensis, latifolia*)
  - Arid West freshwater emergent marsh
  - Salix lasiolepis
  - Naturalized warm-temperate riparian and wetland group
- Nontidal brackish perennial emergent wetland
  - Arid West freshwater emergent marsh
  - Schoenoplectus (*acutus, californicus*)
  - Typha (*angustifolia, domingensis, latifolia*)
  - Naturalized warm-temperate riparian and wetland group
- Valley/foothill riparian
  - *Rubus armeniacus*
  - *Sambucus nigra*

  - *Salix lasiolepis*
  - *Vitis californica*
  - *Salix exigua*
  - *Rosa californica*
  - *Salix gooddingii*
  - Southwestern North American riparian/wash scrub
  - Salix *laevigata*
  - *Salix lu*cida
  - Southwestern North American introduced riparian scrub
- Tidal freshwater emergent wetland
  - Schoenoplectus (*acutus, californicus*)
  - *Schoenoplectus americanus*
  - Typha (*angustifolia, domingensis, latifolia*)
  - Arid West freshwater emergent marsh
  - Tidal freshwater emergent wetland
  - Naturalized warm-temperate riparian and wetland group
- Tidal brackish emergent wetland
  - Schoenoplectus (*acutus, californicus*)
  - *Schoenoplectus americanus*
  - Typha (*angustifolia, domingensis, latifolia*)
  - Arid West freshwater emergent marsh
- Naturalized warm-temperate riparian and wetland group

Potential Nesting Habitat

Potential nesting habitat includes the same following natural community and vegetation types when they do not overlap with previously occupied colony habitat (defined above).

2.9.5.2 Breeding Foraging Habitat

Modeled breeding foraging habitat includes the following landcover types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021), and DCP Vernal Pool Complex (Witham et al. 2014; Chico State Research Foundation, Geographical Information Center 2019; California Department of Water Resources and GEI Consultants Inc. 2020) layers that occur within 3.1 miles (5 kilometers) of previously occupied colony habitat or potentially suitable colony habitat.

- Alkaline seasonal wetland
    - All types
- Grassland
    - All types
- Other seasonal wetlands
    - All types
- Vernal pool complex
    - All types

Modeled breeding foraging habitat also includes the following landcover types from the 2018 Statewide Crop Mapping (Land IQ and California Department of Water Resources 2021) layer and Delta, Sacramento County, and San Joaquin County Land Use Survey (Land IQ 2019; California Department of Water Resources 2016, 2020) layers that occur within 3.1 miles (5 kilometers) of previously occupied colony habitat or potentially suitable colony habitat.

- Alfalfa and alfalfa mixtures
- Fallow
- Miscellaneous grain and hay
- Miscellaneous grasses
- Mixed pasture
- Rice
- Sunflowers
- Unclassified fallow
- Upland herbaceous
- Wheat
- Wild rice

Modeled breeding foraging habitat also includes the following semiagricultural land cover types (Farmland Mapping Staff 2016a, 2016b, 2016c, 2016d, 2018).

- Livestock feedlots
- Dairies
- Poultry farms

2.9.5.3 Non-Breeding Foraging Habitat

Modeled tricolored blackbird non-breeding foraging habitat includes the same landcover types as breeding foraging habitat (Section 2.9.5.2, *Breeding Foraging Habitat*) but occurs outside of the 3.1-mile (5 kilometers) buffer around previously occupied colony habitat or potentially suitable colony habitat (where breeding foraging habitat occurs). For easy reference, these landcover types are listed below.

- Alkaline seasonal wetland
    - All types
- Grassland
    - All types
- Other seasonal wetlands
    - All types
- Vernal pool complex
    - All types

Modeled non-breeding foraging habitat also includes the following landcover types from the 2018 Statewide Crop Mapping (Land IQ and California Department of Water Resources 2021) layer and Delta, Sacramento County, and San Joaquin County Land Use Survey (Land IQ 2019; California Department of Water Resources 2016, 2020) layers that occur outside of 3.1 miles (5 kilometers) of previously occupied colony habitat or potentially suitable colony habitat.

- Alfalfa and alfalfa mixtures
- Fallow
- Miscellaneous grain and hay
- Miscellaneous grasses
- Mixed pasture
- Rice
- Sunflowers
- Unclassified fallow
- Upland herbaceous
- Wheat
- Wild rice

Modeled non-breeding foraging habitat also includes the following semiagricultural land cover types (Farmland Mapping Staff 2016a, 2016b, 2016c, 2016d, 2018).

- Livestock feedlots
- Dairies
- Poultry farms

2.9.5.4 Foraging Habitat Value Categories

Tricolored blackbird foraging habitat is made up primarily of agricultural lands and grasslands. Agricultural foraging habitat value differs by crop type. Given the dynamic nature of the agricultural landscape and the variability of crop patterns and conditions seasonally and annually, only a proportion of the agricultural landscape is suitable or available for foraging in any given year.

Sufficient information is available on the use of different foraging habitat types to generally categorize their value as breeding season foraging habitat. Table 2.9-1 categorizes modeled land cover types according to four relative value classes: very high, high, moderate, and low. These value classes correspond to the mitigation requirement for tricolored blackbird with regard to maintaining high- to very high–value types on protected mitigation lands.

Table 2.9-1. Tricolored Blackbird Breeding Foraging Habitat Value Classes

Foraging Habitat Value Class	Breeding Season Foraging Habitat
Very High	Native pasture, nonirrigated native pasture, annual grasslands, vernal pool grasslands, alkali grasslands, unsprayed alfalfa, unsprayed sunflower, unsprayed mixed alfalfa
High	Sunflower, alfalfa and mixed alfalfa, mixed pasture, induced high water table native pasture, nonirrigated mixed pasture, dairies
Moderate	Miscellaneous grass pasture, fallow lands cropped within 3 years, new lands prepped for crop production, livestock feed lots, organic rice
Low	Wheat, mixed grain and hay, farmsteads, rice

2.10 Crotch Bumble Bee

In June 2019, the California Fish and Game Commission accepted a petition to list Crotch bumble bee (*Bombus crotchii*) as a candidate for listing as endangered under CESA (California Department of Fish and Wildlife 2020b:26). Agricultural interests challenged CDFW’s ability to list the species, claiming that CESA’s definition of a “species” does not cover insects. In November 2020, the Sacramento Superior Court ruled that insects are not eligible for listing under CESA (*Almond Alliance of California v. California Fish and Game Commission*); however, in May 2022, the California Court of Appeal reversed the prior ruling, deciding that invertebrates are eligible for CESA listing.[11] CDFW reinstated candidate status for listing under CESA on September 30, 2022. Its NatureServe ranking is G3G4/S1S2. Crotch bumble bee has no federal regulatory status.

2.10.1 Geographic Distribution and Status

Crotch bumble bee has a relatively limited distribution, occurring primarily in California and northern Baja California, Mexico. In California, Crotch bumble bee historically occurred on the Pacific Coast, in the western desert, Central Valley, and adjacent foothills (California Department of Fish and Wildlife 2023b; Williams et al. 2014: 114–116, 132). The species range map suggests the species is no longer present in portions of California’s Pacific Northwest, Central Coast, and inland deserts (California Department of Fish and Wildlife 2023b). The species relative abundance on the landscape remains low (< 5%) similar to historical frequencies (Richardson et al. 2022).

Crotch bumble bee has potential to occur throughout the study area where suitable habitat exists. There are three CNDDB occurrences within the study area, one is a collection occurrence from 1926 near Antioch and one is from 1959 near Tracy (California Department of Fish and Wildlife 2020c). A recent occurrence, from 2020, is at the Cosumnes River Preserve Visitor Center within a human

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[11] *Almond All. of Cal. v. Fish & Game Comm'n* (2022) 79 Cal.App.5th 337, 294 [Cal. Rptr. 3d 603]

planted area of native plants (California Department of Fish and Wildlife 2024). It should be noted that there have not been systematic surveys throughout the study area and a lack of more recent records does not mean the species is no longer present or at additional locations.

2.10.2 Life History and Habitat Requirements

Bumble bees live in annual colonies. The queens, which dispersed from the colony the prior fall, emerge from overwintering sites in the spring, forage on pollen and nectar, and establish new colonies. Like most other species of bumble bees, Crotch bumble bees typically nest in underground cavities such as animal burrows (Williams et al. 2014), though nests have also been reported in aboveground structures that provide suitable cavities (Koch et al. 2012:9), such as old bird nests and empty tree cavities. Colonies are established by mated queens who produce female workers to forage for pollen and nectar, defend the colony, and feed developing larvae (Koch et al. 2012:9). The mated queens also produce males, which leave the nest in search of mates. Queens produced at the end of the colony cycle mate before entering overwintering sites. Little information is known about the overwintering sites used by Crotch bumble bee (Xerces Society for Invertebrate Conservation et al. 2018). Generally, bumble bees overwinter in soft, disturbed soil (Goulson 2010), or under leaf litter or other debris (Williams et al. 2014).

The flight periods for queens, workers, and male Crotch bumble bees vary (Thorp et al. 1983). The flight period for Crotch bumble bee queens in California is from late February to late October, peaking in April with a second peak in July. The flight period for workers and males in California is from late March through September (California Department of Fish and Wildlife 2019:17). Little is known about Crotch bumble bee overwintering sites but other bumble bee species are known to overwinter in soft soil or under leaf litter and debris (California Department of Fish and Wildlife 2019:17); queens will also overwinter in cavities below the ground and occasionally in wood piles or rock walls.

Bumble bees have three main habitat requirements: flowering plants that provide pollen and nectar, nest sites, and overwintering sites (Hatfield et al. 2012). Crotch bumble bee occurs primarily in areas dominated by grassland and scrub habitats, although it also occurs in a wider variety of habitats providing the three main habitat needs. Crotch bumble bee is a generalist forager, using a vast array of flowering plants. Thorp et al. (1983) reported that Crotch bumble bee had been recorded foraging on 186 different plant species in 15 families and 33 genera. Thorp et al. (1983) reported that the most frequently used genera were *Asclepias*, *Chaenactis*, *Lotus*, *Lupinus*, *Medicago*, *Penstemon*, *Phacelia*, and *Salvia*. Koch et al. (2012) reported plants in the genera *Antirrhinum*, *Phacelia*, *Clarkia*, *Dendromecon*, *Eschscholzia*, and *Eriogonum* as being typical food plants of Crotch bumble bee, while Williams et al. (2014) included the genera *Asclepias, Chaenactis, Lupinus, Medicago, Phacelia,* and *Salvia* as examples of Crotch bumble bee food plants. Richardson (2022) reported that Crotch bumble bee observations and collections from California are associated most commonly with plants from the families Fabaceae (66 observations), Apocynaceae (47), Asteraceae (28), Lamiaceae (27), and Boraginaceae (12). Habitat for this species is not specific because the food plant genera used by Crotch bumble bee (*Antirrhinum*, *Phacelia*, *Clarkia*, *Dendromecon*, *Eschscholzia*, and *Eriogonum*) are widely distributed in different habitats (Koch et al. 2012:82).

2.10.3 Species Threats

Crotch bumble bee faces several threats including habitat modification, pesticides and herbicides, competition with managed bees, disease, and population dynamics (California Department of Fish and Wildlife 2019:18). Agricultural expansion and intensification, particularly within the Crotch bumble bee's core range in the Central Valley, as well as urbanization in southern California, have resulted in extensive habitat loss (Xerces Society for Invertebrate Conservation et al. 2018). Use of pesticides on agricultural crops is thought to be particularly detrimental to bumble bees, because pesticides not only affect target croplands, but also adjacent areas through pesticide drift, which may include natural habitats supporting bumble bees. Pesticides may kill bumble bees directly, and sublethal doses can affect the foraging and nesting behavior of bumble bees, reducing productivity (Thompson 2003; Desneux et al. 2007; Hopwood et al. 2012). Systemic insecticides, such as those containing neonicotinoids, accumulate in the pollen and nectar of flowers and thus affect adult bumble bee health and survival as well as larval development and survival, thereby reducing productivity of a colony. Herbicide may affect flowering plants on which Crotch bumble bee rely. Competition with honey bees has been shown to reduce foraging efficiency, worker size, and reproductive success of bumble bees (Goulson and Sparrow 2009; Thomson 2004). Honey bees may also transmit diseases to bumble bees at flowers (Klemens and Volmar 2006). A fungal pathogen (*Nosema bombi*) associated with commercial bumble bee production facilities has been implicated in the rapid and widespread declines in some bumble bees (Xerces Society for Invertebrate Conservation et al. 2018), though it has not been identified as a specific cause of Crotch bumble bee decline.

Crotch bumble bees colonial life cycle and mode of reproduction (i.e., single locus complementary sex determination system), results in reduced genetic diversity. Coupled with dramatic declines in range and relative abundance, the loss of genetic diversity can pose significant threats to small, isolated bumble bee populations (Whitehorn et al. 2009; Xerces Society for Invertebrate Conservation et al. 2018).

2.10.4 Suitable Habitat Definition

As described in Section 2.10.2, *Life History and Habitat Requirements,* suitable habitat for Crotch bumble bee consists of any of the natural or cultivated land cover types listed in Section 2.10.5, *Habitat Suitability Model*, where listed food plant genera, nesting sites, and overwintering sites are most likely to occur in the study area.

2.10.5 Habitat Suitability Model

Habitat suitable for nesting, foraging, and overwintering was modeled to include land cover types that provide food plant genera (e.g., *Antirrhinum, Phacelia, Clarkia, Dendromecon, Eschscholzia,* and *Eriogonum*) for nectar and pollen sources; overwintering sites that would not be inundated or saturated; and nest sites (Chapter 4, Figures 4.10-1 through 4.10-57). The plant genera used by Crotch bumble bee are widely distributed in different habitats (Koch et al. 2012:82); therefore, the habitat model for Crotch bumble bee includes grasslands, alkaline seasonal wetlands, vernal pool complex, and other seasonal wetlands. The underlying land cover data was unable to capture some specific areas where flower resources may occur, such as urban parks and gardens; thus, the species model may miss some areas where bumble bees may occur. However, these types of land use features are very unlikely to be affected by the project. Overall, the model is considered conservative and likely overestimates the extent of habitat in other portions of the study area (Chapter 4, Figures 4.10-1 through 4.10-57). Information on species habitat model development is included in Appendix 4B, Section 4B.5.1.1.

Modeled habitat includes the following types from the Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographic Information Center 2019), the Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographic Information Center 2018), and the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c; California Department of Water Resources 2021).

- Alkaline seasonal wetlands
    - All types

Modeled habitat also includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographic Information Center 2019), and the Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographic Information Center 2018).

- Grassland
    - All types

Modeled habitat includes the following types from the DCP Vernal Pool Complex (Witham et al. 2014; Chico State Research Foundation, Geographic Information Center 2019; California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c; California Department of Water Resources 2021).

- Vernal pool complex
    - All types

Modeled habitat also includes the following types from the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, California Department of Water Resources 2021):

- Other seasonal wetlands

2.11 Mason's Lilaeopsis

Mason's lilaeopsis (*Lilaeopsis masonii*) is state-listed as rare under the California Native Plant Protection Act (November 1979). It is not listed under the federal or California endangered species acts. Its Heritage Ranking in the CNDDB is G2/S2, which means that globally (G) and within the state (S), the species is considered imperiled (California Department of Fish and Wildlife 2020d:iii, 82).

The California Rare Plant Rank of 1B.1 for Mason's lilaeopsis indicates that it is rare, threatened, or endangered in California and elsewhere, and it is seriously endangered in California (California Native Plant Society 2020; California Department of Fish and Wildlife 2020d:iv, 82). Plants with a rank of 1B may meet the definitions of rare, threatened, and endangered as defined in CEQA Section 15380 (California Department of Fish and Wildlife 2020d:iv).

A taxonomic review of Mason's lilaeopsis concluded that Mason's lilaeopsis is genetically indistinguishable from the more common and widespread *Lilaeopsis occidentalis* and that the morphological differences observed between the coastal and inland forms are due to environmental plasticity (Fiedler et al. 2011:142). The report recommends not recognizing *L. masonii* as a separate species and recommends removing it from the state's list of rare plants. However, the paper acknowledges that *L. masonii* has been useful as an umbrella species for conservation planning efforts.

2.11.1 Geographic Distribution and Status

Mason's lilaeopsis is endemic to California and is known from 198 occurrences, all but one of which are presumed extant (California Department of Water Resources 2011; California Department of Fish and Wildlife 2020e). The range of Mason's lilaeopsis extends from Napa and Solano Counties in the north, to Contra Costa and Alameda Counties in the south, to Marin County in the west, and to Sacramento and San Joaquin Counties in the east (California Department of Fish and Wildlife 2020e).

Mason's lilaeopsis is found throughout the Delta along rivers and sloughs; the majority of known occurrences, 158, are within the study area (California Department of Water Resources 2011; California Department of Fish and Wildlife 2020e). Most occurrences are from the central and west Delta. In the south Delta, occurrences are predominately along Old River and Middle River. In the north Delta, Mason's lilaeopsis occurs in the Cache Slough complex and near Delta Meadows.

Over 300 stands of Mason's lilaeopsis were found during 2009 surveys (California Department of Water Resources 2011), including sites north of Prospect and Liberty Islands, in an almost 12-mile-long stand of plants along the banks of the Deep Water Ship Channel and scattered locations along the Yolo Bypass Toe Drain. Nineteen stands of Mason's lilaeopsis were found during 2010 surveys (California Department of Water Resources 2011). Mason's lilaeopsis was found in tidal freshwater emergent wetlands on the waterways between Webb Tract and Woodward Island, the south shore of Bacon Island, and the southeast corner of Fabian Tract on Old River. Twenty-six additional stands of Mason's lilaeopsis were found during 2011 surveys on in-channel islands, levees, and old wooden pilings along the South Mokelumne River north of Bouldin Island, San Joaquin River near Prisoner's Point on Mandeville Island, and Old River near Fay Island (California Department of Water Resources 2011:S-3).

2.11.2 Life History and Habitat Requirements

Mason's lilaeopsis is a semiaquatic perennial herb that blooms between April and November (Constance and Wetherwax 2012). It primarily spreads vegetatively by creeping rhizomes or by being dislodged and floating to new sites. Because it is a rhizomatous plant, the number of individuals in a population is difficult to determine. Thus, population size is often expressed as "several colonies" or in square feet. Reported colony sizes range from less than 1 square foot to 3,000 square feet (less than 1 square meter to 700 square meters)(California Department of Fish and Wildlife 2019).

Mason's lilaeopsis is found in brackish or fresh water habitats that are inundated by waves or tides, such as estuarine wetlands and immediately below the banks of tidal sloughs, rivers, and creeks (Golden and Fiedler 1991:5; Fiedler and Zebell 1993:6; California Department of Fish and Wildlife 2020e; California Native Plant Society 2020). It is a colonizing species that establishes on newly deposited or exposed sediments (California Native Plant Society 2020). Some reports suggest that Mason's lilaeopsis is not substrate specific because it is found in organic mucks, silty clays, and even pure sand throughout its range (Golden and Fiedler 1991:5). Other reports find that it has a preference for low tidal flats on clay or silty soils (Witham and Kareofelas 1994:11, 16, 19). It is usually found along the edges of tule marshes (Witham and Kareofelas 1994:16) and occasionally found distributed in soil pockets along riprap-lined levees (Golden and Fiedler 1991:5). It has been found in areas with high soil salinity, but those sites are not optimum habitat (Fiedler and Zebell 1993:33). Within the Delta, Mason's lilaeopsis is not found upstream from where tides affect water levels (Suisun Ecological Workgroup 1997:11).

Plant species commonly associated with Mason's lilaeopsis in the Delta include California tule (*Schoenoplectus californicus*), whorled marsh pennywort (*Hydrocotyle verticillata*), and low bulrush (*Isolepis cernua*) (Golden and Fiedler 1991:6-7). In the sloughs west of Liberty Island, at the south end of the Sacramento River Deep Water Ship Channel, Mason's lilaeopsis grows in a narrow band between the mudflats and mesic terrestrial vegetation. In Suisun Marsh and other places, Mason's lilaeopsis is predominantly associated with California tule, low bulrush, and three-ribbed arrowgrass (*Triglochin striata*) (Suisun Ecological Workgroup 1997:11); California Department of Fish and Wildlife 2020e). During the Delta Habitat Conservation and Conveyance Program 2009 to 2011 surveys, some of the species associated with Mason's lilaeopsis included hardstem bulrush (*Schoenoplectus acutus*), water iris (*Iris pseudacorus*), marshpepper (*Persicaria hydropiper*), giant reed (*Arundo donax*), whorled marsh pennywort, nutsedge (*Cyperus* sp.), iris-leaved rush (*Juncus xiphioides*), common buttonbush (*Cephalanthus occidentalis*), red willow (*Salix laevigata*), smooth beggartick (*Bidens laevis*), water pygmyweed (*Crassula aquatica*), Himalayan blackberry (*Rubus armeniacus*), common reed (*Phragmites australis*), sneezeweed (*Helenium puberulum*), Pacific aster (*Symphyotrichum chilense*), Santa Barbara sedge (*Carex barbarae*), common rush (*Juncus effusus*), seep monkeyflower (*Mimulus guttatus*), dallis grass (*Paspalum dilatatum*), and hedge false bindweed (*Calystegia sepium*) (California Department of Water Resources 2011:2-10, 4-3, 6-3).

2.11.3 Species Threats

The primary threat to Mason's lilaeopsis is the loss of marsh and shoreline habitat. In addition to human activity associated with fishing and hunting access posing a threat from trampling (Witham and Kareofelas 1994:11, 16, 19), the major threats to this species are considered to be habitat loss, invasions of nonnative species, and exposure to toxins. Some of the processes and activities that threaten this habitat include erosion, flood-management improvements (e.g., channel stabilization, levee maintenance and construction, dredging), dumping spoils, agriculture, recreation, and changes in water quality (California Native Plant Society 2020; California Department of Fish and Wildlife 2020c). A long-term threat is the stabilization of banks and mudflats due to highly regulated water flow regimes, which can cause floodplain habitat to be less dynamic (Fiedler and Zebell 1993:6, 19, 33). Successional changes in marsh vegetation, brought on by invasions of nonnative species such as water hyacinth (*Eichhornia crassipes*) to denser vegetation types or to types that can grow in the intertidal area pose an additional threat (California Native Plant Society 2020; Zebell and Fiedler 1996:47–48; California Department of Fish and Wildlife 2020c). Additionally, the practice of closing off or diking salt marshes limits the available habitat. Petroleum product spills could have a significant impact on tidal flat biota, and nonbiodegradable litter such as plastics could collect near the tidal drift line, inhibiting plant establishment and growth (Witham and Kareofelas 1994:11, 16, 19).

2.11.4 Suitable Habitat Definition

Mason's lilaeopsis grows within the upper tidal zone, at the interface between tidal waters and terrestrial vegetation. It grows in openings within many different vegetation alliances but also grows in unvegetated areas, such as on the riprap of levees and areas with waterside development.

2.11.5 Habitat Model Description

Mason's lilaeopsis grows within the upper tidal zone, at the interface between tidal waters and terrestrial vegetation. It grows in barren soil microsites within many different vegetation alliances but also grows in unvegetated areas, such as on the riprap of levees and areas with waterside development. Therefore, the habitat model for Mason's lilaeopsis is based primarily on the tidal perennial habitat land cover type, which includes the tidal channel habitat type of both the Delta Vegetation and Land Use Update and the DWR 2020 Aquatic Resources Delineation. Within the project area, Mason's lilaeopsis habitat was geographically defined as the area extending 10 feet landward from the boundary of the tidal channel land cover type. This area is expected to encompass the upper tidal zone that experiences daily tidal inundation and deposition of waterborne sediments. It encompasses many different vegetation types, but it also includes developed areas and levees where riprap has been placed. The extent of modeled habitat in the study area is depicted in Chapter 4, Figure 4.11-1. Information on species habitat model development is included in Appendix 4B, Section 4B.5.1.1.

A constraint layer was created in the geographic information system (GIS) to remove modeled habitat areas that were deemed unsuitable, including the interior of Clifton Court Forebay and, after inspection of aerial site photography, other areas such as boat docks and port facilities.

2.12 Boggs Lake Hedge-Hyssop

An analysis of take for this non-covered CESA-listed plant species is included to disclose potential impacts from tidal restoration mitigation actions described in the CMP (Appendix 5A). Tidal restoration mitigation actions would be covered by separate, site-specific ITP applications (as described in Appendix 5A).

Boggs Lake hedge-hyssop (*Gratiola heterosepala*) is listed as endangered under the CESA (Cal. Code Regs. tit. 14, § 670.2, subd. (a)(23) (A)). It is not listed under the federal ESA. Its Heritage Ranking in the CNDDB is G2/S2, which means that both globally and within the state it is considered imperiled, meaning at high risk of extinction due to very restricted range, very few populations (often 20 or fewer), steep population declines, or other factors making it vulnerable to extirpation (California Department of Fish and Wildlife 2020d; California Native Plant Society 2022a).

The California Rare Plant Rank of 1B.2 for Boggs Lake hedge-hyssop indicates that it is rare, threatened, or endangered in California and elsewhere. Its state threat level (.2) indicates that it is somewhat endangered in California (California Department of Fish and Wildlife 2020d; California Native Plant Society 2022a. Plants with rank of 1B are considered to meet the definitions of rare, threatened, or endangered as defined in the California Native Plant Protection Act (Fish & G. § 1901) or the CESA (Fish & G. §§ 2062, 2067) (California Department of Fish and Wildlife 2020d).

2.12.1 Geographic Distribution and Status

Boggs Lake hedge-hyssop occurs in northeastern California on the Modoc Plateau in Siskiyou, Modoc, Lassen, and Shasta Counties and also south in the Central Valley to Fresno County. There are also records from Boggs Lake in Lake County (California Department of Fish and Wildlife 2020c; California Native Plant Society 2022a). It is widely distributed throughout the range of vernal pool habitat (Barbour et al. 2007), including one occurrence in Oregon, where it is state-listed as threatened (California Native Plant Society 2022a). Population sizes range from small numbers to thousands of plants (California Department of Fish and Wildlife 2020c; California Native Plant Society 2022a). There are 89 known occurrences, two of which are extirpated and one of which is possibly extirpated (California Department of Fish and Wildlife 2020c).

Several populations have been reported to occur sporadically on and in the vicinity of Jepson Prairie Preserve and the Gridley Preserve, one of which is in the study area (Witham 2006; Barbour et al. 2007; California Department of Fish and Wildlife 2020c). There are no reported occurrences in the southwestern portion of the study area, but that area is within the species' range and potentially suitable claypan vernal pool habitat occurs there. No additional occurrences were found during focused surveys (California Department of Water Resources 2011).

2.12.2 Life History and Habitat Requirements

Boggs Lake hedge-hyssop is a semi-aquatic annual herb of the plantain family (Plantaginaceae) that is less than 10 centimeters (4 inches) tall and blooms from April to August (Estes 2012; California Native Plant Society 2022a). The lower portions of the brownish-red fleshy stems are glabrous, and the upper portions are glandular-sticky with puberulent hairs (Estes 2012). Leaves are small (less than 2 centimeters [0.8 inch]) and rounded at the tips. The predominantly yellow corolla has five lobes: two are yellow and fused; three are white and separate (Estes 2012). A related species that is much more common—bractless hedge-hyssop (*Gratiola ebracteata*)—can be distinguished from Boggs Lake hedge-hyssop by its more elongate and pointed sepals and mostly white flowers. Boggs Lake hedge-hyssop seeds can lie dormant in the soil for years, and the number of vegetative plants in a population can vary greatly from year to year (U.S. Fish and Wildlife Service 2005).

Boggs Lake hedge-hyssop has been reported from various habitats, including the edges of marshes and natural lakes, stock ponds, swales, and vernal pools (Witham 2006; Barbour et al. 2007; California Department of Fish and Wildlife 2020c). It has been observed in several types of vernal pools, including basalt flow, hardpan, claypan, and alkaline playa pools. In the study area, the species commonly associated with Boggs Lake hedge-hyssop include toothed downingia (*Downingia cuspidata*), dwarf downingia (*Downingia pusilla*), Fremont's goldfields (*Lasthenia fremontii*), and white-headed navarretia (*Navarretia leucocephala* subsp. *bakeri*) (Barbour et al. 2007; California Department of Fish and Wildlife 2020c).

2.12.3 Species Threats

Vernal pool loss through development, damage by intensive grazing, trampling, off-road vehicles, and invasive nonnative species are generally cited as threats (California Department of Fish and Wildlife 2020c; California Native Plant Society 2022a). Grazing generally has negative effects on this species, especially because of accompanying trampling (California Department of Fish and Wildlife 2020c).

2.12.4 Suitable Habitat Definition

Habitat for Boggs Lake hedge-hyssop includes vernal pool complex, degraded vernal pool complex, and alkali seasonal wetland complex.

Vernal pool complex habitat consists of vernal pools and uplands that display characteristic vernal pool and swale visual signatures that have not been significantly affected by agricultural or development practices.

2.12.5 Habitat Model Description

The Boggs Lake hedge-hyssop habitat model includes vernal pools, alkaline seasonal wetlands, and some seasonal wetlands. Vernal pool complexes in the western part of the study area often occur in a mosaic with alkaline seasonal wetlands; many of the species that occur in the vernal pool complex also occur in the alkaline seasonal wetland complex within this mosaic of natural communities. The modeled habitat relies on both aquatic resource delineation data that was collected for a smaller portion of the study area, in what is called the *delineation study area*, and suitable habitats found in the Great Valley Vernal Pool Data (Witham et al. 2014), Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update, and Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018, 2019). The extent of modeled habitat in the study area is depicted in Chapter 4, Figure 4.12-1. Information on species habitat model development is included in Appendix 4B, Section 4B.5.1.1.

Inside the Delineation Study Area

Modeled habitat includes the following types from the Great Valley Vernal Pool Data and DWR 2020 Aquatic Resources Delineation (Witham et al. 2014; California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, 2021).

- Vernal pool complex
    - Alkaline wetland
    - Vernal pool

Modeled habitat also includes the following types from the DWR 2020 Aquatic Resources Delineation (California Department of Water Resources and GEI Consultants Inc. 2020; California Department of Water Resources 2020c, 2021).

- Alkaline seasonal wetland complex
    - Alkaline wetland
- Other seasonal wetlands
    - Seasonal wetlands

Outside the Delineation Study Area

Modeled habitat includes the following types from the Great Valley Vernal Pool Data and Delta Vegetation and Land Use Update (Witham et al. 2014; Chico State Research Foundation, Geographical Information Center 2019).

- Vernal pool complex
  - All types

Modeled habitat also includes the following types from the Sand Hill Wind Repowering SEIR Land Cover Dataset (ICF 2018), East Bay RCIS 2017 Land Cover Dataset (ICF 2017a), Delta Vegetation and Land Use Update (Chico State Research Foundation, Geographical Information Center 2019), and Great Valley Ecoregion 2018 Vegetation Dataset (Chico State Research Foundation, Geographical Information Center 2018).

- Alkaline seasonal wetland complex
  - All types

Outside the delineation study area, density class information from the Great Valley Vernal Pool Data (Witham et al. 2014) was used to report an estimated wetted acre. This includes the following cover classes: <2%, 2%–5%, 5%–10%, >10%, and 100% for individual pools. In the statutory Delta, the cover classes reported go only as high as 5%–10%.

2.13 Colusa Grass

An analysis of take for this non-covered CESA-listed plant species is included to disclose potential impacts from tidal restoration mitigation actions described in the CMP (Appendix 5A). Tidal restoration mitigation actions would be covered by separate, site-specific ITP applications (as described in Appendix 5A).

Colusa grass (*Neostapfia colusana*) is federally listed as threatened (62 FR 14338-14352), and state-listed as endangered (Cal. Code Regs. tit. 14, § 670.2, subd. (a)(24)(A); California Department of Fish and Wildlife 2022). Colusa grass is known from the Central Valley with scattered occurrences from Colusa County to Merced County (Reeder 2012:1468). It grows in the bottoms of large, deep vernal pools (California Department of Fish and Wildlife 2020c). The natural community type in the study area that provides habitat for Colusa grass is vernal pool complex. Threats to Colusa grass are competition with nonnative plants, agriculture, development, overgrazing, and flood-management actions (California Native Plant Society 2022b).

The California Rare Plant Rank of 1B.1 for Colusa grass indicates that it is rare, threatened, or endangered in California and elsewhere. Its state threat level (.1) indicates that it is seriously threatened in California (California Department of Fish and Wildlife 2020d; California Native Plant Society 2022b). Plants with rank of 1B are considered to meet the definitions of rare, threatened, or endangered as defined in the California Native Plant Protection Act (Fish & G. § 1901) or the CESA (Fish & G. §§ 2062, 2067) (California Department of Fish and Wildlife 2020d).

2.13.1 Geographic Distribution and Status

Historically, Colusa grass occurrences are recorded from Colusa, Merced, Solano, Stanislaus, and Yolo Counties. The current distribution encompasses the same regions, with the majority of occurrences in the Southern Sierra Foothills Vernal Pool Region identified by the Vernal Pool Recovery Plan; one or two occurrences in the San Joaquin Valley Vernal Pool Region in Merced County, and four occurrences in the Solano-Colusa Vernal Pool Region in southeastern Yolo and central Solano Counties (U.S. Fish and Wildlife Service 2005:II-58). Colusa grass has apparently been extirpated from Colusa County. Three CNDDB records for Colusa grass are within the study area, which occur at Jepson Prairie west of Cache Slough Complex in Solano County and in Yolo County west of Yolo Bypass Wildlife Area (California Department of Fish and Wildlife 2020c).

2.13.2 Life History and Habitat Requirements

Colusa grass is an annual grass that grows in the deepest parts of large, deep vernal pools that retain water until May or June and flowers during summer months (U.S. Fish and Wildlife Service 2005:II-60). The species also occurs in the beds of intermittent streams and artificial ponds and also grows on pond margins (U.S. Fish and Wildlife Service 2005:II-61).

Long periods of inundation are required for seeds to germinate in late spring when little standing water is left in pools (U.S. Fish and Wildlife Service 2005:II-60–II-61). Colusa grass typically grows in single-species stands, but often co-occur with other vernal pool species in the same pond. The species requires clay, silty clay, or silty clay loam soils with underlying impermeable layers ranging from claypan to lime-silica or iron-silica cemented hardpan and tuffaceous alluvium (U.S. Fish and Wildlife Service 2005:II-62).

2.13.3 Species Threats

Habitat loss generally is a result of agricultural conversion from rangelands to intensive farming, urbanization, aggregate mining, infrastructure projects (such as roads and utility projects), and recreational activities (such as off-highway vehicles and hiking) (U.S. Fish and Wildlife Service 2005:I-1, I-16–I-28). Habitat fragmentation occurs when vernal pool complexes are broken into smaller groups or individual vernal pools and become isolated from each other as a result of activities such as road development and other infrastructure projects (U.S. Fish and Wildlife Service 2005:I-1, I-17).

Inappropriate grazing practices have also been identified as a threat; these include inappropriate timing and intensity of grazing, as well as elimination of grazing in areas where nonnative grasses dominate the uplands surrounding vernal pools. Appropriate grazing regimes help control nonnative plants such as Italian ryegrass (*Lolium multiflorum*) and waxy mannagrass (*Glyceria declinate*), which, if unchecked, can increase thatch buildup, decrease ponding duration, and decrease the aquatic habitat available to Colusa grass (U.S. Fish and Wildlife Service 2012b).

Human disturbances and changes in land use practices can alter the hydrology of temporary waters and result in a change in the timing, frequency, or duration of inundation in vernal pools, which can create conditions that render existing vernal pools unsuitable for vernal pool species (U.S. Fish and Wildlife Service 2005:I-1, I-16, I-17, I-23 – II-24).

Climate change is expected to affect vernal pool hydrology through changes in the amount and timing of precipitation inputs to vernal pools and the rate of loss through evaporation and evapotranspiration. It is unknown at this time whether climate change in California will result in a localized, relatively small cooling and drying trend, or a warmer trend with higher precipitation events. However, it is possible that either scenario would result in negative effects on vernal pool plant species. Cooling and drying trends could adversely affect the Colusa grass through decreased inundation periods that do not allow the species sufficient time to complete its life cycle. In contrast, warmer conditions could increase inundation periods, which would not necessarily be a negative effect because increased inundation periods would increase available habitat for Colusa grass (U.S. Fish and Wildlife Service 2012b).

Specific threats to Colusa grass habitat identified in the 2005 vernal pool recovery plan included the following.

- Contamination from inundation by poultry manure, herbicide applications, and contamination of groundwater by industrial chemicals.
- Habitat conversion, including agricultural conversion, especially in Stanislaus County; urbanization, especially at the proposed University of California campus and associated community development in eastern Merced County, proposed construction of a new prison and a landfill, a proposed flood control project in eastern Merced County, and runoff alterations are a threat to the two Yolo County occurrences.
- Almost all of the extant occurrences of *Neostapfia colusana* are subject to livestock grazing, thus to the extent inappropriate grazing practices are still being followed at certain sites, these sites may be threatened.
- Competition from invasive native and nonnative plants, especially in combination with adverse hydrology changes and adverse grazing practices.
- Vandalism (i.e., trampling near urban areas) and foraging by grasshopper outbreaks.
- Small population size at sites with fewer than 100 individuals.

2.13.4 Suitable Habitat Definition

As described above in Section 2.13.2, *Life History and Habitat Requirements*, suitable habitat for Colusa grass includes large vernal pools that provide appropriate hydroperiod and conditions for germination.

2.13.5 Habitat Model Description

The Colusa grass habitat model is the same as that for Boggs Lake hedge-hyssop (Chapter 4, Figure 4.12-1). Please see Section 2.12.5, *Habitat Model Description*, for a description of the model.

2.14 Solano Grass

An analysis of take for this non-covered CESA-listed plant species is included to disclose potential impacts from tidal restoration mitigation actions described in the CMP (Appendix 5A). Tidal restoration mitigation actions would be covered by separate, site-specific ITP applications (as described in Appendix 5A).

Solano grass (*Tuctoria mucronata*) was listed as endangered throughout its range under the ESA on September 29, 1978 (43 FR 44810). In March 2009, USFWS published a 5-year review recommending that the species remain listed as endangered. Solano grass is also state-listed as endangered (Cal. Code Regs. tit. 14, § 670.2, subd. (a)(24(I); California Department of Fish and Wildlife 2022).

The California Rare Plant Rank of 1B.1 for Solano grass indicates that it is rare, threatened, or endangered in California and elsewhere. Its state threat level (.1) indicates that it is seriously threatened in California (California Department of Fish and Wildlife 2020d; California Native Plant Society 2022c). Plants with rank of 1B are considered to meet the definitions of rare, threatened, or endangered as defined in the California Native Plant Protection Act (Fish & G. § 1901) or the CESA (Fish & G. §§ 2062, 2067) (California Department of Fish and Wildlife 2020d).

2.14.1 Geographic Distribution and Status

Solano grass is known from only three occurrences in the southwestern Sacramento Valley in Solano and Yolo Counties, where it grows in vernal pools (California Department of Fish and Wildlife 2020c).

2.14.2 Life History and Habitat Requirements

Solano grass is an annual grass that grows in alkaline playas, intermittent lakes, and "relatively small" vernal pools. Seeds are presumed to germinate in May or June and the species flowers in June and sets seeds in July (U.S. Fish and Wildlife Service 2005:II-107).

Solano grass requires turbid northern claypan vernal pools within annual grassland. The species often co-occur with other vernal pool species in the same vernal pool complex (U.S. Fish and Wildlife Service 2005:II-108).

2.14.3 Species Threats

Habitat loss generally is a result of agricultural conversion from rangelands to intensive farming, urbanization, aggregate mining, infrastructure projects (such as roads and utility projects), and recreational activities (such as off-highway vehicles and hiking) (U.S. Fish and Wildlife Service 2005:I-1, I-16–I-28). Habitat fragmentation occurs when vernal pool complexes are broken into smaller groups or individual vernal pools and become isolated from each other as a result of activities such as road development and other infrastructure projects (U.S. Fish and Wildlife Service 2005:I-1, I-17).

Inappropriate grazing practices have also been identified as a threat; these include inappropriate timing and intensity of grazing, as well as elimination of grazing in areas where nonnative grasses dominate the uplands surrounding vernal pools. Appropriate grazing regimes help control nonnative plants such as Italian ryegrass (*Lolium multiflorum*) and waxy mannagrass (*Glyceria declinate*), which, if unchecked, can increase thatch buildup, decrease ponding duration, and decrease the aquatic habitat available to Solano grass (U.S. Fish and Wildlife Service 2012b).

Human disturbances and changes in land use practices can alter the hydrology of temporary waters and result in a change in the timing, frequency, or duration of inundation in vernal pools, which can create conditions that render existing vernal pools unsuitable for vernal pool species (U.S. Fish and Wildlife Service 2005:I-1, I-16, I-17, I-23, II-24).

Climate change is expected to affect vernal pool hydrology through changes in the amount and timing of precipitation inputs to vernal pools and the rate of loss through evaporation and evapotranspiration. It is unknown at this time whether climate change in California will result in a localized, relatively small cooling and drying trend, or in a warmer trend with higher precipitation events. However, it is possible that either scenario would result in negative effects on vernal pool invertebrate species. Cooling and drying trends could adversely affect the Solano grass through decreased inundation periods that do not allow the species sufficient time to complete its life cycle. In contrast, warmer conditions could increase inundation periods, which would not necessarily be a negative effect because increased inundation periods would increase available habitat for Solano grass (U.S. Fish and Wildlife Service 2012b).

Specific threats to Solano grass habitat identified in the 2005 vernal pool recovery plan included the following.

- Habitat degradation from discing, excavation, trampling by hunters, inappropriate grazing practices, and contamination from herbicide runoff, application of salt, and contamination of groundwater from industrial contaminants.
- Competition from aggressive plant species.
- Overcollection.
- Small population size.

2.14.4 Suitable Habitat Definition

As described above in Section 2.14.2, *Life History and Habitat Requirements*, suitable habitat for Solano grass includes vernal pools that provide appropriate hydroperiod and conditions to complete its life cycle.

2.14.5 Habitat Model Description

The Solano grass habitat model is the same as that for Boggs Lake hedge-hyssop (Chapter 4, Figure 4.12-1). Please see Section 2.12.5, *Habitat Model Description*, for a description of the model.

2.15 References

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2.15.2 Personal Communications

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Meese, R. J. [b]. Staff Research Associate, Department of Environmental Science and Policy (SEP) and Information Center for the Environment (ICE), University of California, Davis. August 19, 2020—Transmittal of tricolored blackbird portal GIS dataset.

Reece, Kevin. Senior Environmental Scientist. Regulatory Compliance, Division of Integrated Science and Engineering, California Department of Water Resources, West Sacramento, CA. November 8 and 19, 2021—Emails containing Excel files <WY19_Salvage_Genetic Results.xlsx> and <WY20-WY21_Salvage_Genetic Results.xlsm> sent to Marin Greenwood, Aquatic Ecologist, ICF, Sacramento, CA.

Wunderlich, Veronica. Senior Environmental Scientist, Division of Integrated Science and Engineering, California Department of Water Resources, West Sacramento, CA. January 12, 2024 – email to Danika Tsao, Senior Wildlife Biologist, ICF regarding giant garter snake occurrence from Jersey Island. ICF, Sacramento, CA.