Appendix F
Species Account

F1.1 Steelhead Trout (*Oncorhynchus mykiss)*

F1.1.1 Legal Status

F1.1.1.1 State

California state species protection status listings are governed by the California Endangered Species Act (CESA). Steelhead are not listed under the CESA.

F1.1.1.2 Federal

All steelhead (the anadromous form of *Oncorhynchus mykiss*) in the study area belong to the South-Central California Coast steelhead (SCCCS) Distinct Population Segment (DPS), which is federally listed as threatened (62 Federal Register [FR]: 43937-43954) under the federal Endangered Species Act (ESA). In 2016, the National Marine Fisheries Service (NMFS) completed a 5-year status review of the SCCCS DPS and recommended that it remain classified as a threatened species. The SCCCS DPS includes all naturally spawned anadromous steelhead populations below natural and human-made impassable barriers in streams from the Pajaro River (inclusive) to, but not including, the Santa Maria River, California.

F1.1.1.3 Critical Habitat

Section 3 of the ESA (16 U.S.C. 1532(5)) defines *critical habitat* as

> "(i) the specific areas within the geographical area occupied by the species, at the time it is listed on which are found those physical or biological features (I) essential to the conservation of the species and (II) which may require special management considerations or protection; and (ii) specific areas outside the geographical area occupied by the species at the time it is listed."

The freshwater primary constituent elements (PCEs) that define the physical and biological features of steelhead critical habitat are (1) spawning habitat, including spawning substrate, and adequate water quantity and quality; (2) freshwater rearing habitat including floodplain connectivity and natural escape and velocity cover; and (3) freshwater migration corridors free of obstructions, with water quantity and quality conditions that allow movement (70 FR 52488–52627).

Critical habitat was designated for all steelhead populations across California in 2005 (70 FR 52488; Figure F1-1). Critical habitat for SCCCS in the Salinas River watershed was designated from the mouth of the Salinas River upstream to 7.5 miles below the Santa Margarita Lake, as well as the Arroyo Seco River, Nacimiento River (below the Nacimiento Dam), San Antonio River (below the San Antonio Dam), and the upper Salinas River tributaries (70 FR 52488–52627; Figure F1-2). The PCEs of critical habitat for steelhead in each subbasin of the Salinas River are listed in Table F1-1.

Table F1-1. Number of Stream Miles Designated as Critical Habitat for South-Central California Coast Steelhead within Selected Subbasins of the Salinas River Watershed

Subpopulation	Spawning	Rearing	Migration
Arroyo Seco	68.5	68.5	84.6
San Antonio/Nacimiento	20.6	20.6	20.6
Upper Salinas	21.1	40.2	48.1
Lower Salinas	2.4	9.0	149.1

F1.1.2 Taxonomy

The taxonomic history and nomenclature of steelhead is convoluted and has been modified several times throughout history. The species is commonly known by its colloquial names, *trout* and *rainbow trout*, although it has been described with at least 22 scientific names in five genera (Scott and Crossman 1973). Until 1989, the primary scientific name used for steelhead from western North America was *Salmo gairdneri*. However, it was shown that steelhead were more similar to Pacific salmon (*Oncorhynchus*) than to Atlantic salmon (*Salmo*), and that *Salmo gairdneri* was the same species as the previously described *Salmo mykiss* (Smith and Stearley 1989). Therefore, the scientific name *Oncorhynchus mykiss* was adopted for steelhead and rainbow trout in 1989. Rainbow trout are found in freshwater and do not migrate out to the ocean, while steelhead are anadromous and migrate out to the ocean and return to freshwater to spawn.

F1.1.3 Distribution

F1.1.3.1 State

Historical

Steelhead were one of the most widely distributed species in the world. Within California, they were historically found along the entire coast and inland in the Sacramento and San Joaquin River drainages (Moyle 2002; Figure F1-1). The historical distribution also included most southern California streams to the United States–Mexico border and into Baja California.

Recent

Steelhead are currently found throughout coastal California and the Sacramento and San Joaquin River drainages of the Central Valley. However, there is a limited distribution within southern California streams; due to water infrastructure development and climate change, many populations have been extirpated or are present only as remnant populations with occasional runs of diminished size (National Marine Fisheries Service 2012).

F1.1.3.2 Study Area

Historical

The Salinas River watershed is the largest coastal watershed contained entirely within California and contains two subbasins: the lower Salinas River watershed, which includes the Gabilan Creek and Arroyo Seco River watersheds, and the upper Salinas River watershed, which includes the San Antonio, Nacimiento, and Estrella River watersheds (National Marine Fisheries Service 2013).

Steelhead were historically observed throughout both subbasins, and the population was largely supported by spawning and rearing habitat in the upper Salinas, Nacimiento, San Antonio, and Arroyo Seco Rivers (California Department of Fish and Game 1965). The Salinas River watershed historically provided approximately 98.9 stream miles of available habitat for steelhead (Becker et al. 2010). However, the mainstem Salinas River likely provided poor spawning and rearing habitat due to its muddy and sandy substrate, although it was and is an essential migration corridor to quality spawning and rearing habitat in the tributaries (Titus et al. 2002).

Recent

The Salinas River watershed is currently estimated to provide approximately 55.0 stream miles of available habitat for steelhead (Becker et al. 2010). The Arroyo Seco River contains the majority of spawning habitat in the basin and half of the rearing habitat (National Marine Fisheries Service 2007). Water infrastructure development, land use, and water management practices have gradually reduced available habitat and instream flows in the watershed. Dams on the upper mainstem of the Salinas, Nacimiento, and San Antonio Rivers have blocked access to historical spawning and rearing habitat. The majority of steelhead are now confined to the Arroyo Seco River due to its relatively close proximity to the Pacific Ocean. While resident individuals of steelhead persist in the upper Salinas River watershed (including above the Nacimiento and San Antonio Dams) (Titus et al. 2002), several factors, including Salinas River flows, currently prohibit this subset of steelhead from contributing to the overall steelhead population in the Salinas River watershed.

F1.1.4 Natural History

F1.1.4.1 Habitat Requirements

Steelhead are largely found in cool, clear, fast-flowing rivers and streams containing numerous riffles and cover (Moyle 2002). While these waterways are generally forested, snow-fed streams, steelhead trout are also found in rain-fed, intermittent streams in central California (Boughton et al. 2009). Water temperature is an important habitat factor. Optimal growth occurs at 15–18 degrees Celsius (°C), and mortality typically results at 24–27°C, although new research is revealing populations of trout that are sustaining life in conditions previously considered lethal (Moyle 2002; Verhille et al. 2016; Poletto et al. 2017). Myrick and Cech (2004) found optimal temperatures to be 7–10°C for eggs and alevin and 1–25°C for juveniles, with optimal growth occurring at 19°C. Thermal refugia, or areas with cooler temperatures, such as confluence pools, are important for maintaining populations in warmer streams (Sutton et al. 2007). Steelhead are typically found in streams with dissolved oxygen concentrations above 6.5 milligrams per liter (mg/L) (near saturation levels, in many cases), although they can survive at levels as low as 1.5–2.0 mg/L for short periods of time (Davis 1975; Mathews and Berg 1997).

Streambed substrate is an important habitat factor as spawning occurs in places where the streambed is composed of gravelly substrate and fast-moving water, usually in riffles or pool tails. Gravel sizes of 1–13 centimeters (cm) are generally preferred for egg-laying redds (Moyle 2002). Substrate size is correlated with steelhead growth, and spawning bed enhancement can improve embryo survival (Merz et al. 2004).

Stream cover is another key habitat feature, with overhanging riparian vegetation and instream woody debris shown to be an essential component of juvenile rearing habitat (Shirvell 1990; Bugert et al. 1991; Quinones and Mulligan 2005; Thompson et al. 2012). Juvenile fry often have poor swimming ability and, as a result, they move into shallow, low-velocity areas in side channels and along channel margins to escape high velocities and predators (Everest and Chapman 1972). Juveniles progressively move toward deeper water as they grow (Bjornn and Reiser 1991). The presence of large woody debris in streams has also been shown to be especially important for pool formation (Thompson et al. 2012). Steelhead tend to use riffles and other habitats not strongly associated with cover during summer rearing more than other salmonids.

To complete the migratory phase of their life cycle, steelhead require connectivity with the ocean during several time periods of the year. Habitat conditions in the Salinas River Lagoon are generally not suitable for steelhead spawning or egg incubation but potentially support rearing when conditions are right. When the river mouth is open, the lagoon is tidally influenced and sustains saltwater conditions, and migration to and from the ocean is possible. When the river mouth is closed, the lagoon is typically freshwater with good water quality conditions, specifically when Salinas River inflow is adequate, and no saltwater intrusion occurs. However, during these semi-lentic periods, stratification of the lagoon may occur, with a solute-rich and oxygen-depleted stratum of water on the bottom of the channel (hypolimnion), which is not suitable for rearing juveniles in certain locations. When the water in the estuary is stratified, the water in the top layer (epilimnion) may provide available rearing habitat for steelhead, although elevated temperatures and low dissolved oxygen levels can occur here as well. Accordingly, the lagoon is believed to be used primarily as a migration corridor by adult and juvenile steelhead (Denise Duffy and Associates 2015).

F1.1.4.2 Movement

O. mykiss generally have one of two distinct life patterns: resident rainbow trout and sea-run or anadromous steelhead. Some resident rainbow trout do migrate within a river system for the purpose of spawning or foraging; however, most rainbow trout often spend their entire lives within a few hundred meters of stream or within the same lake (Moyle 1976).

Steelhead life history strategies are the most variable of all salmonids, and times spent in freshwater and in the ocean vary according to geography, life history patterns, and effects of natural phenomena and abiotic and biotic factors. Most individuals spend 1–3 years in fresh water and 1–4 years in the ocean before returning to fresh water to spawn (Shapovalov and Taft 1954; Barnhart 1986; Busby et al. 1996; McEwan 2001). While in the ocean, steelhead probably do not range too far from the coast, although ocean catch data are limited (Moyle 2002). Most anadromous salmonids (e.g., Chinook salmon [*O. tshawytscha*]) die after spawning, but steelhead are iteroparous, meaning they may survive to spawn more than once. Steelhead may spawn up to four times per life span; however, of the steelhead that spawn multiple times, 70–85% spawn only twice (Barnhart 1986).

Adult steelhead migrate to fresh water between November and June, often peaking in February. Adult escapement monitoring in the Salinas River watershed has revealed highly variable timing of upstream migration, which has occurred as early as the first half of December and as late as the end of March (Table F1-2). Adult migration generally occurs after periods of high flow, and only when the lagoon has previously breached. Spawning begins shortly after adult fish reach spawning areas. Most of the spawning in the Salinas River watershed occurs in the tributary rivers and streams.

After a period of 1 or more years, juvenile steelhead undergo the biological process of *smoltification* in which juvenile salmonids become physiologically adapted for downstream migration and entry into saltwater. Smoltification may commence sometime in mid- to late winter as juvenile steelhead become fully ready to make the migration sometime in spring. In California, the outmigration of steelhead smolts typically begins in March and ends in late May or June (Satterthwaite et al. 2009). In the Carmel River (a coastal river), most juvenile steelhead migrate to the ocean between April and June (Snider 1983). This is the typical period for the smolt migration of steelhead in coastal watersheds along the western United States (Busby et al. 1996). Younger juveniles and those that have not undergone smoltification may disperse downstream and rear in mainstem, estuarine, and lagoon habitats. Juvenile steelhead often migrate downstream in search for available habitat, leading to significant percentages of the juvenile population rearing in coastal lagoons and estuaries (Bjornn 1971; Shapovalov and Taft 1954; Zedonis 1992; Hayes et al. 2008). This adaptation of rearing in coastal lagoons and estuaries prior to smoltification is thought to be an important component of steelhead life history at a time when physiological adaptation, foraging, and refugia from predators are critical (Healey 1982; Simenstad et al. 1982).

Downstream outmigration monitoring in the Salinas River watershed has revealed that juvenile outmigration peaks as a result of increased stream flow and turbidity associated with storm events (Figure F1-3). This relationship is particularly apparent on the Arroyo Seco River, owing to the larger number of downstream migrants relative to other trapping locations. Notably, it appears that juvenile steelhead in the Salinas River watershed are able to initiate downstream migration in response to increases in flow, irrespective of month. Whereas in other river systems with more constant flow, outmigration of juvenile steelhead can occur during all months of the year. To cope with this challenge, steelhead in the watershed appear to respond well to environmental cues, though these cues may occur outside the currently monitored timeframe. Outmigration monitoring using a rotary screw trap typically takes place from early March until late May, and inspection of annual flow and migration patterns, particularly in the Arroyo Seco River, reveals that emigration is likely to occur before and after this period (evidenced by documentation of steelhead as early as the first day of monitoring and as late as the last day of monitoring). Based on scale analysis of captured individuals, steelhead in the Salinas River watershed appear to migrate at four different ages (young of the year, 1+, 2+; and 3+; Figure F1-4). The majority of fish migrated downstream at Age 1 (67%), followed by Age 2 individuals (30%), Age 3 individuals (2%) and a handful of young of the year (FISHBIO, unpublished data).

The mainstem Salinas River is a migration corridor for adult steelhead migrating from the ocean to spawn in tributaries (National Marine Fisheries Service 2007Kelts (adults that have just spawned), smolts, and juveniles use the river to migrate downstream to the ocean or lagoon (National Marine Fisheries Service 2007Before the Nacimiento and San Antonio Reservoirs were constructed, the Salinas River had little or no summertime flow, in part because of an imbalance between the rate of groundwater withdrawal from pumping and recharge from natural flows (National Marine Fisheries Service 2003r in the mainstem Salinas River, which is currently limited by the availability of adequate flows to provide passage over long distances to suitable spawning and rearing habitat (National Marine Fisheries Service 2007. Adequate migration flows vary annually due to changes in channel geometry, although g levees, channel maintenance, road crossings, and removal of riparian vegetation have reduced the availability and quality of migration habitat for steelhead (National Marine Fisheries Service 2007; Monterey County Water Resources Agency 2013.Monterey County Water Resources Agency 2013).

Age and Growth analysis of captured individuals in the Arroyo Seco provides evidence that juvenile production can occur even in years (winters) without connectivity to the marine environment (i.e., no breaching of the lagoon's sandbar). Three individuals collected in spring 2017 were determined to belong to year classes 2015 (n=1) and 2016 (n=2). This is a clear indication that *O. mykiss* in the

Salinas River basin exhibit a resident or partially migratory life history, permitting population persistence during extended periods (multi-year) of isolation from the marine environment.

Table F1-2. South Central California Coast Steelhead Life History in the Salinas River

Steelhead		Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec
Adult Migration	Pacific Ocean to Salinas River and Tributaries												
Spawning	Upper Salinas Tributaries												
Egg Incubation	Upper Salinas Tributaries												
Juvenile Rearing	Upper Salinas Tributaries												
Juvenile Movement[a]	Upper Salinas Tributaries to Salinas River and Pacific Ocean												

[a] Juvenile movement may occur outside of the indicated months due to increased stream flows, but studies are only conducted from early March to May; therefore, there are no data to support movement during other months.

F1.1.4.3 Ecological Relationships

The Salinas River watershed subpopulation of steelhead resides in an inland ecoregion, which is typified by drier and warmer conditions than the coastal region. This population also has longer migration routes and differing hydrologic regimes, which confer unique selective regimes that likely supported and may still support unique life history traits that have allowed these steelhead to persist in this ecoregion. Fish surviving in this environment need to possess the ability to migrate longer distances under more variable hydrologic conditions than in shorter, wetter coastal areas and the ability to acclimate to warmer water temperatures. Lastly, they likely display increased plasticity between anadromous and resident forms of steelhead, as this permits them to better survive periodic drought conditions when reduced flows in the mainstem prevent migration to and from the ocean. The retention of these traits within the DPS may take on added importance as climate conditions increase the likelihood of serious droughts, which is expected (see section 3.5.9 for details on climate change). Historically, different geographic and life history components that were minor producers during one climatic regime have dominated during others. Hilborn et al. (2003) used this observation to demonstrate that the bio-complexity of fish stocks is critical for maintaining their resilience to environmental change (i.e., portfolio effect), and it is likely that this resilience is important for steelhead populations in the Salinas River.

Migratory behavior of adult steelhead is of particular interest in the Salinas River as monitoring has shown opportunistic migration of adult fish at all times of the year, which is an unusual life history strategy for steelhead populations (FISHBIO in prep). Escapement monitoring has revealed that adult steelhead migration into the lagoon coincides with or occurs after periods of increased flow, and only in years the lagoon is connected to the ocean. However, a prolonged amount of time can lapse between the lagoon disconnecting from the ocean and the first migratory adult steelhead observed at the weir (located at river mile 2.75), resulting in uncertainty about the life history of adult steelhead migration. Notably, in 2011–2012, the lagoon was closed during the period of weir operation, but it remained open the previous summer and until September 21, 2011. It is unknown if steelhead passages in early 2012 are attributable to fish that reared in the lagoon environment or fish that had entered the lagoon from the ocean before the sandbar closed. Although low levels of abundance of steelhead in the lagoon and lack of documentation of adult fish in the lagoon habitat suggest that rearing may occur in the lagoon, this somewhat unusual behavior is likely better explained by ocean maturation and temporary staging in the lagoon until flow conditions improved. No upstream passages have been documented in years when the lagoon has not breached. However, delayed migration—sometime after the lagoon has closed—has been noted following an initial increase in upstream passage. This suggests that steelhead exhibiting ocean-run life history traits may opportunistically enter the lagoon when it is connected to the ocean, and they commence upstream migration after a staging period in the lagoon that may last up to several weeks.

Steelhead exhibiting estuary-life history traits may opportunistically enter the lagoon as a juvenile or subadult and reside in the lagoon over summer if environmental and water quality conditions permit. The two life-history strategies could explain what appears to be two different migration periods in some years. For example, an initial migration period occurred in December 2012 and January 2013 when the lagoon was connected to the ocean and may have been composed primarily of ocean-run steelhead, and a second migration period occurred in February and March 2013 when the lagoon was closed, and which may have been comprised primarily of estuary-run steelhead (Figure F1-3).

Based on watershed size, location, ecological context, and overall status of SCCCS, a viable steelhead population in the Salinas River has the potential to ameliorate the overall extinction risk of the DPS, because it lessens fragmentation in the distribution of SCCCS and contributes to the genetic diversity of the species. The Nacimiento River and San Antonio River subpopulations (part of the upper Salinas River) are two of the three populations at highest risk of extirpation in the SCCCS DPS. If the Salinas River watershed subpopulations were lost, the only remaining subpopulations in the interior ecoregion would be those of the Pajaro River basin. Extinction risk profiles suggest that habitat loss has been acute in the Pajaro River basin and that the subpopulations' abundance, distribution, growth rate, and genetics are in poor condition. The risk of losing the entire inland geographic area inhabited by SCCCS is high. Thus, as a substantial component of the inland ecoregion, the Salinas subpopulations are important to the conservation of ecological diversity of the SCCCS DPS.

Wild populations generally have some degree of genetic population structure based on biogeographic patterns that range from complete genetic isolation to free genetic exchange. These biogeographic patterns have important implications for genetic management and extinction risk because they are often altered by human actions that can affect fitness and local adaptation (Meffe and Carroll 1997). The spatial relationship between subpopulations in the SCCCS DPS is one of increasing isolation. This combined with declines in abundance is leading to the imminent loss of four of the 12 subpopulations (National Marine Fisheries Service 2013).

Steelhead, like many spatially structured species, exhibit some degree of metapopulation dynamics, whereby local populations are connected through migration corridors and productivity of any given local population may be the result of local habitat conditions and/or level of migration with other populations in the metapopulation. This means that replacement of individuals to sustain the population is achieved either by reproduction from within the population or immigration from outside source populations. For SCCCS subpopulations, straying between subpopulations is an important factor in maintaining metapopulation structure (Hill et al. 2002; Keefer and Caudill 2012). In the SCCCS DPS, NMFS (2013) concluded that no current SCCCS subpopulation has the requisite viability to function as a source population for migrants. Although some exchange of strays may still occur at low levels, the role of strays for bolstering population size has been greatly diminished across the entire DPS. Lack of migrant sources adds demographic and genetic risk to the DPS.

Connectivity between subpopulations is an important factor affecting gene flow and recolonization potential (Good et al. 2005) and is influenced by migration distance and ease of migration. Patterns of isolation by distance are reflected in genetic signatures of multiple steelhead populations along the California coast (Garza et al. 2014), suggesting that the greater the migration distance, the less reproductive interaction occurs between subpopulations. While challenges to migration do not preferentially deter straying, they do reduce the success of any adult attempting to migrate and increase the degree of isolation. Demographic and genetic connectivity among subpopulations in the Salinas River watershed (i.e., upper Salinas, San Antonio/Nacimiento, and Arroyo Seco Rivers) is important for maintaining the Salinas watershed populations as a whole and for preventing erosion of genetic diversity.

Steelhead populations of the Salinas River watershed play a significant role in the survival of the SCCCS DPS because (1) they represent a large distributional component of the overall range of the DPS, (2) they inhabit ecologically distinct areas unique to the DPS, and (3) they exhibit unique life history traits (National Marine Fisheries Service 2013). To be considered viable, a DPS should contain multiple subpopulations, maintain wide geographic distribution, and contain subpopulations that display diverse life-histories and phenotypes (McElhany et al. 2000). These Salinas River watershed populations contribute to all three of these viability criteria.

The loss of the populations in the Salinas River watershed would mean the removal of the largest diverse populations of SCCCS in the entire DPS. In terms of watershed acreage and stream miles, the Salinas River is the largest river in the DPS, comprising approximately 48% of the DPS in terms of acreage and total stream miles (National Marine Fisheries Service 2007). Currently, the Salinas River watershed has approximately 19% of the DPS in terms of miles of occupied spawning and/or rearing habitat (Table F1-3). Of the five larger watersheds in the DPS, the Salinas River has the most occupied habitat remaining. Without the Salinas River watershed population, only smaller coastal populations and the Pajaro River basin populations would remain, and the total amount of occupied habitat in the DPS would be reduced by nearly 20% (Figure F1-5).

Table F1-3. Miles of Occupied Stream Habitat within Watersheds of the South-Central California Coast Steelhead Distinct Population Segment

Watershed	Currently Occupied Habitat (miles)	Proportion of Occupied Habitat in the Distinct Population Segment
Salinas River	149	19%
Pajaro River	144	18%
Carmel River	92	11%
Big Sur	36	4%
Little Sur	15	2%
Small Coastal Streams	368	46%
Source: Adapted from National Marine Fisheries Service 2007		

F1.1.5 Population Status and Trends

F1.1.5.1 Population Trend

Distinct Population Segment

Limited available data on current steelhead abundance suggest the overall population in the SCCCS DPS is extremely small. Estimating the trend in population size is difficult because the run size for most watersheds is unknown and major impacts (i.e., dams) leading to subsequent declines occurred prior to most modern fish investigations in the SCCCS DPS. The sporadic and intermittent presence of steelhead in many watersheds in the SCCCS DPS further confounds assessment efforts. Nonetheless, investigations conducted since 1996 (Busby et al. 1996; Boughton et al. 2006) indicate that of the 39 watersheds that historically supported anadromous runs, virtually all continue to be occupied by native steelhead, though most of the populations are at historically low levels (National Marine Fisheries Service 2013).

Status reviews indicate that steelhead populations in the region have declined dramatically from about 27,000 fish estimated at the turn of the century (Busby 1996; Good et al. 2005; Williams et al. 2011). In the mid-1960s, California Department of Fish and Game (CDFG; California Department of Fish and Game 1965) estimated that the DPS-wide run size was about 17,750 adults. No recent estimates of the population have been made at the DPS scale; however, estimates for five river systems within the DPS (Pajaro, Salinas, Carmel, Little Sur, and Big Sur) indicate runs of fewer than 500 adults. Previous estimates of run sizes in those rivers had been on the order of 4,750 adults (California Department of Fish and Game 1965). Time-series data for the DPS only exist for the Carmel River, and indicate a decline of 22% per year from 1963 to 1993. More recent data from the Carmel River indicates that the abundance may have increased slightly, although it is difficult to determine whether this reflects population growth or a limited data set (Good et al. 2005).

The recovery plan for the SCCCS DPS estimated the recovery potential of the population to be low based on (1) a small number of extant populations vulnerable to extirpation due to loss of accessibility to freshwater spawning and rearing habitat; (2) low abundance; (3) degraded estuarine habitats; and (4) altered watershed processes essential to maintain freshwater habitats (National Marine Fisheries Service 2013). Threats are expected to be of a moderate magnitude in smaller watersheds, with a higher risk in larger watersheds with major water supply and flood control facilities such as the Salinas River. Future conflict is expected due to existing and anticipated future development, habitat degradation, and conflict with land development and associated flood control activities and water supplies.

Study Area

Estimates of steelhead abundance within the Salinas River watershed are limited, with no data available for recent years. However, impacts on critical habitat in the watershed have been accompanied by a progressive decline in steelhead abundance, notably in the tributaries (i.e., San Antonio and Nacimiento Rivers) and mainstem of the upper Salinas River. Specific estimates of the steelhead decline have not been well documented, but several infrequent estimates exist. The U.S. Fish and Wildlife Service (USFWS) estimated an average run size of 900 fish in 1951. In 1983, the population was estimated to be fewer than 500 adults (National Marine Fisheries Service 2007) with numerous observations of steelhead distributed throughout the headwaters of the upper basin, the major tributaries draining the western side of the Salinas Valley, and in Gabilan Creek (Titus et al. 2002).

More recent data indicate that the steelhead population in the Salinas River watershed is consistently declining due to low survivorship across multiple life stages. These conditions were likely exacerbated by the recent drought that occurred from 2012 to 2016. The population may be currently supported by both resident fish and those straying from other watersheds (National Marine Fisheries Service 2007).

Adult escapement monitoring in the lower Salinas River has revealed a modest population of steelhead returning each year. Since 2011, between 0 and 43 fish have returned each year, although sampling did not cover the entire migration window (FISHBIO in prep). Migration timing of steelhead was highly variable from year to year, occurring as early as the first half of December and as late as the end of March. Typically, adult migration coincided with or occurred after periods of increased flow, and only in years the lagoon was connected to the ocean.

In the biological opinion for the Salinas Valley Water Project (SVWP; National Marine Fisheries Service 2007), NMFS concluded that the Salinas River run of steelhead had likely declined to approximately 50 adult fish per year (EDAW 2001; National Marine Fisheries Service 2007). They concluded that the population was at risk due to the low abundance of each subpopulation and the fact that small populations have a greater risk of extinction due to genetic bottlenecking and environmental stochasticity (e.g., drought, disease, wildfire; Gilpin and Soule 1986; Pimm et al. 1988; McElhany et al. 2000).

F1.1.6 Threats

The populations of steelhead in the study area face numerous threats and stressors, most of which come from anthropogenic sources. Stressors, as defined by NMFS (2007), are physical, chemical, or biological conditions that limit the production of steelhead within the range of the species. Common threats to steelhead statewide include water degradation, lack of cold water, and low and variable stream flows due to logging, road construction, land use practices, and urbanization, as well as constricted habitat, reduced habitat suitability, and food web alteration (Moyle 1995). In the Salinas River watershed, lack of flows, barriers to migration, high water temperatures, and degraded habitat are among the biggest threats facing the species (National Marine Fisheries Service 2013).

Several threats to Salinas River watershed steelhead have been identified by NMFS (2013) based upon the leading stressors affecting properly functioning conditions of critical habitat in the watershed (Table F1-4). The sources of these stressors have also been identified (Table F1-5), based on methodology detailed in NMFS (71 FR 833–862). Flow-related passage issues appear to be among the leading stressors in the watershed, as evidenced by impaired migration between the ocean and estuary and upstream spawning and rearing habitats. A variety of sources are responsible for this impairment including groundwater pumping, surface water diversions, and dams associated with agricultural and urban developments. Although reaches of the Salinas River historically went dry during portions of the year, water use in the watershed has severely exacerbated these issues, leading to impairment of upstream migration of adult steelhead and downstream migration of juveniles during the majority of the year. In addition, changes in channel configuration from channelization and gravel mining, loss of riparian habitat, and agricultural encroachment into the floodplain has also affected surface flows.

Table F1-4. Sources of Threats to the Salinas River Watershed Subpopulations of the South-Central California Coast Steelhead Distinct Population Segment

	Top Stressors			
Subpopulation	**1**	**2**	**3**	**4**
Arroyo Seco	Flow-related passage	Barriers	Summer Base flow	None
San Antonio/Nacimiento	Barriers	Competition	None	None
Upper Salinas	Summer base flow, flow-related passage	Summer base flow, flow-related passage	Water temperature	Barriers
Lower Salinas	Flow-related passage	Degraded estuarine habitat	Toxic contamination	Channelization

Source: Adapted from National Marine Fisheries Service 2013.

Table F1-5. Sources of Threats to the Salinas River Watershed Subpopulations of the South-Central California Coast Steelhead Distinct Population Segment

	Sources			
Subpopulation	**1**	**2**	**3**	**4**
Arroyo Seco	Salinas River flows	Gravel mining, water diversions, and road crossings	Groundwater and surface diversions	None
San Antonio/ Nacimiento	Large dams	Introduced trout	None	None
Upper Salinas	Groundwater and surface diversions	Large dams	Groundwater and surface diversions and grazing	Dams, roads
Lower Salinas	Dams, groundwater and surface diversions	Dams, diversion, and flood control	Agriculture and urbanization	Agriculture and urbanization

Source: Adapted from National Marine Fisheries Service 2013.

Subpopulations of steelhead occupying the Salinas watershed show a strong pattern of flow-related passage issues and reduced summer base-flows as primary stressors to the populations. This suggests that all life stages of steelhead are impaired by these stressors in the watershed. Reduced base flows and flow-related passage impair the quality of freshwater rearing habitat by reducing the amount of available rearing space, exacerbating high temperatures, and otherwise reducing the survival of steelhead fry, parr, and pre-smolts. Sources of these threats are the same as those affecting migration by lowering of groundwater levels (i.e., groundwater pumping, surface water diversions, and dams associated with agricultural and urban developments).

Threats common to all subpopulations of steelhead in the SCCCS DPS are discussed below.

F1.1.6.1 Anthropogenic Influences

One of the major causes of the decline of steelhead in the Salinas River watershed is the decrease in quality and function of critical habitat. Habitat destruction and fragmentation have been linked to increased rates of species extinction (Davies et al. 2001), and in the SCCCS DPS, steelhead have declined as a result of habitat degradation resulting from water diversions, large dams, agricultural practices, urbanization, loss of wetland and riparian zones, roads, grazing, gravel mining, and logging. Water storage, withdrawal, conveyance, and diversions for agriculture, flood control, domestic, and hydropower purposes have greatly reduced or eliminated historically accessible steelhead habitat. Modification of natural flow regimes by dams and other water-control structures have resulted in increased water temperatures, changes in fish community structures, depleted flows during migration, spawning, rearing, flushing of sediments from spawning gravels, and reduced gravel recruitment. While different factors have had varying levels of influence, the general trend has been one of increasing pressure on aquatic resources, particularly in the lower reaches of the watershed. In addition, this degradation of critical habitat has exacerbated the adverse effects of natural environmental variability such as drought, poor ocean conditions, and predation.

Land-use activities associated with urban development, mining, agriculture, ranching, and recreation have significantly altered steelhead habitat quantity and quality. Associated impacts of these activities include alteration of stream bank and channel morphology; alteration of ambient stream water temperatures; degradation of water quality; elimination of spawning and rearing habitats; fragmentation of available habitats; elimination of downstream recruitment of spawning gravels and large woody debris; removal of riparian vegetation resulting in increased stream bank erosion; and increased fine sedimentation input into spawning and rearing areas. The net effect of these activities is the loss of channel complexity, pool habitat, suitable gravel substrate, and large woody debris, all of which are critical for steelhead production.

A significant percentage of estuarine habitats have been lost, particularly in the northern and southern portions of the DPS, where the majority of the wetland habitat historically occurred. The condition of remaining wetland habitats is in many cases highly degraded, with many wetland areas at continued risk of loss or further degradation (National Marine Fisheries Service 2013). Although numerous historically harmful practices have been halted, much of the historical damage remains to be addressed, and any restoration activities will require a significant amount of time to complete.

Water Use

Natural hydrological cycles in the Salinas River watershed have been altered by depletion and storage of natural flows, particularly within larger streams in the upper Salinas watershed that provide habitat to the SCCCS DPS. Dams, surface water diversions, and groundwater extraction are common across the Salinas River watershed. Loss of surface flows through the operation of dams or surface water diversions has increased juvenile steelhead mortality due to impaired migration from insufficient flows or habitat blockages, loss of rearing habitat due to dewatering and blockage, stranding of fish resulting from rapid flow fluctuations, entrainment of juveniles into unscreened diversions, and increased water temperatures (Bergren and Filardo 1993). Dams also negatively affect the hydrology, sediment transport processes, and geomorphology of the affected drainages. In addition, dams and reservoirs often provide opportunities for recreational fishing and can lead to the introduction nonnative predators and/or competitors (e.g., largemouth and smallmouth bass, carp, crayfish, western mosquitofish).

Re-establishing surface flows and/or maintaining hydrologic connections and physical access between the ocean and upper watersheds would expand access to historically important spawning and rearing habitats, which is essential to recovery of the Salinas River watershed subpopulation. Increased surface flows would improve the overall habitat conditions (amount and complexity) for steelhead, as well as the existing populations of native residualized steelhead that currently are isolated above dams and reservoirs.

Land Use Practices

Human population density is high in the Salinas River watershed, with substantial agricultural development occurring along the mainstem of the Salinas River and tributaries, which can magnify potential impacts on steelhead even though most of the watershed remains undeveloped. Agricultural development on lower floodplains has resulted in channelization, removal of riparian vegetation, and simplification of channel structure and function, as well as the elevation of fine sediments, pesticides and fertilizers, which can lead to elevate nutrient levels and increase biological oxygen demand. Public ownership of lands in the study area (U.S. National Forest and Bureau of Land Management (BLM) lands, military bases, etc.) can be extensive, although these public lands are generally concentrated in the upper watersheds (Hunt & Associates 2008).

Flood Control, Levees, and Channelization

Extensive channelization has occurred along the lower Salinas River, which has been realigned, resulting in loss or degradation of the riparian corridor and streambed (National Marine Fisheries Service 2008; Hunt & Associates 2008). Flood-control practices and subsequent channelization of streams and development of levees can impair the function and quality of stream habitats (National Marine Fisheries Service 1996; Brown et al. 2005; Jeffres et al. 2008). Habitat impairments in the Salinas River watershed may have resulted in several detrimental habitat features for steelhead including increased water temperature, incision of the streambed and loss of structural complexity and instream refugia (meanders, pools, undercut banks, etc.), loss of bed and bank habitat, increased sedimentation, turbidity, and substrate embeddedness, and excessive nutrient loading (Newcombe and Jensen 1996; Newcombe 2003; Naiman et al. 2005; Jeffres et al. 2008; Richardson et al. 2010).

Estuarine Loss

The Salinas River Lagoon is used by steelhead as rearing areas for juvenile steelhead as well as a staging area for smolts acclimating to saline conditions in preparation for entering the ocean and adults acclimating to freshwater in preparation for upstream migration and spawning (National Marine Fisheries Service 2008). Located at the downstream end of the watershed, it has been subjected to numerous threats, which have adversely affected the estuarine function in a variety of ways (e.g., degradation of water quality, modification of hydrologic patterns, changes in species composition). Approximately 10% of the historical estuary habitat remains in the Salinas River due to land use conversion resulting in filling, diking, and draining the lagoon. In addition, the habitat complexity and ecological functions of the estuary have been substantially reduced as a result of the loss of shallow-water habitats such as tidal channels, degradation of water quality through both point and non-point waste discharges, and artificial breaching of the seasonal sandbar at the mouth of the river, which can reduce and degrade steelhead rearing habitat by reducing water depths and the surface area of estuarine habitat.

Fishing Harvest

Despite a dearth of good historical accounts of the amount of steelhead harvested along the California coast (Jensen and Swartzell 1967), Shapovalov and Taft (1954) found that very few steelhead were caught by commercial salmon trollers at sea but considerable numbers were taken by sports anglers in Monterey Bay. Anecdotal reports of recreational fishing and poaching of adult steelhead further up in the watershed (Franklin 1999) suggests a relatively high level of historical fishing pressure.

Currently, despite the listing of the SCCCS DPS as threatened under the ESA, recreational angling for steelhead continues to be permitted in nearly all coastal drainages in south-central California including areas above currently impassible barriers. NMFS has previously concluded that recreational harvest is a limiting factor for SCCCS (Busby et al. 1996; Good et al. 2005), and angling in anadromous portions of coastal rivers and streams has been somewhat restricted through modification of the California Department of Fish and Wildlife's (CDFW's) angling regulations.

Artificial Propagation

Releasing large numbers of hatchery fish can pose a threat to steelhead populations through genetic impacts, competition for food and other resources, predation of hatchery fish on wild fish, and increased fishing pressure on wild stocks as a result of hatchery production (Waples 1991). The genetic impacts of artificial propagation programs are primarily caused though hybridization of hatchery and wild fish, which can reduce the genetic integrity and diversity that protect against changes in the environment. Steelhead in the Salinas River may be of "mixed genetic origin" due to the stocking of steelhead by the Monterey Bay Salmon and Steelhead Trout Project (Becker et al. 2010). Hatchery steelhead have also been introduced into the Salinas River watershed above the Nacimiento and San Antonio dams for recreational angling (National Marine Fisheries Service 2013). Stocking of nonnative steelhead in anadromous reaches of the Nacimiento River has also occurred until recently. Currently, CDFW limits stocking to non-anadromous waters using triploid rainbow trout (California Department of Fish and Wildlife and U.S. Fish and Wildlife Service 2010). In spite of previous stocking practices, genetic testing of SCCCS has not detected substantial interbreeding between naturally spawned steelhead and hatchery reared steelhead (Girman and Garza 2006; Clemento et al. 2009; Abadia-Cardoso et al. 2011; Christie et al. 2011).

F1.1.6.2 Environmental Influences

Climate Change

Global warming has been scientifically validated as an anthropogenically driven phenomenon by the United Nations Framework Convention on Climate Change, the Intergovernmental Panel on Climate

Change, and others (United Nations Framework Convention on Climate Change 2006), and is expected to result in the warming of the atmosphere from increased greenhouse gas emissions. These changes will affect physical, chemical, ecological, and biological processes throughout the oceans, the biosphere, and the world's water cycle. Changes in the distribution and abundance of a wide array of biota suggest that the current warming trend has great potential to affect species' distribution and survival (Davies et al. 2001; Schneider and Root 2002), with the population extinction rate increasing in proportion to the magnitude of climate fluctuations (Good et al. 2005). In California, it is expected that there will be a predicted increase in critically dry years (Cayan et al. 2006), which will lead to a lack of surface flow in streams. Future climate change may therefore substantially increase risk to the species by exacerbating dry conditions. More information on climate change effects on steelhead is available in Section 3.5.9.

Ocean Conditions

Variability in marine environmental factors has been shown to substantially affect North Pacific salmon production (Beamish and Bouillion 1993; Beamish et al. 1997). For example, El Niño conditions, which occur every 3–5 years, negatively affect ocean productivity (Beamish et al. 1997). Prolonged periods of poor marine survival can affect the viability of populations, as was evidenced by the salmon fishery collapse during 2008–2009 (Lindley et al. 2009). Steelhead populations have persisted through these poor ocean periods, although these historically occurred under better habitat conditions. It is less certain how the SCCCS DPS will fare in periods of poor ocean survival when their freshwater, estuary, and nearshore marine habitats are degraded (Good et al. 2005).

Disease

Infectious disease is one of many factors that can influence adult and juvenile steelhead survival. Specific diseases such as *Ceratomyxosis*, *Columnaris*, *Furunculosis*, bacterial kidney disease, infectious hematopoietic necrosis virus, redmouth and black spot disease, erythrocytic inclusion body syndrome, and whirling disease, among others, are present in the SCCCS DPS and are known to affect steelhead. Very little current or historical information exists for steelhead to quantify changes in infection levels and mortality rates over time. In many cases, warm water temperatures, which are expected to occur more frequently in the Salinas River watershed, can contribute to the spread of infectious disease. However, studies have shown that native fish tend to be less susceptible to pathogens than hatchery cultured and reared fish (Buchanan et al. 1983).

Predation

Introductions of nonnative aquatic species (including fishes and amphibians) and habitat modifications (e.g., dams and impoundments, altered flow regimes, etc.) have resulted in increased predator populations in numerous river systems, thereby increasing the level of predation experienced by native salmonids (Busby et al. 1996). Nonnative species, particularly fishes and amphibians such as largemouth and smallmouth bass (*Micropterus* spp.) and bullfrogs (*Lithobates catesbeianus*) have been introduced and spread widely. These species can prey upon rearing juvenile steelhead (and their conspecific resident forms), compete for living space, cover, and food, and act as vectors for nonnative diseases (Cucherousset and Olden 2011).

Artificially induced summer low-flow conditions may also benefit nonnative species, exacerbate spread of diseases, and permit increased predation. NMFS (2013) concluded that the information available on these impacts on steelhead did not suggest that the DPS was in danger of extinction, or likely to become so in the foreseeable future, because of predation. However, small populations such as SCCCS can be more vulnerable to extinction through the synergistic effects of other threats, and the role of predation may be heightened under conditions of periodic low flows or high temperatures characteristic of the SCCCS DPS habitats.

Predation by marine mammals is not believed to be a major factor contributing to the decline of steelhead on the West Coast relative to other factors. However, both harbor seals (*Phoca vitulina*) and California sea lions (*Zalophus californianus*) are present within Monterey Bay, and populations of both cetaceans have increased along the Pacific Coast (National Marine Fisheries Service 1999). Previous studies (Hanson 1993) have shown that the foraging behavior of California sea lions and harbor seals with respect to anadromous salmonids was minimal, and that predation on salmonids appeared to be coincidental with the salmonid migrations rather than dependent upon them. Nevertheless, this type of predation is worth noting as it may have substantial impacts in localized areas (e.g., below a dam), although there is no evidence of this in the Salinas watershed.

Collectively, all of the factors listed above have severely degraded steelhead migration, spawning, and rearing habitat in the Salinas River watershed and are largely responsible for the decline of steelhead in the watershed. Steelhead migration habitat has been degraded by dams and their operations, which preclude access to spawning and rearing habitats and limit stream flows. Flood-control efforts have scoured the mainstem and reduced resting and hiding cover, while also contributing to reduced migration opportunities at low flows. Spawning and rearing habitat has been degraded by dam operations that reduce the amount of habitat space available and/or may disrupt redds. Lagoon management has created conditions in which few steelhead can successfully rear in the Salinas River Lagoon. Agriculture contributes pollutants (including nutrients and toxic contaminants), reduces dissolved oxygen levels, and increases temperatures to the lower river and Salinas River Lagoon. Fish planting may have degraded the genetic viability of wild steelhead. Many of these conditions are expected to continue, and possibly get worse, in the future.

F1.1.7 Recovery Planning

F1.1.7.1 State

The ESA mandates that the NMFS develop and implement recovery plans for the conservation (recovery) of listed species. Recovery plans are available for all DPS's of steelhead in California (except Northern California Steelhead) including the California Central Valley DPS (National Marine Fisheries Service 2014), Central California Coast DPS (National Marine Fisheries Service 2016), SCCCS DPS (National Marine Fisheries Service 2013), and Southern California DPS (National Marine Fisheries Service 2012).

NMFS issued a recovery plan for SCCCS in 2013 with the goal of preventing extinction in the wild and ensuring the long-term persistence of viable, self-sustaining populations of steelhead distributed across the DPS. The SCCCS recovery planning area includes those portions of coastal watersheds that are seasonally accessible to anadromous steelhead entering from the ocean, as well as the upper portions of watersheds above anthropogenic fish passage barriers that have historically contributed to the maintenance of anadromous populations (National Marine Fisheries Service 2013). Based in large part on Boughton et al. (2006), NMFS has divided 39 watersheds in which SCCCS have occurred historically into 4 biogeographic population groups (BPGs): Interior Coast Range, Carmel Basin, Big Sur Coast, and San Luis Obispo Terrace. The Interior Coast Range

BPG includes 5 populations of steelhead based on genetic and distributional information that largely correspond to one population per watershed (Boughton et al. 2006).

F1.1.7.2 Study Area

The development and implementation of the recovery plan for the SCCCS DPS is considered vital to the continued persistence and recovery of anadromous steelhead in the Salinas River. The Salinas River recovery planning area includes those portions of the watershed that are seasonally accessible to anadromous steelhead entering from the ocean, as well as the upper portions of watershed above anthropogenic fish passage barriers that historically contributed to the maintenance of anadromous populations. Implementation of the recovery plan will require the continued development of site-specific and project-specific information and involvement of interested stakeholders to ensure that recovery actions are effective and sustainable.

Recovery plans developed under the ESA are guidance documents, not mandatory regulatory documents. However, the ESA envisions recovery plans as the central organizing tool for guiding the recovery of listed species. The SCCCS recovery plan serves as a guideline for achieving recovery goals by describing the criteria by which NMFS would measure species recovery, the strategy to achieve recovery, and the recommended recovery actions necessary to achieve viable populations of steelhead within the SCCCS recovery planning area. Recovery does not necessarily require restoring watersheds to a pre-development, pristine state, but restoring riverine functions to the point that they support viable populations of wild steelhead.

The LTMP includes the Interior Coast Range BPG. The Salinas River watershed's larger size has allowed sufficient geographic isolation among populations of SCCCS to maintain multiple populations (Boughton et al. 2006). Within the Salinas River watershed, steelhead form three distinct populations in Gabilan Creek, Arroyo Seco, and upper Salinas River, which includes the Nacimiento and San Antonio Rivers (Boughton et al. 2006). For recovery planning, all populations within the Salinas River watershed have been designated as Core 1 populations. The Core 1 classification signifies highest priority populations for recovery based on the (1) intrinsic potential of the population in unimpaired conditions, (2) role of the population in meeting spatial and/or redundancy viability criteria, (3) severity of the threats facing the population, and (4) capacity of the watershed and population to respond to critical recovery actions. Such a strategy aims to restore the natural selective regime under which steelhead evolved, which is key to the species' long-term survival. The proposed strategy looks for opportunities for sustainable water and land-use practices, restores river and estuary processes that naturally sustain steelhead habitats, provides diverse opportunities for steelhead within the natural range of ecological adaptability, sustains ecosystem services for humans by reinforcing natural capital and the self-maintenance of watersheds and river systems, and builds natural and societal adaptive capacity to deal with climate change.

Many complex and inter-related biological, economic, social, and technological issues must be addressed in order to recover anadromous steelhead in the Salinas River. Policy changes at the federal, state, and local levels will be necessary to implement many of the recovery actions identified in this recovery plan. For example, without substantial strides in water conservation, efficiency, and re-use throughout south-central California, flow conditions for anadromous salmonids will limit recovery. Similarly, recovery is unlikely without programs to restore properly functioning historic habitats, such as estuaries, and access to upstream spawning and rearing habitat, particularly above dams.

Extensive, high-quality habitat exists above a large number of passage barriers in the SCCCS river systems. These areas are currently not included within the SCCCS DPS as defined in the listing rule (71 FR 834). However, because these habitat areas constitute a majority of the prime steelhead spawning and rearing habitat within the species' historic range, they are identified as recovery actions. In addition, restoring flows, access to spawning and rearing habitats, and instream habitat conditions (including estuarine conditions) necessary to support steelhead are also principal recovery actions to restore the Salinas River subpopulation and will require continuing active management in a region with a large human population and extensively developed land uses.

Many of the recovery actions identified in the recovery plan address watershed-wide processes that are also the focus of other local, state and federal programs (e.g., wildfire regime, erosion and sedimentation, runoff and waste discharges), which will benefit a wide variety of native species (including federally listed species or species of special concern) by restoring natural ecosystem functions. Some of the listed species which co-occupy coastal watersheds with SCCCS include tidewater goby (*Eucyclogobius newberryi*), foothill yellow-legged frog (*Rana boylii*), California least tern (*Sternula antillarum browni*), California red-legged frog (*Rana draytonii*), western pond turtle (*Actinemys marmorata*), Arroyo toad (*Bufo californicus*), least Bell's vireo (*Vireo bellii pusillus*), and western snowy plover (*Charadrius nivosus;* U.S. Fish and Wildlife Service 2015). Additionally, Pacific lamprey (*Entosphenus tridentata*)—another anadromous fish species occupying south-central California watersheds and whose numbers have declined significantly—can also be expected to benefit from many of the recovery actions identified in the recovery plan.

In addition to benefiting natural communities in the Salinas River watershed, restoration of steelhead habitats will also provide substantial benefits for human communities. These include, but are not limited to, improving and protecting the water quality of important surface and groundwater supplies, reducing damage from periodic flooding resulting from floodplain development, and controlling invasive exotic animal and plant species that can threaten water supplies and increase flooding risks. Restoring and maintaining ecologically functional watersheds also enhances important human uses of aquatic habitats occupied by steelhead; these include activities such as outdoor recreation, environmental education, field-based research of both physical and biological processes of coastal watersheds, aesthetic benefits, and the preservation of tribal and cultural heritage values.

Although the recovery of SCCCS is expected to be a long process, the NMFS Technical Recovery Team (TRT) recommends certain actions that should be implemented as soon as possible to help facilitate the recovery process for the Salinas River steelhead (Table F1-6). These include identifying a set of core populations on which to focus recovery efforts, protecting extant parts of inland populations, identifying refugia habitats, protecting and restoring estuaries, and collecting population data (Boughton et al. 2007).

Table F1-6. Critical Recovery Actions for Core 1 Populations in the Interior Coast Range Biological Population Group of the South-Central California Coast Steelhead Distinct Population Segment

Population	Critical Recovery Actions
Salinas River	Develop and implement operating criteria to ensure the pattern and magnitude of groundwater extractions and water releases from Salinas Dam to provide the essential habitat functions to support the life history and habitat requirements of adult and juvenile steelhead. Physically modify all fish passage impediments, including the Salinas Dam, to allow steelhead natural rates of migration to upstream spawning and rearing habitats, and passage of smolts and kelts

Population	Critical Recovery Actions
	downstream to the estuary and ocean. Manage instream mining to minimize impacts to mitigation, spawning, and rearing habitat, and protect spawning and rearing habitat in major tributaries, including the Arroyo Seco. Identify, protect, and where necessary, restore estuarine rearing habitats, including management of artificial breaching of the sandbar at the river's mouth.
Arroyo Seco River	Develop and Implement operating criteria to ensure the pattern and magnitude of groundwater extractions from the Arroyo Seco and lower Salinas River provide the essential habitat functions to support the life history and habitat requirements of adult and juvenile steelhead. Physically modify fish passage impediments, including concrete road crossing and diversion structure to allow steelhead natural rates of migration to upstream spawning and rearing habitat, and passage of smolts and kelts downstream to the estuary and ocean.
San Antonio River	Develop and implement operating criteria to ensure the pattern and magnitude of groundwater extractions and water releases, including bypass flows around diversions and dams (e.g. San Antonio Dam), to provide the essential habitat functions to support the life history and habitat requirements of adult and juvenile steelhead. Physically modify San Antonio Dam to allow steelhead natural rates of migration to upstream spawning and rearing habitats, and passage of smolts and kelts downstream to the estuary and ocean.
Nacimiento River	Develop and implement operating criteria to ensure the pattern and magnitude of water extractions and water release, including bypass flows around diversions and dams (e.g. Nacimiento Dam) to provide the essential habitat functions to support the life history and habitat requirements of adult and juvenile steelhead. Physically modify Nacimiento Dam to allow steelhead natural rates of migration to upstream spawning and rearing habitats, and passage of smolts and kelts downstream to the estuary and ocean.

Source: National Marine Fisheries Service 2013

F1.1.8 Existing Conservation Actions in the Study Area

In April 2010, the Monterey County Water Resources Agency (MCWRA) began operation of the Salinas River Diversion Facility (SRDF) as part of the SVWP. Operation of the SRDF involves release of water from Nacimiento Reservoir or San Antonio Reservoir to the Salinas River throughout the irrigation season with impoundment and diversion at the SRDF located at about river mile 4.8 near the upper part of the Salinas River Lagoon. The purpose of the project is to provide surface water that can be used to diminish groundwater extraction and reduce the amount of salt water intrusion into the groundwater basin. Details on project operations can be found in Section 2.3.1.1, *Reservoirs*, under *Water Releases*.

Flows requirements for the SRDF influence water quality conditions in the lagoon during the dry season and likely improve water quality overall. Previous to implementation of the SVWP there was no requirement for provision of flow to the lagoon and there was generally no flow to the lagoon following the last storm events in the spring. Although this was likely consistent with natural river flow patterns before development of the Salinas Valley, dry season flows to the lagoon likely improve water quality conditions and help to maintain a hospitable rearing environment for steelhead in the lagoon.

Restoration of the steelhead resources of the Salinas River system depends largely on protecting habitat (i.e., managing land and water resource use) in the Arroyo Seco watershed, as this comprises the majority of high quality steelhead rearing and spawning habitat remaining in the basin. Effective restoration is also contingent on providing migration flows between the Arroyo Seco confluence with the Salinas River and the river mouth, and ensuring passage at barriers both within the Arroyo Seco watershed and in the lower Salinas River. The Monterey County Public Works Department removed a major fish passage barrier at the Thorne Road Crossing on the Arroyo Seco River in 2008, and The Nature Conservancy purchased a conservation easement on Los Vaqueros Ranch in the Arroyo Seco watershed in 2010, protecting 1,337 acres of land along 2 miles of the Arroyo Seco mainstem and a portion of Vaqueros Creek, a tributary to the Arroyo Seco.

Operation of the SVWP likely enhances fish passage opportunities into the Arroyo Seco due to flow requirements that improve connectivity during important migratory periods. Increased flow releases from the Nacimiento and San Antonio reservoirs improve upstream and downstream migration opportunities for steelhead. As part of the requirements of the biological opinion for the SVWP (National Marine Fisheries Service 2007), MCWRA conducts monitoring of steelhead smolt outmigration from March 15 to May 31 in the Arroyo Seco, Salinas, and Nacimiento Rivers. The SVWP Fish Habitat and Monitoring Program, is intended to (1) quantify the presence of the threatened steelhead trout in the lower Salinas River system (population monitoring), (2) monitor river flows to ensure adequate water for fish passage (migration monitoring), and (3) monitor water quality to determine habitat suitability (National Marine Fisheries Service 2003). Results of this monitoring have helped to elucidate key data gaps in the Salinas River. Reports on this monitoring can be found on MCWRA's website at: http://www.co.monterey.ca.us/government/government-links/water-resources-agency/programs/fish-monitoring#wra.

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Attachment F1
Figures

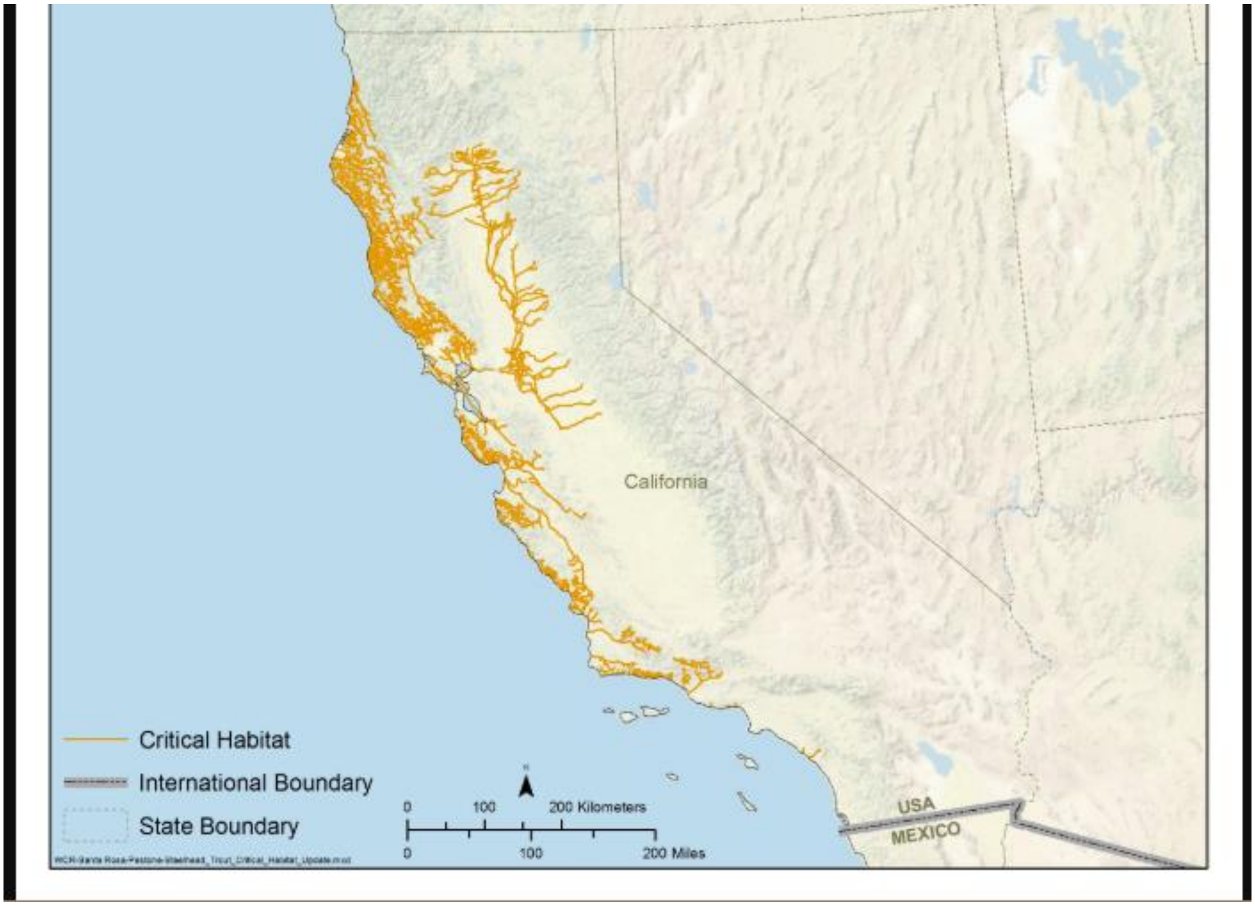

Figure F1-1. State-Wide Critical Habitat for Steelhead

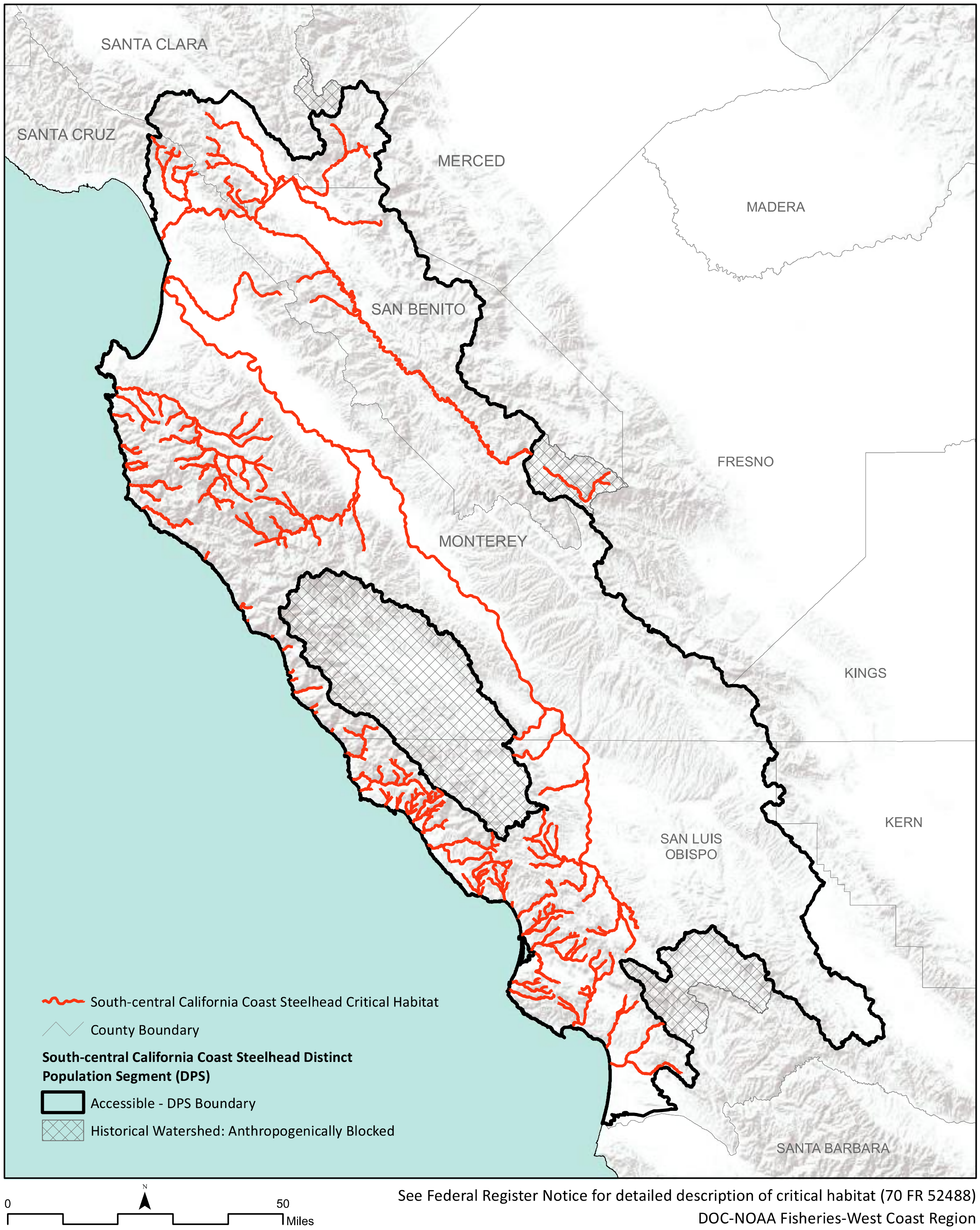

Figure F1-2. Critical Habitat for South-Central California Coast Steelhead

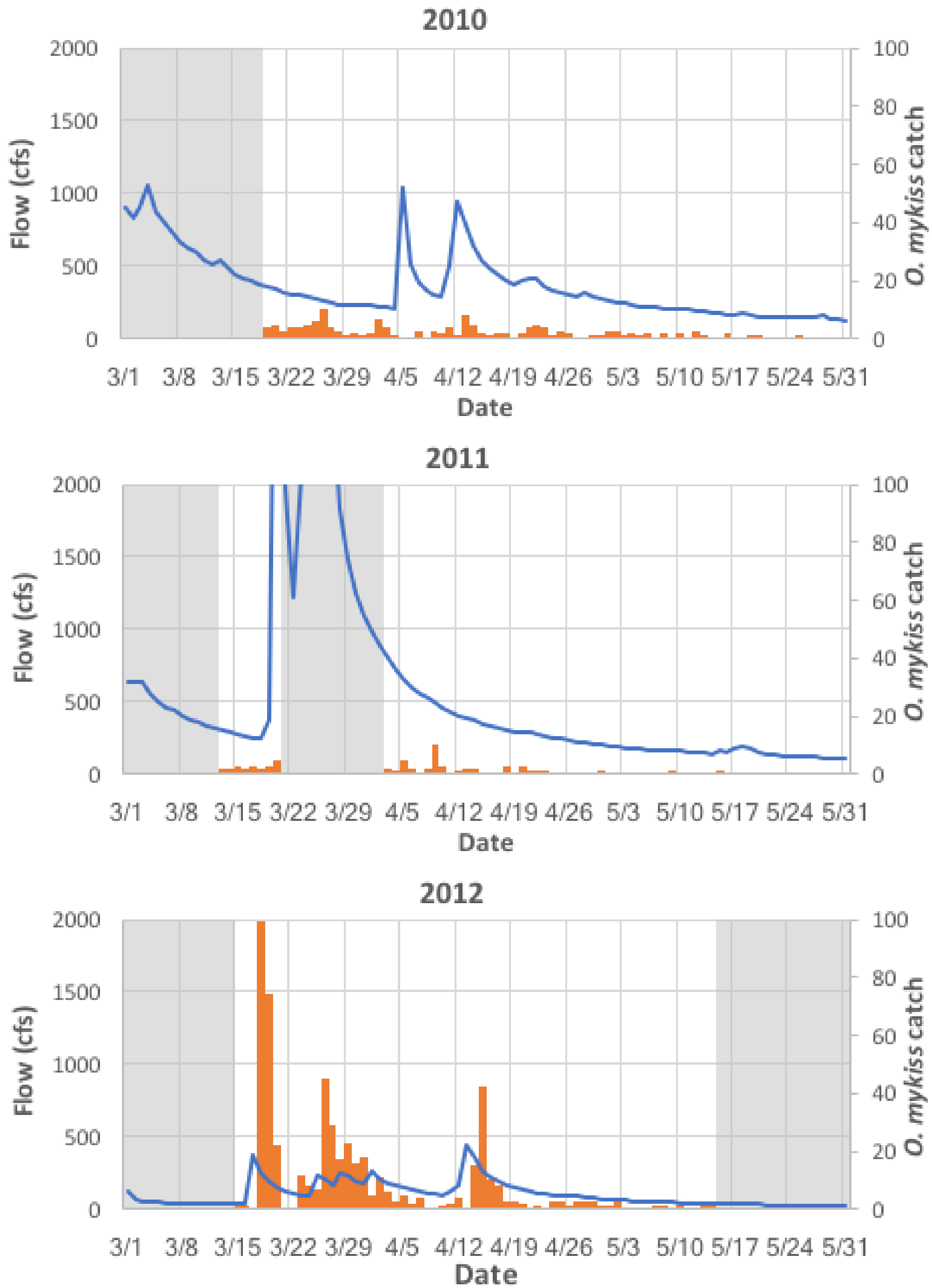

Figure F1-3. Rotary Screw Trap Catch of Steelhead (Orange Bars) Relative to Discharge (Blue Line) in Arroyo Seco in 2010, 2011, 2012. *Grey background indicates when the trap was not deployed.*

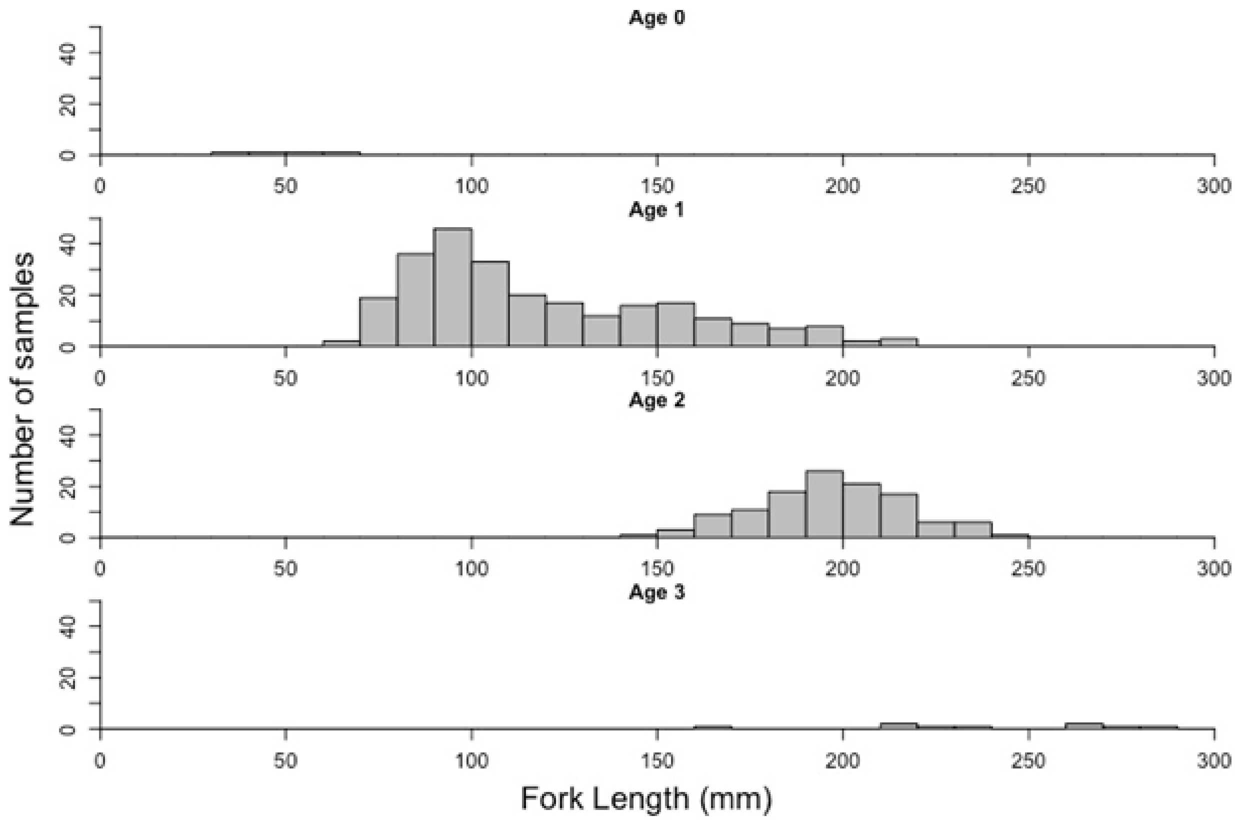

Figure F1-4. Length distribution of *O. mykiss* at different ages sampled at the Arroyo Seco River Rotary Screw Trap.

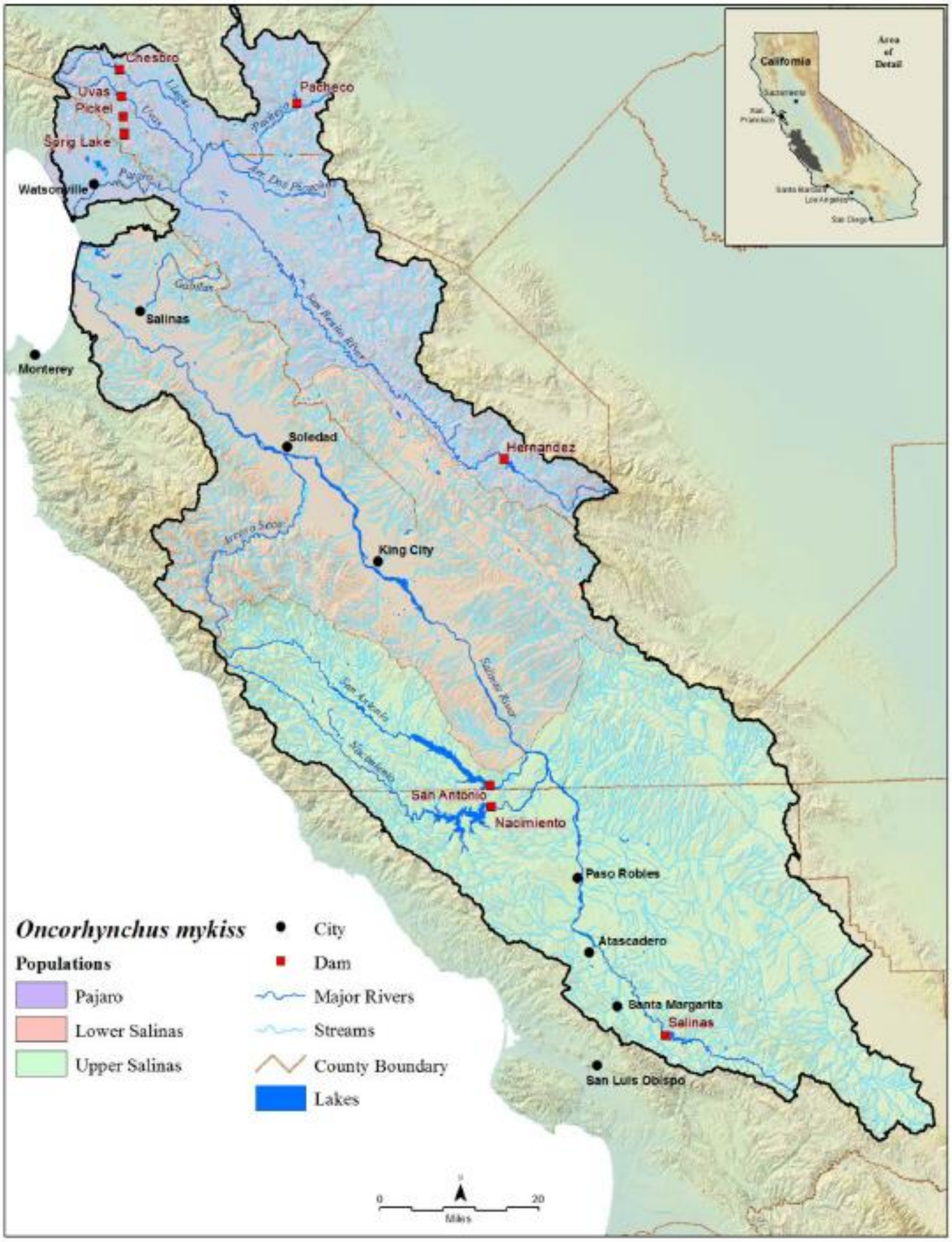

Figure F1-5. Steelhead Habitat in the Salinas and Pajaro River Watersheds

Appendix F2
Tidewater Goby Species Account

F2.1 Tidewater Goby (*Eucyclogobius newberryi)*

F2.1.1 Legal Status

F2.1.1.1 State

Tidewater goby is identified as a Species of Special Concern.

F2.1.1.2 Federal

Tidewater goby is listed as endangered under the ESA (59 FR 5494-5498), although it has since been proposed to reclassify tidewater goby as threatened (79 FR 14340–14362). Reasons for downlisting include (1) the number of localities known to be occupied has nearly tripled since listing (from 43 to 114), (2) the increase in occupied localities indicates that the tidewater goby is more resilient in the face of severe drought events than believed at the time of listing, and (3) threats identified at the time of listing have been reduced or are not as serious as previously thought.

F2.1.1.3 Critical Habitat

Critical habitat for tidewater goby was re-designated in 2013 to cover approximately 12,156 acres (4,920 hectares) of estuaries and lands in portions of Del Norte, Humboldt, Mendocino, Sonoma, Marin, San Mateo, Santa Cruz, Monterey, San Luis Obispo, Santa Barbara, Ventura, Los Angeles, Orange, and San Diego Counties, California (also see 78 FR 8745). This re-designation increased the amount of critical habitat for tidewater goby, which was previously based on a January 2008 ruling that designated 10,003 acres of critical habitat throughout the state of California.

The critical habitat designation in the study area for tidewater goby includes Bennett Slough (north of the study area) and the Salinas River (78 FR 8759; Figures F2-1a–1c).

F2.1.2 Taxonomy

Tidewater goby was first described as a new species by Girard (1856) as *Gobius newberryi*. Gill (1863) erected the genus *Eucyclogobius* for this distinctive species. The intraspecific phylogeny of tidewater goby is highly geographically structured. Crabtree's (1985) genetic work on tidewater goby shows fixed allelic differences at the extreme northern and southern ends of the range and some variation in central California. Each of these northern and southern populations is distinct from each other and from those central populations that have been sampled. The other more centrally distributed populations are relatively similar to one another. This study was based on 12 localities distributed over most of the range. The precise limits of allozyme differentiation are not known. The results of this study indicate that there is a very low level of gene flow between the populations sampled. Many of the populations may be diverging genetically from each other due to discrete, seasonally closed estuaries, where tidewater gobies have low dispersal ability (Crabtree 1985).

Dawson et al. (2001) analyzed mitochondrial DNA and cytochrome b sequences of individual tidewater gobies collected from 31 locations between 1990 and 1999. Their study revealed six major phylogeographic groups in four clusters—the San Diego clade south of Los Angeles and Point Buchon, a lone Estero Bay group from central California, and San Francisco and Cape Mendocino groups from northern California—that genetically vary. Barriers to gene flow likely exist in the vicinities of Los Angeles, Seacliff, Point Buchon, Big Sur, and Point Arena. Finer scale phylogeographic structure within these regions is suggested by genetic differences between estuaries but is poorly resolved by current analysis (Dawson et al. 2001). Dawson et al. (2001) found that phylogenetic relationships between and patterns of molecular diversity within the six groups are consistent with repeated and sometimes rapid northward and southward range expansions out of central California, likely caused by Quaternary climate change. The modern geographic and genetic structure of tidewater goby has probably also been influenced by patterns of expansion and contraction, colonization, extirpation, and gene flow linked to Pliocene-Pleistocene tectonism, Quaternary coastal geography and hydrography, and historical human activities (Dawson et al. 2001). The deepest phylogenetic gap in *Eucyclogobius* coincides with phylogeographic breaks in several other coastal California taxa in the vicinity of Los Angeles, suggesting common extrinsic factors have had similar effects on different species in this region. In contrast, evidence of gene flow exists across the biogeographic boundary at Point Conception (Dawson et al. 2001). Furthermore, the degree of morphological variation between the phylogeographical groups was examined in 833 museum specimens from 25 localities including samples from extirpated populations. The examination of these specimens for morphological differences support the six recovery units, which are based on phylogeographic analysis (Dawson et al. 2001) and on the variation of the head lateral line canals (Ahnelt et al. 2004).

F2.1.3 Distribution

F2.1.3.1 Statewide

Historical

The tidewater goby, a fish species endemic to California, is found primarily in waters of coastal lagoons, estuaries, and marshes. Tidewater historically ranged from Tillas Slough (mouth of the Smith River, Del Norte County) to Agua Hedionda Lagoon (northern San Diego County; Figure F2-2).

Recent

Tidewater gobies are currently found throughout their known historic range but occupy fewer locations than historically, having been extirpated from some sites as a result of drainage, water quality changes, introduced predators, and drought. Tidewater goby is thought to have occurred in as many as 124 different locations during recent decades, but it currently can be found in only about 96 of those historic locations, and only about 54 of those 124 populations are thought to be secure at this time. Tidewater gobies can recolonize habitats when favorable habitat conditions are restored and individuals repopulate this restored habitat, either through natural dispersal or through human-assisted reintroduction. Tidewater gobies are naturally absent from areas where the coastline is steep and streams do not form lagoons or estuaries. Several large natural gaps occur in the species' distribution from northern Sonoma County to Del Norte County, where steep rocky shorelines dominate the coastline, and salt marsh and stream estuaries do not naturally occur (78 FR 8746–8819).

F2.1.3.2 Study Area

Historical

Although tidewater goby was historically found in the Salinas River, it was last documented in the Salinas River Lagoon in 1951, until recent observations in 2013 and 2014 (Hagar Environmental Services 2014). Tidewater goby has also been found in Bennett Slough (northern end of Elkhorn Slough; U.S. Fish and Wildlife Service 2005). No information is available on the species' historical distribution within the lagoon.

Recent

In 2013, a few individuals were found while conducting routine lagoon monitoring, with both individuals observed along the sandbar at the northwestern edge of the lagoon. In 2014, tidewater goby was the second most abundant fish species after threespine stickleback. One of the individuals was captured at the mouth of the lagoon near the usual location of breaching, four of the individuals were captured along the sandbar at the northwestern edge of the lagoon, and 53 individuals were captured near the Highway 1 Bridge (Hagar Environmental Services 2014). A doctoral student with the University of California, Los Angeles, conducted multiple surveys in the Salinas River Lagoon and Old Salinas River beginning in 2014, and was able to document and collect tidewater gobies during each visit (B. Spies, pers. comm.). However, his collection information does not detail the number or sizes of tidewater gobies that were observed during each survey, but rather provides valuable information on population persistence (Hellmair et al. 2018).

Tidewater goby distribution surveys were conducted in October 2018. Tidewater gobies were found at each sampled location along the sandbar at/near the breach site and along the southwest shoreline of the lagoon until water depth precluded sampling (upstream from the wildlife refuge parking area; Hellmair et al. 2018). This finding contrasts with survey results from most previous years, when the distribution of tidewater goby appeared restricted to the lower lagoon (with exception of the year 2014, when the species was documented as far upstream as the Highway 1 bridge). Contrary to expectations, tidewater gobies were not found in the vicinity of the OSR slidegate. During past surveys, the species was regularly found in this area, and in the OSR in the vicinity of the Monterey Dunes Way road crossing. Although this location was not sampled in October 2018 due to permit restrictions, high tidewater goby densities were also expected in this area (B. Spies, pers. comm.).

Numbers of tidewater goby captured with each seine haul during the 2018 survey ranged from 0 (near OSR slidegate, OSR and Hwy 1 Bridge) to 3. At sampling sites where the species was detected, every seine haul captured at least one goby. Due to these low capture numbers, estimation of index densities is not biologically meaningful. However, despite low captured numbers in individual seine hauls, tidewater goby appeared to be widely distributed within the lagoon, suggesting that the species was abundant during this time (Hellmair et al. 2018).

F2.1.4 Natural History

F2.1.4.1 Habitat Requirements

The tidewater goby favors the stable conditions provided by estuarine environments subject to minimal tidal fluctuation; such conditions typically occur when lagoons are cut off from the ocean by beach sandbars. All life stages of the tidewater goby are typically found in areas of low to moderate salinity (commonly less than 12 parts per thousand [ppt]). However, tidewater gobies have been documented in waters with salinity levels from 0 to 42 ppt or higher (as a comparison, sea water is about 34 ppt). They are commonly found at temperatures from 8 to 25°C (46 to 77°F); Irwin and Soltz 1984; Swift et al. 1989; Worcester 1992). Recent information suggests that gobies have a wide tolerance for salinity, oxygenation, and temperature, especially over short time periods or seasonally.

Tidewater gobies are bottom dwellers and are typically found in lagoon margin habitat at water depths of less than 3 feet, although they can occur at water depths up to 15 feet in large lagoons. They typically inhabit areas of slow-moving water, avoiding strong wave action or currents. Particularly important to the persistence of the species in lagoons is the presence of backwater, marshy habitats, which provide refuge habitat during winter flood flows. Tidewater gobies prefer a sandy substrate for breeding, but they can be found on rocky, mud, and silt substrates as well. Optimal lagoon habitats are shallow, sandy-bottomed areas, surrounded by beds of emergent vegetation. Open areas are critical for breeding, while vegetation is critical for overwintering survival (providing refuge from high flows) and probably for feeding. Tidewater goby often show a close association with widgeon grass (ruppia). Tidewater goby appears to spend all life stages in lagoons, estuaries, and river mouths, although it has been documented in slack freshwater habitats as far as 5 miles upstream from San Antonio Lagoon in Santa Barbara County (U.S. Fish and Wildlife Service 2005.

Tidewater goby habitat is subject to fluctuation of physical conditions on a seasonal basis, and estuarine processes can facilitate dispersal of subpopulations. Tidewater gobies may enter marine environments only when flushed out of lagoons, estuaries, and river mouths by normal breaching of the sandbars following storm events. However, these may be natural mechanisms of dispersal between suitable habitats on a local basis, where conditions are favorable to retain a sufficiently robust breeding population in the natal site. Gobies are unlikely to persist where daily tidal fluctuations cause substantial portions of the breeding population to be flushed from natal sites on a regular basis, or where tidal fluctuations cause breeding substrates to be dewatered.

USFWS (2005) has identified several criteria for lagoon conditions that favor tidewater gobies. These include little or no channelization, lagoon closure to the ocean for much of the year (i.e., tidal fluctuation is absent or minimal), fresh unconsolidated sand (optimal for reproduction), and high-quality inflowing water to increase the habitable area of a lagoon in summer. Nutrient-rich inflow (e.g., agricultural or urban runoff) is undesirable and can cause algal blooms, deplete oxygen, and lead to hydrogen sulfide formation. Additionally, presence of nonnative predatory fish may pose a risk for tidewater goby, as centrarchid fish (sunfish and bass) and tidewater gobies are not usually found together and may not be able to coexist.

F2.1.4.2 Movement

Gobies may move upstream during winter rains and high flows of inlet streams as well as during the summer when algal blooms and hydrogen sulfide forms in the substrate and enters the water column. During this period, most fish are found at the upper end of lagoons where freshwater inflow occurs or at the seaward end where occasional waves wash into the lagoon.

Tidewater goby reproduces predominantly in the summertime when most estuaries/coastal lagoons are closed by sand berms. This knowledge of life history, combined with genetic data, strongly suggests that tidewater goby larvae do not generally have access to the sea or at least do not exhibit the long distance marine dispersal often associated with larval fish. On the other hand, some populations are known to have recolonized, documenting that dispersal does occur. The available evidence suggests that (1) adult tidewater gobies rather than larvae are involved in dispersal (Hellmair and Kinziger 2014), (2) dispersal occurs in association with high stream-flow events that open estuaries to the sea during the winter rainy season (Lafferty et al. 1999), and (3) dispersal along the coast is greatly facilitated by sandy substrate and is limited by rocky coastal substrate. This last inference is consistent with the preference of this benthic fish for sandy bottoms for reproduction and is supported by mitochondrial sequence data (Dawson et al. 2001). This limited dispersal by tidewater goby contrasts with dispersal in the closely related arrow goby, which lives in open marine habitats permitting larval dispersal and exhibits minimal regional genetic differentiation (Dawson et al. 2002). The closest known source population to recolonize the Salinas River Lagoon is in Elkhorn Slough. The mouth of Elkhorn Slough is about 7 miles north of the Salinas River Lagoon and is connected to the lagoon via the Old Salinas River.

F2.1.4.3 Ecological Relationships

Tidewater goby generally live for only 1 year, with few individuals living longer than a year. Reproduction can occur at all times of the year (i.e., protracted iteroparity). Spawning activity peaks twice, once during the spring and again in the late-summer. Fluctuations in reproduction are probably due to death of breeding adults in early summer and colder temperatures or hydrologic disruptions in winter. Male tidewater gobies begin digging breeding burrows in relatively unconsolidated, clean, coarse sand, in April or May after lagoons close to the ocean. After hatching, the larval tidewater gobies emerge from the burrow and swim upward to feed on plankton. Juvenile tidewater gobies become benthic dwellers at 16–18 mm standard length. Tidewater gobies are known to be preyed upon by native species such as small steelhead, prickly sculpin, and staghorn sculpin. Tidewater goby feeds on a broad range of invertebrates and is the only fish species known to date that can fully digest invasive New Zealand mudsnails (*Potamopyrgus antipodarum*), which may constitute a high proportion of its diet when the snails are abundant (Hellmair et al. 2011

USFWS characterizes tidewater goby populations (i.e., localities) along the California coast as metapopulations (a group of distinct subpopulations that are genetically interconnected through occasional exchange of animals). While individual populations may be periodically extirpated under natural conditions, a metapopulation is likely to persist through colonization or recolonization events that establish new populations (U.S. Fish and Wildlife Service 2007 2007). different population structures across its geographic range (extinction–colonization dynamics in the south vs. drift in isolation in the north; Kinziger et al. 2015). Local extirpations may result from one or a series of factors, such as the drying up of some small streams during prolonged droughts, water diversions, and estuarine habitat modifications (2007). Some localities where tidewater gobies have been extirpated apparently have been recolonized when extant populations were present within a relatively short distance of the extirpated population (i.e., less than 6 miles). More recently, another tidewater goby researcher has suggested that recolonizations have typically been between populations separated by no more than 10 miles. Flooding during winter rains can contribute to recolonization of estuarine habitats where tidewater goby populations have previously been extirpated.

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F2.1.5 Population Status and Trends

F2.1.5.1 Population Trend

State

The population is presumably stable; however, no long-term monitoring program is available for the tidewater goby, and population dynamics are not well documented for this species. Population trends over the past 10 years or three generations is uncertain but probably within the natural range of variation (three generations span fewer than 10 years). Deriving population size estimates for tidewater goby is difficult because of the variability in local abundance. In addition, seasonal changes in distribution and abundance further hamper efforts to estimate population size, especially for a short-lived species. For example, when lagoons are breached due to flood events during the rainy seasons, tidewater goby populations decrease and then recover during the following summer. Tidewater goby populations also vary greatly with the varying environmental conditions (e.g., drought, El Niño) among years; this environmental variation is a normal phenomenon, but one that makes the determination of trends difficult.

Study Area

The tidewater goby population is presumably increasing. The species had not been documented in the Salinas River Lagoon from 1951 until 2013, when two individuals were found during routine Lagoon monitoring. By 2014, tidewater goby were the second most abundant fish species (after threespine stickleback) observed that year and the fifth most abundant out of 17 species captured over the 4-year survey period. A total of 58 tidewater gobies were observed in 2014, with individuals captured at three different sampling locations within the lagoon. It is likely that the gobies captured during 2013–2014 surveys dispersed from nearby Bennett Slough or Moro Cojo Slough, although no genetic studies have been conducted to confirm this hypothesis (78 FR 8,746-8819).

Recent survey information suggests that the tidewater goby population in the Salinas River Lagoon has most likely persisted since recolonization. As this species rarely lives longer than one year (Hellmair et al. 2014), continuous presence of tidewater goby in the Salinas River Lagoon (and the Old Salinas River) are a strong indication that the species can successfully reproduce in the Salinas River Lagoon over multiple generations. While the exact size of the population is unknown, repeated collections since 2013 confirm that the lagoon provides suitable habitat for tidewater goby growth, survival, and reproduction (Hellmair et al. 2018).

Despite only being found in low densities during the October 2018 survey, overall results suggest that the species was abundant during this time. It should be noted that tidewater goby populations can vary drastically in abundance from year to year – from thousands to millions – depending on whether conditions are favorable during their peak reproductive season (summer, when the likelihood of natural breaching is lowest; Hellmair et al. 2011). The length range of captured tidewater gobies (15 mm) is greater than that found for some tidewater goby populations along the North Coast, which are at an elevated risk of extirpation due to their constrained reproductive period. A reproductive period approximately four months in duration – as estimated for the Salinas River Lagoon population of tidewater goby, suggests a medium level of resilience to environmental disturbance.

F2.1.6 Threats

Tidewater goby is threatened by modification and loss of habitat resulting from coastal development, channelization of streams and estuaries, diversions of water flows, groundwater overdrafting, and alteration of water flows. Potential threats also include discharge of agricultural and sewage effluents, increased sedimentation from improper agricultural activities, unnatural breaching of estuaries and lagoons, upstream alteration of natural sediment flows, introduction of predatory fishes and invasive plants, direct habitat damage, and watercourse contamination resulting from vehicular activity in the vicinity of lagoons.

Coastal developments that modify or destroy coastal brackish-water habitat are a major factor adversely affecting tidewater goby. In many locations, the brackish zone, preferred by tidewater goby, has been modified or eliminated by human-created barriers such as dikes and levees. Coastal lagoons and marshes have been drained and reclaimed for agricultural, residential, and industrial developments. In addition, coastal road and railroad construction has severed the connection between marshes and the ocean, resulting in unnatural water temperature and salinity profiles, and waterways have been dredged for navigation and harbors, resulting in direct losses of wetland habitats as well as indirect losses due to associated changes in salinity. Ongoing threats include loss and alteration of habitat resulting from development projects, flood control, anthropomorphic breaching of coastal lagoons, and freshwater withdrawal. However, current laws and regulations have reduced or eliminated the threat of both large- and small-scale habitat loss and alteration.

Upstream water diversions adversely affect tidewater goby by altering downstream flows, thereby diminishing the extent of habitats that occurred historically at the mouths of many rivers and creeks in California. Alterations of flows upstream of coastal lagoons have already changed the distribution of downstream salinity regimes. Upstream water diversions may change the salinity distribution in estuaries and lagoons and may reduce the size and distribution of goby populations.

The accidental and purposeful introduction of native or nonnative species, particularly predatory fishes and amphibians, has been responsible for drastic reductions in populations of tidewater gobies at some sites (U.S. Fish and Wildlife Service 2007). The introduction of other nonnative species that may compete with tidewater gobies is another cause of decline.

About 50% of the remaining populations are considered vulnerable to extinction due to severe habitat degradation (U.S. Fish and Wildlife Service 2007). Populations in large habitats that are close to other occupied habitats are most likely to persist, but habitat alteration and introduced species may eliminate the species from even large habitats (Lafferty et al. 1999). Failure of tidewater gobies to recolonize habitats after local extirpation may be the result of habitat degradation of the extirpated locality, rather than an inability to recolonize (Lafferty et al. 1999).

F2.1.7 Recovery Planning

F2.1.7.1 Statewide

USFWS released the recovery plan for the tidewater goby in 2005 (U.S. Fish and Wildlife Service 2005). The primary objective of the plan was to manage the threats to and improve the population status of the tidewater goby sufficiently to warrant reclassification (from endangered to threatened status) or delisting. The species was given a recovery priority number of 7C (on a scale of 1–18), per criteria published in the Federal Register (48 FR 43098 and 51985), indicating a species with moderate threats and a high potential for recovery but with some degree of conflict between the species' recovery efforts and economic development. The strategy for achieving this objective is designed to (1) preserve the diversity of tidewater goby habitats throughout the range of the species, (2) preserve the natural processes of recolonization and population exchange that enable population recovery following catastrophic events, (3) and preserve the genetic diversity as it is understood now and in the future.

Recovery criteria were developed by subdividing the geographic distribution of the tidewater goby into 6 recovery units, encompassing a total of 26 Sub-Units defined according to genetic differentiation and geomorphology. According to USFWS (2005), downlisting of tidewater goby may be considered when:

- Specific threats to each metapopulation, such as habitat destruction and alteration (e.g., coastal development, upstream diversion, channelization of rivers and streams, discharge of agriculture and sewage effluents), introduced predators (e.g., centrarchid fishes), and competition with introduced species (e.g., yellowfin and chameleon gobies), have been addressed through the development and implementation of individual management plans that cumulatively cover the full range of the species.
- A metapopulation viability analysis based on scientifically credible monitoring over a 10-year period indicates that each Recovery Unit is viable. The target for downlisting is for individual Sub-Units within each Recovery Unit to have a 75 percent or better chance of persistence for a minimum of 100 years. Specifically, the target is for at least 5 Sub-Units in the North Coast Unit, 8 Sub-Units in the Greater Bay Unit (including the Salinas River), 3 Sub-Units in the Central Coast Unit, 3 Sub-Units in the Conception Unit, 1 Sub-Unit in the Los Angeles/Ventura Unit, and 2 Sub-Units in the South Coast Unit to individually have a 75 percent chance of persisting for 100 years.

If a metapopulation viability analysis projects that all recovery units are viable, and individual Sub-Units within each Recovery Unit have a 95% or better chance of persistence for a minimum of 100 years, then tidewater goby may be considered for delisting (U.S. Fish and Wildlife Service 2005).

F2.1.7.2 Study Area

The Salinas River is included in the GB11 sub-unit of the recovery plan (U.S. Fish and Wildlife Service 2005). The available tidewater goby habitat in the river encompasses approximately 100 hectares (250 acres). Approximately 20%of the adjacent land is owned and managed by the Salinas National Wildlife Refuge; the remaining adjacent lands are privately owned. At the time the recovery plan was published, tidewater gobies had not been observed in the river since 1951. The recovery plan notes the status of the Salinas River estuary as "Water Quality Limited" as designated by State Water Resources Control Board (State Water Board). Pollutants and stressors (and their respective potential sources in parentheses) are listed in the plan and include fecal coliform (past sewage discharge), pesticides (agriculture, irrigated crop production, agricultural storm runoff, agricultural irrigation tailwater, agricultural return flows, nonpoint source), nutrients (agriculture), salinity/chlorides (agriculture, natural sources, nonpoint source), and sedimentation/siltation (agriculture, irrigated crop production, range grazing-riparian and/or upland, agricultural storm runoff, road construction, land development, channel erosion, nonpoint source; U.S. Fish and Wildlife Service 2005).

Actions needed for recovery in the Salinas River and statewide include (1) monitoring, protecting and enhancing currently occupied tidewater goby habitat; (2) conducting biological research to enhance the ability to integrate land use practices with tidewater goby recovery and revise recovery tasks as pertinent new information becomes available; (3) evaluating and implementing translocation where appropriate; and (4) increasing public awareness about tidewater gobies.

In March 2014, the USFWS announced a 12-month finding on a petition to reclassify the tidewater goby as threatened under the ESA. After review of all available scientific and commercial information, USFWS found that downlisting the tidewater goby from endangered to threatened was warranted and proposed to reclassify tidewater goby as threatened under the ESA (79 FR 14340). However, to date, no reclassification has been made and tidewater goby are still federally listed as endangered.

F2.1.8 Existing Conservation Actions in the Study Area

Since the listing of tidewater goby in 1994, several conservation efforts have been undertaken by various federal, State, and local agencies and by private organizations. The following briefly describes some regulatory protection and conservation measures currently in place for the population as a whole and for the Salinas River subpopulation.

F2.1.8.1 Survey, Monitoring, and Research

USFWS has developed a survey protocol to facilitate the determination of presence or absence of the tidewater goby in habitats that have potential to support it (U.S. Fish and Wildlife Service 2005). The primary use for this protocol is for project-level surveys in support of requests for consultation under section 7 of the ESA, as amended. Additionally, this protocol may also be used for section 10(a)(1)(B) permit applications, and to determine general presence–absence for other management purposes. Several assessments of the tidewater goby population in various localities have been conducted using these methods.

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Attachment F2

Figures

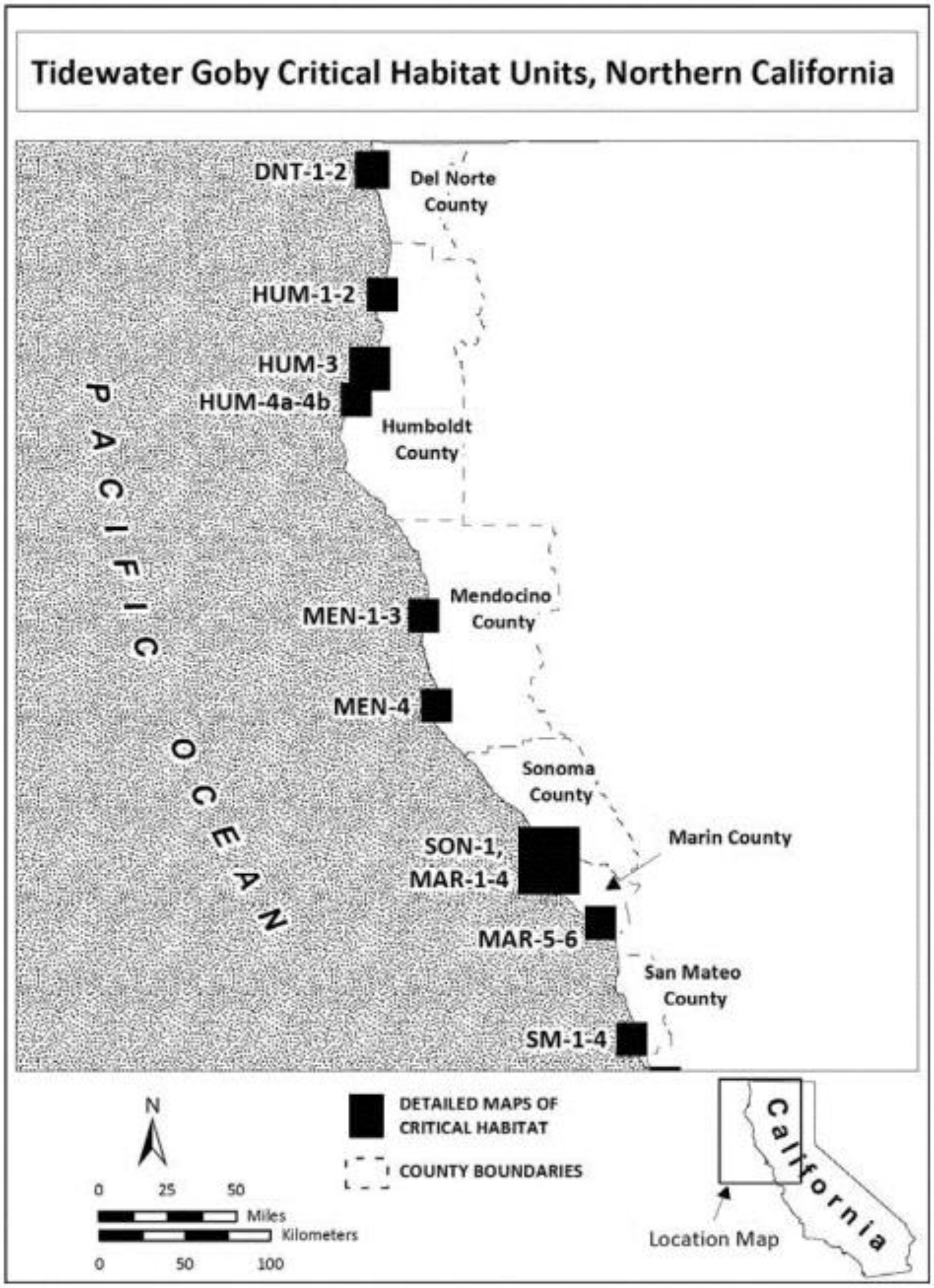

Figure F2-1a. Tidewater Goby Critical Habitat—Northern California

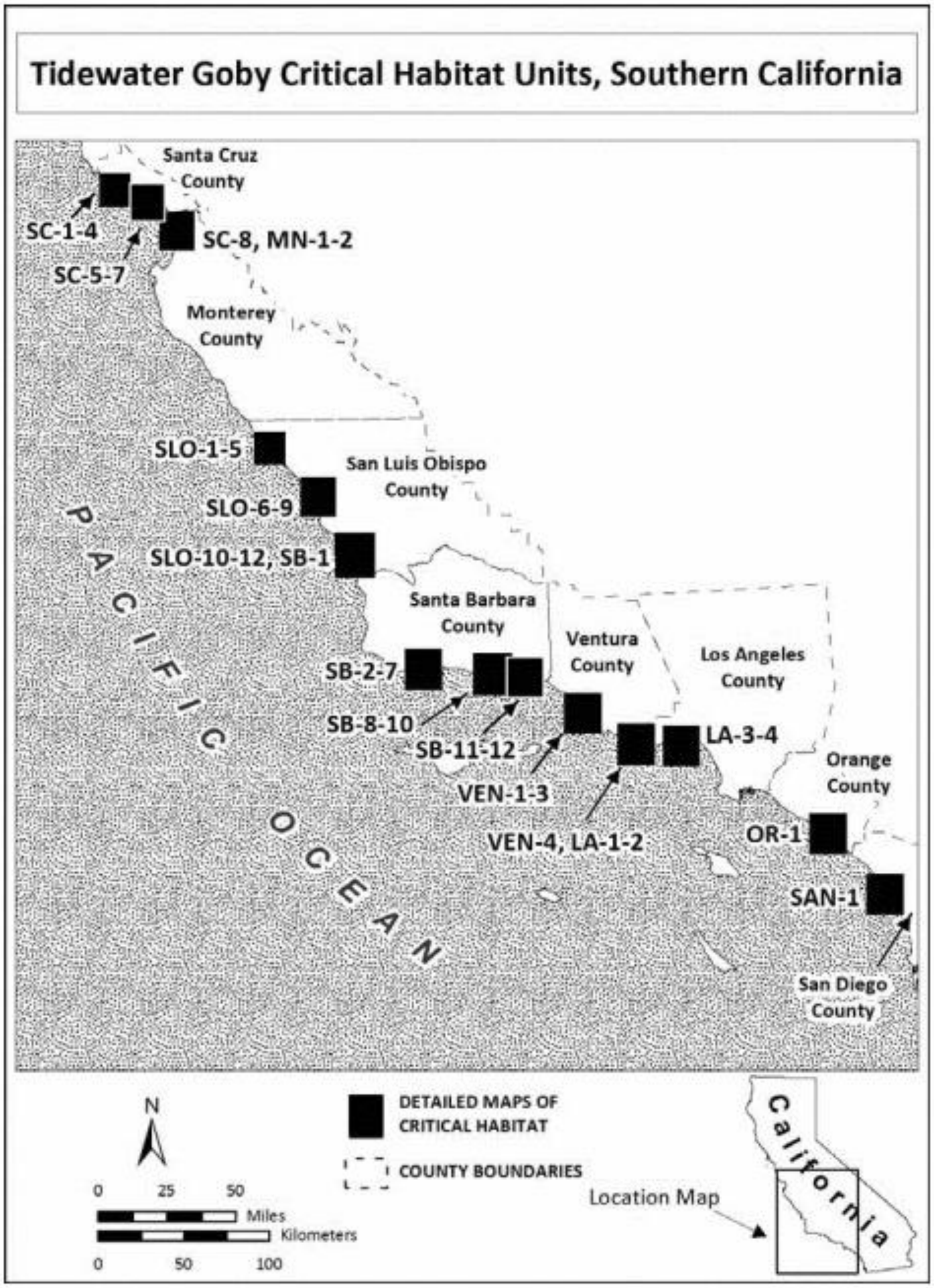

Figure F2-1b. Tidewater Goby Critical Habitat—Southern California

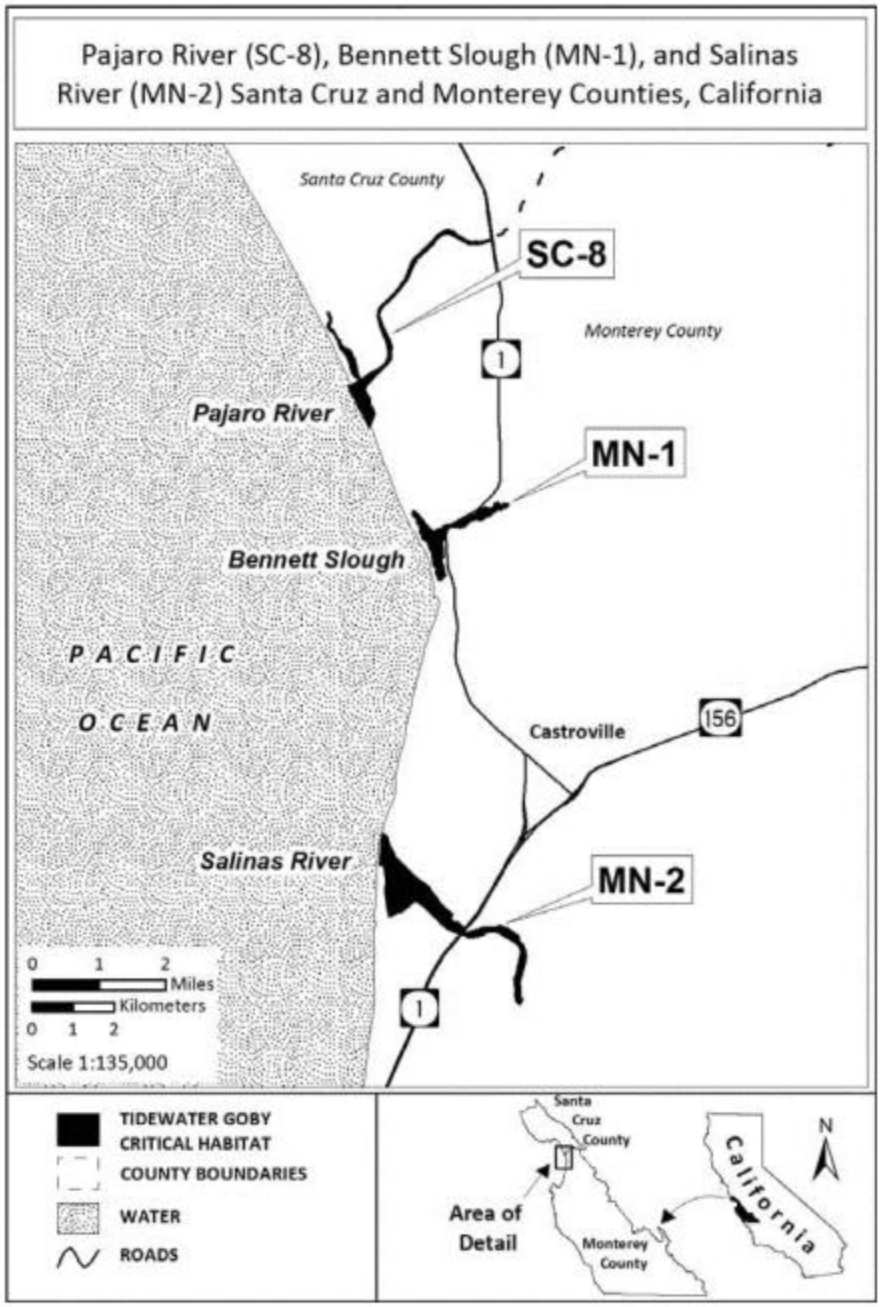

Figure F2-1c. Tidewater Goby Critical Habitat—Santa Cruz and Monterey Counties, California

Figure F2-2. Distribution of Tidewater Goby in California

F3.1 California Red-Legged Frog (*Rana draytonii)*

F3.1.1 Legal Status

F3.1.1.1 State

California red-legged frog is identified as a State Species of Special Concern (Thomson et al. 2016:100–105).

F3.1.1.2 Federal

California red-legged frog is federally listed as threatened throughout its range in California. It was listed by USFWS on May 23, 1996 (61 FR 25813).

F3.1.1.3 Critical Habitat

Critical habitat was designated in 2006 (71 FR 19243), and revised on March 17, 2010 (75 FR 12816).

F3.1.1.4 Notes

This species is synonymous with *Rana aurora draytonii.*

F3.1.2 Taxonomy

Rana draytonii was initially described as a distinct species. In 1917, Camp (Grinnell and Camp 1917 in Shaffer et al. 2004) reclassified it as a subspecies of *Rana aurora*. Based on DNA and morphological differences, Shaffer et al. (2004) suggested *R. draytonii* as distinct from *Rana aurora* and *Rana cascadae*; this distinction is recognized by CDFW and USFWS.

F3.1.3 Distribution

F3.1.3.1 State

Historical

Historically, the California red-legged frog ranged throughout the Sierra Nevada foothills and the Coast Range Mountains from southern Mendocino County south to Baja California Norte, Mexico (Thomson et al. 2016: 100-105) below 3,500 feet.

Recent

California red-legged frog is currently found primarily in central California coastal drainages from Marin County south to Baja California, Mexico, Sierra and in isolated drainages in the Sierra Nevada, northern Coast, and northern Transverse Ranges (U.S. Fish and Wildlife Service 2002).

F3.1.3.2 Study Area

Historical

There have been no comprehensive surveys for California red-legged frog within the Salinas River and its major tributaries. Historic occurrences are known from the following locations.

- Nacimiento River, near the Nacimiento-Ferguson Road (1981).
- San Lorenzo Creek, seven miles west of King City (1949).
- Chalone Creek in Pinnacles National Monument (1939) (McGraw 2008).

Recent

California red-legged frogs have been observed on a few occasions throughout the study area. An occurrence was reported near the east bank of the Salinas River in streamside emergent vegetation near river mile 5 in River Management Unit 7. In 1999, a juvenile California red-legged frog was observed along the edge of the Salinas River between river mile 5 and the lagoon (Monterey County Water Resource Agency 2016). Between 2002 and 2004, California red-legged frogs were observed in an unnamed tributary to Natividad Creek, northeast of the town of Salinas. There are several occurrences of California red-legged frogs in the sloughs, ditches, and other agricultural water features surrounding Oakdale and Prunedale, as well as in the region of Elkhorn Slough, northwest of the town of Salinas (California Department of Fish and Wildlife 2018).

Scientists from The Nature Conservancy have observed California red-legged frogs within Los Vaqueros Creek (Arroyo Seco tributary) and upper Gabilan Creek; staff from Pinnacles National Monument have reported them in Chalone Creek (McGraw 2008; California Department of Fish and Wildlife 2018). Several occurrences of California red-legged frog were identified in visual and aquatic surveys of stock ponds and other water features on Dorrance Ranch in the foothills west of Spence and Chualar between 2006 and 2012; several previously occupied locations on Dorrance Ranch were re-surveyed in 2014 with no detections found (California Department of Fish and Wildlife 2018).

F3.1.4 Natural History

F3.1.4.1 Habitat Requirements

California red-legged frogs breed in ponds (natural and artificial); they also use marshes, streams, freshwater sections of lagoons, and other waterways throughout their range. Breeding takes place primarily in ponds and less frequently in stream pools (Thomson et al. 2016:100–105). Breeding ponds typically require some density of emergent vegetation to which females deposit egg masses; however, adult density in breeding ponds was found to not be dependent on percent cover (0%, </= 15%, > 15%) of emergent vegetation (Thomson et al. 2016:100–105). Breeding pools must remain wetted for 15–23 weeks to allow for development from eggs to metamorphosed juveniles (U.S. Fish and Wildlife Service 2002).

After breeding, adults typically disperse to nearby upland habitat that includes shaded, slow-moving streams; spaces under downed trees, logs, rocks and vegetation; agricultural features such as drains, watering troughs, abandoned sheds, or hay-ricks; cracks in the bottom of dried ponds; small mammal burrows; and moist leaf litter (U.S. Fish and Wildlife Service 2002).

F3.1.4.2 Movement

Populations appear to consist of both migratory (11–22% of the adult population) frogs that move 660–9,240 feet and resident frogs that remain at the breeding site (Bulger et al. 2003). Fellers and Kleeman (2007) found that adult female frogs were more frequently migratory than males, although migration behavior did not differ between the sexes among those individuals that did migrate. Frogs have been observed to make long-distance movements that are straight-line, point-to-point migrations rather than using corridors between habitats. Dispersing frogs in northern Santa Cruz County traveled distances from 0.25 mile to more than 2 miles without regard to topography, vegetation type, or riparian corridors (U.S. Fish and Wildlife Service 2002).

F3.1.4.3 Ecological Relationships

Within the study area, California red-legged frog distribution and habitat requirements overlaps with habitat niches occupied by steelhead trout, tidewater goby, foothill yellow-legged frog, California tiger salamander, arroyo toad, and western pond turtle. Steelhead occupy the Salinas River and its undammed tributaries as well as the lagoon and require intact riparian corridors to maintain cool water temperatures and provide complex underwater habitats such as undercut banks. Tidewater goby occupies lagoon habitat and particularly low flow, backwater habitats. California tiger salamander occupies grasslands and oak woodlands with stock ponds or seasonal wetlands in coastal and inner coastal ranges. Arroyo toad occurs primarily in the upper tributaries of coastal drainages and occurs in sandy to gravelly streamside pools and adjacent riparian habitat. Western pond turtle occurs in a wide variety of permanent and intermittent aquatic habitats (e.g., streams and rivers, lagoon backwaters, lakes and ponds) and requires adjacent uplands with open grasslands and sandy soils. Conservation of these species and their habitat will likely benefit California red-legged frog (U.S. Fish and Wildlife Service 2002).

F3.1.5 Population Status and Trends

F3.1.5.1 Population Trend

State

California red-legged frog population has declined 70% throughout the range; it is extirpated from 24 of the originally occupied 46 counties (U.S. Fish and Wildlife Service 2002). Declines have occurred across the range of the species but have been greatest in the southern portion of the range. In central coastal California, populations are more robust (U.S. Fish and Wildlife Service 2007).

Study Area

California red-legged frogs were once widespread and abundant in the inner coast ranges between the Salinas and San Joaquin Valleys. Currently, no more than 10% of the historic localities within the Salinas River hydrographic basin and inner coast ranges still support the species (McGraw 2008).

F3.1.6 Threats

Threats to the California red-legged frog in the Salinas Valley include agriculture, livestock (cattle grazing and/or dairies), mining, nonnative species (bullfrogs [*Rana catesbeiana*], mosquito fish [*Gambusia affinis*], and predatory fish), recreation, urbanization and water management/diversions/reservoirs (U.S. Fish and Wildlife Service 2002). The greatest threat facing California red-legged frog is likely habitat loss and alteration as a result of urbanization and agriculture; agricultural development also increases pesticide exposure, which may have strong negative impacts, especially for cholinesterase-inhibiting pesticides although the species still persists in some dense agricultural settings in Monterey and Santa Cruz Counties (Thomson et al. 2016:100–105). Conversion of wetlands and modifications to wetland hydrology also likely have detrimental effects.

F3.1.7 Recovery Planning

F3.1.7.1 State

The California red-legged frog recovery plan was authored in 2002 by USFWS. The recovery plan identifies eight recovery units and 35 Core Areas; each recovery unit has different strategies to best meet the goal of delisting (U.S. Fish and Wildlife Service 2002).

F3.1.7.2 Study Area

The Central Coast and Diablo Range and Salinas Valley Recovery Units overlap with the study area. The status of recovery in the Central Coast portion of the study area is “high”; there are “many existing populations, many areas of high habitat suitability, and low to high levels of threat.” The status of recovery in the Diablo Range and Salinas Valley is “medium”; there are “numerous existing populations, some areas of medium habitat suitability and high levels of threats” (U.S. Fish and Wildlife Service 2002). The overall recovery strategy includes (1) protecting existing populations; (2) restoring and creating habitat the will be protected and managed in perpetuity; (3) surveying and monitoring populations and conducting research on the biology of and threats to the subspecies; and (4) reestablishing populations within the historic range (U.S. Fish and Wildlife Service 2002).

F3.1.8 Existing Conservation Actions in the Study Area

Elkhorn Slough National Estuarine Research Reserve (ESNERR) is a concentration of protected lands to the north of Salinas Lagoon. ESNERR is a federal reserve where research and management focuses on understanding and protecting ecosystem function and rare species habitat. In addition, there are several properties to the north, east, and south of the reserve that are also protected and managed to benefit ecosystem function. Freshwater restoration efforts target California red-legged frog, western pond turtle, and the Santa Cruz long-toed salamander.

To the south of the Salinas River Lagoon, near the city of Seaside, the Fort Ord Reuse Authority is currently working on the Fort Ord Habitat Conservation Plan (FOHCP). Once established, the FOHCP will protect and manage a portion of lands within the former base at Fort Ord to benefit native flora and fauna, particularly the rare and endangered species that occur within the planning area including the California red-legged frog and California tiger salamander.

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Appendix F4
California Tiger Salamander Species Account

F4.1 California Tiger Salamander (*Ambystoma californiense*)

F4.1.1 Legal Status

F4.1.1.1 State

California tiger salamander is listed as threatened under the CESA; this listing is not divided into DPSs (see *Federal* subsection below).

F4.1.1.2 Federal

There are several DPSs of California tiger salamander. The Central population of California tiger salamander was federally listed as threatened in 2004 (69 FR 47212). The Central California DPS is restricted to disjunct populations that form a ring along the foothills of the Central Valley and Inner Coast Range from San Luis Obispo, Kern, and Tulare Counties in the south, to Sacramento and Yolo Counties in the north.

F4.1.1.3 Critical Habitat

Critical Habitat was designated for the Central California population in 2005 (70 FR 49379).

F4.1.2 Taxonomy

Formerly regarded as a subspecies of *A. tigrinum*, California tiger salamander (*Ambystoma californiense*) was first described by Gray in 1853 based on specimens that had been collected in Monterey, California. Based on recent studies of the genetics, geographic distribution, and ecological differences among the members of *A. tigrinum* complex, California tiger salamander has been determined to represent a distinct species (69 FR 68567–68609). The biogeographical and genetic information supporting the recognition of the Santa Barbara County and Sonoma County populations as DPSs under the federal ESA are reviewed in those listing decisions (65 FR 3095; 3095–3109, 68 FR 53:13497–13520).

F4.1.3 Description

California tiger salamander is a large, stocky salamander, with a broad, rounded snout. Total body length of adults range approximately from 6 to 9.5 inches (U.S. Fish and Wildlife 2017). Females are about 7 inches. *Tiger* comes from the white or yellow bars on California tiger salamanders. The background color is black. The belly varies from almost uniform white or pale yellow to a variegated pattern of white or pale yellow and black (U.S. Fish and Wildlife Service 2017).

F4.1.4 Distribution

F4.1.4.1 State

Historical

California tiger salamander is endemic to California. Historically, California tiger salamander was endemic to the San Joaquin–Sacramento Valley and the Central Coast and was likely found in most low-elevation grassland-oak woodland plant communities (Shaffer et al. 2013 in U.S. Fish and Wildlife Service 2017). Although this species still occurs within much of its range, it has been extirpated from many areas it once occupied (Stebbins and McGinnes 1985; Fisher and Shaffer 1996). The loss of California tiger salamander populations has been due primarily to habitat loss within their historic range (Fisher and Shaffer 1996).

Based on genetic analysis, there are six populations of California tiger salamanders, distributed as follows: (1) Santa Rosa area of Sonoma County, (2) Bay Area (central and southern Alameda, Santa Clara, western Stanislaus, western Merced, and the majority of San Benito Counties), (3) Central Valley (Yolo, Sacramento, Solano, eastern Contra Costa, northeast Alameda, San Joaquin, Stanislaus, Merced, and northwestern Madera Counties), (4) southern San Joaquin Valley (portions of Madera, central Fresno, and northern Tulare and Kings Counties), (5) Central Coast range (southern Santa Cruz, Monterey, northern San Luis Obispo, and portions of western San Benito, Fresno, and Kern Counties), and (6) Santa Barbara County (U.S. Fish and Wildlife Service 2017).

Recent

The Central California tiger salamander occurs in the following counties: Alameda, Amador, Calaveras, Contra Costa, Fresno, Kern, Kings, Madera, Mariposa, Merced, Monterey, Sacramento, San Benito, San Mateo, San Joaquin, San Luis Obispo, Santa Clara, Santa Cruz, Stanislaus, Solano, Tulare, Tuolumne, and Yolo. Recent genetic studies also show that there has been little, if any, gene flow between the Central California DPS, the Sonoma County DPS, and the Santa Barbara County DPS for a substantial period of time (Shaffer et al. 2004, 2013). In addition, genetic studies have shown that within the Central California DPS there is genetic differentiation between four sub-groups that corresponds with the geographic distribution of those groups: (1) Southern San Joaquin Valley; (2) Central Valley; (3) Bay Area; and (4) Central Coast Range (Shaffer et al. 2004, 2013).

F4.1.4.2 Study Area

Central California tiger salamander is distributed throughout much of the study area. The California Natural Diversity Database includes approximately 90 occurrences of this species in the study area. The occurrences are mainly concentrated around Fort Hunter Liggett and the Fort Ord National Monument (California Department of Fish and Wildlife 2018).

F4.1.5 Natural History

F4.1.5.1 Habitat Requirements

California tiger salamanders require two major habitat components: aquatic breeding sites and terrestrial aestivation or refuge sites. California tiger salamanders inhabit valley and foothill grasslands and the grassy understory of open woodlands, usually within 1 mile of water (Thomson et al. 2016).

California tiger salamander is terrestrial as an adult and spends most of its time underground in subterranean refugia. Underground retreats usually consist of ground-squirrel burrows and occasionally human-made structures. Adults emerge from underground to breed, but only for brief periods during the year. Tiger salamanders breed and lay their eggs primarily in vernal pools and other ephemeral ponds that fill in winter and often dry out by summer (Loredo et al. 1996); they sometimes use permanent human-made ponds (e.g., stock ponds), reservoirs, and small lakes that do not support predatory fish or bullfrogs (see *Ecological Relationships* discussion below) (Stebbins 1972; U.S. Fish and Wildlife Service 2017). Streams are rarely used for reproduction, but California tiger salamanders have been reported in ditches with seasonal wetland habitat and in slow-flowing swales and creeks (Alvarez et al. 2013).

California tiger salamander is particularly sensitive to the duration of ponding in aquatic breeding sites. Because tiger salamanders have a long developmental period, the longest lasting seasonal ponds or vernal pools are the most suitable type of breeding habitat for this species; these pools are also typically the largest in size (Thomson et al. 2016). At least 10 weeks are required to complete metamorphosis (Feaver 1971); however, 4–5 months are usually required (Shaffer and Trenham 2005). Aquatic sites that are considered suitable for breeding should pond or retain water for a minimum of 10 weeks. Optimum breeding sites are ephemeral and should dry down for at least 30 days before the rain being in the fall (around August or September) to prevent nonnative predators from establishing (U.S. Fish and Wildlife Service 2017). Moreover, large vernal pool complexes, rather than isolated pools, probably offer the best quality habitat; these areas can support a mixture of core breeding sites and nearby refuge habitat (Jennings and Hayes 1994). USFWS (2017) states that, to remain viable, populations of California tiger salamanders require at least four ponds on preserves of no less than 3,398 acres, and that the ponds should have variation in depth and ponding duration so that at least some fill during different environmental conditions (e.g., low annual rainfall). USFWS determined the minimum preserve size based on the 1.3-mile maximum dispersal distance (i.e., a preserve with a radius of 1.3 miles is 3,398 acres). USFWS also explains that four ponds should provide the necessary amount of redundancy to ensure long-term habitat availability (U.S. Fish and Wildlife Service 2017).

The suitability of California tiger salamander habitat is proportional to the abundance of upland refuge sites are near aquatic breeding sites. California tiger salamanders primarily use California ground squirrel burrows as refuge sites (Loredo et al. 1996; Trenham 2001); Botta's pocket gopher burrows are also frequently used (Barry and Shaffer 1994; Jennings and Hayes 1994), as well as human-made structures. California tiger salamanders also use logs, piles of lumber, and shrink-swell cracks in the ground for cover (Holland et al. 1990). The presence and abundance of tiger salamanders in many areas are limited by the number of small-mammal burrows available; salamanders are typically absent from areas that appear suitable other than their lack of burrows. Loredo et al. (1996) emphasized the importance of California ground squirrel burrows as refugia for California tiger salamanders, and suggested that a commensal relationship existed between the California tiger salamander and California ground squirrel in which tiger salamanders benefit from the burrowing activities of squirrels. In a study conducted near Concord, California, Loredo et al. (1996) found that California ground squirrel burrows were used almost exclusively as refuge sites by California tiger salamanders. Also, tiger salamanders apparently do not avoid burrows occupied by ground squirrels (Loredo et al. 1996).

The proximity of refuge sites to aquatic breeding sites also affects the suitability of salamander habitat. California tiger salamanders are known to travel distances up to 1.4 miles from breeding sites (Trenham et al. 2001;Orloff 2011) and tend to live between approximately 100 yards and 0.6 mile (or more) from their breeding sites (Ford et al. 2013). Based on capture data from a single-season study at Olcott Lake in Jepson Prairie Preserve (Solano County), Trenham and Shaffer (2005) estimated that 95% of adult and subadult tiger salamanders occurred within approximately 0.4 mile of the breeding pond. However, their model also suggested that 85% of subadults were concentrated between 0.1 and 0.4 mile from the pond. During a 5-year study of a proposed housing development in the northwestern corner of the Antioch HCP/NCCP inventory area, Orloff (2011) recorded the majority of captured salamanders at least 0.5 mile from the nearest breeding pond and continuing work at Olcott Lake has documented a few individuals moving up to 0.6 mile from the pond (Orloff 2011). Therefore, although salamanders may migrate up to 1.4 miles from breeding sites, migration distances are likely to be less in areas supporting refugia closer to breeding sites. Also, habitat complexes that include upland refugia relatively close to breeding sites are considered more suitable because predation risk and physiological stress in California tiger salamanders probably increases with migration distance. Orloff (2011) also noted that California tiger salamanders also appear to have fidelity to specific areas of upland habitat.

F4.1.5.2 Reproduction

Adult California tiger salamanders migrate to and congregate at aquatic breeding sites during warm rains, primarily between November and February (Barry and Shaffer 1994). During the winter rains, tiger salamanders breed and lay eggs primarily in vernal pools and other shallow, ephemeral ponds that fill in winter and often dry by summer (Loredo et al. 1996). Eggs are laid singly or in clumps on both submerged and emergent vegetation and on submerged debris in shallow water. In ponds without vegetation, females lay eggs on objects on the pond bottom (Stebbins 1972; Barry and Shaffer 1994; Jennings and Hayes 1994). After breeding, adults leave the breeding ponds and return to their refugia (small mammal burrows, etc.).

After approximately 2 weeks, the salamander eggs begin to hatch into larvae. Once larvae reach a minimum body size they metamorphose into terrestrial juvenile salamanders. At a minimum, salamanders require 10 weeks living in ponded water to complete metamorphosis but in general development is completed in 3–6 months (U.S. Environmental Protection Agency 2013). If a pond dries prior to metamorphosis, the larvae will desiccate and die (69 FR 68567–68609). Juveniles disperse from aquatic breeding sites to upland habitats after metamorphosis (Holland et al. 1990).

F4.1.5.3 Movement

Adult California tiger salamanders migrate to and congregate at aquatic breeding sites during warm rains, primarily between November and April (Barry and Shaffer 1994; Orloff 2011; U.S. Fish and Wildlife Service 2017), with most activity occurring between December and February (California Department of Fish and Game 2010). Tiger salamanders are rarely observed except during this period (Loredo et al. 1996). Dispersal of juveniles from natal ponds to underground refugia occurs during summer months, when breeding ponds dry out. Juveniles disperse from breeding sites after spending a few hours or days near the pond margin (Jennings and Hayes 1994). Dispersal distance varies and may increase with an increase in precipitation (California Department of Fish and Wildlife 2010; Orloff 2011).

Some genetic data suggest low rates of California tiger salamander migration between vernal pool complexes (Irschick and Shaffer 1997) or metapopulations; this suggests that natural colonization after a local extirpation event may be unlikely (Fisher and Shaffer 1994 in U.S. Fish and Wildlife Service 2017). Trenham et al. (2001) showed that pool complexes occupied by California tiger salamander fit a metapopulation model, and dispersal rates between ponds may be high for both first-time and experienced breeders; and dispersal rates are probably high enough to prevent local extirpations within a pool complex. Wang et al. (2009) also found that dispersal through grassland is twice as costly for the species (i.e., difficulty of movement) as through chaparral, and that dispersal through oak woodland is the most costly for California tiger salamander movement. These differences in energetic expenditure may be due to differences in the density of grasses and forbs at ground level in each of these community types.

F4.1.5.4 Ecological Relationships

California tiger salamander larvae and embryos are susceptible to predation by fish (Stebbins 1972; Thomson et al. 2016), and larvae are rarely found in aquatic sites that support predatory fish (Shaffer and Stanley 1992 in Jennings and Hayes 1994;). Aquatic larvae are taken by herons and egrets and possibly garter snakes (Thomson et al. 2016). Shaffer et al. (1993) (in U.S. Fish and Wildlife Service 2017) also found a negative correlation between the occurrence of California tiger salamanders and the presence of bullfrogs; however, this relationship was detected only in unvegetated ponds. This suggests that vegetation structure in aquatic breeding sites may be important for survival. Because of their secretive behavior and limited periods above ground, adult California tiger salamanders have few predators (U.S. Fish and Wildlife Service 2000).

F4.1.6 Population Status and Trends

F4.1.6.1 Population Trend

State

Very little is known about the historical abundance of the Central California tiger salamander. There are no data regarding the absolute number of individuals of this species due to their time spent aestivating in burrows, which makes them difficult to observe (U.S. Fish and Wildlife Service 2017). The available data suggest that most populations consist of relatively small numbers of breeding adults; breeding populations in the range of a few pairs up to a few dozen pairs are common, and numbers above 100 breeding individuals are rare (California Department of Fish and Wildlife 2010). However, this species exhibits high variation in population numbers (Loredo et al. 1996; Trenham et al. 2000; U.S. Fish and Wildlife 2017).

Study Area

Similar to the statewide level of population status and trends, population status and trends in the study area are difficult to assess. Multiple studies on breeding Central California tiger salamander populations, most of which have shown large fluctuation in numbers of breeding adults as well as numbers of larvae produced. In Monterey County, Trenham et al. (2000) found the number of breeding adults visiting a pond varied from 57 to 244 individuals.

The native population of Central California tiger salamanders in the Salinas Valley have been seriously impacted by nonnative tiger salamanders. Approximately 65 years ago (or 30–40 salamander generations), thousands of nonnative eastern or barred tiger salamanders (*Ambystoma tigrinum mavortium*) were introduced from Texas and other parts of the southwestern United States into California by commercial bait dealers. These introductions have been traced to a suspected 15 locations found primarily in the Salinas Valley (Fitzpatrick and Shaffer 2007). Fitzpatrick and Shaffer (2007) conjecture that the hybrid swarm may have remained contained within the Salinas Valley during this time because of its relative high amount of perennial breeding ponds that contain nonnative tiger salamanders compared to other areas to the north that have more natural seasonal pools and native Central California tiger salamanders. Fitzpatrick and Shaffer (2007) determined that the distribution of introduced tiger salamander genes is largely confined to within 7.5 miles of introduction sites and in general, the distribution of hybridization seems to decrease in populations the further they are from the introduction sites in the Salinas Valley (Fitzpatrick and Shaffer 2007; Shaffer et al. 2013 in U.S. Fish and Wildlife Service 2017).

Fitzpatrick and Shaffer (2007) point out that the two areas of the Salinas River watershed with pure or nearly pure native tiger salamanders (Fort Ord and Peachtree Valley) have high concentrations of natural seasonal pools. All California tiger salamanders on Fort Hunter Liggett, which occur in at least 16 locations, are considered hybrids (U.S. Army Garrison Fort Hunter Liggett 2012).

F4.1.7 Threats

The two most significant threats to California tiger salamander throughout its range are widespread habitat loss and habitat fragmentation. These factors have both been caused by conversion of valley and foothill grassland and oak woodland habitats to agricultural and urban development (Stebbins 1985). For example, residential development and land use changes in the California tiger salamander's range have removed or fragmented vernal pool complexes, eliminated refuge sites adjacent to breeding areas, and reduced habitat suitability for the species over much of the Central Valley (Barry and Shaffer 1994; Jennings and Hayes 1994). Grading activities have probably also eliminated large numbers of salamanders directly (Barry and Shaffer 1994). Overall, approximately 75% of habitat for California tiger salamander within its historic range has been lost (Fisher and Shaffer 1996).

The primary threat to the Central Coast Recovery Unit (in which the study area is located) is hybridization between California tiger salamander and barred tiger salamanders (U.S. Fish and Wildlife Service 2017), as described under *Population Trends* above. Hybridization between native and exotic taxa, due to lack of reproductive isolation, can threaten native taxa by causing genetic swamping and reduced genetic diversity of native populations. In rare species such as California tiger salamander, hybridization can also lead to population extirpation. In a study of tiger salamander hybridization conducted in the Salinas Valley, Riley et al. (2003) found that the degree of genetic mixing between native and nonnative salamander depended on breeding habitat type. In artificial ponds, there appeared to be no barriers to gene exchange; however, in vernal pools, significantly fewer hybrid genotypes and more pure parental genotypes were found. These results suggest that the potential for reproductive isolation between the two taxa may be higher in native habitats (U.S. Fish and Wildlife Service 2017).

The introduction of bullfrogs, Louisiana red swamp crayfish, and nonnative fishes (mosquitofish, bass, and sunfish) into aquatic habitats has also contributed to declines in tiger salamander populations (Jennings and Hayes 1994; U.S. Fish and Wildlife Service 2000). These nonnative species prey on tiger salamander larvae and may eliminate larval populations from breeding sites (Jennings and Hayes 1994). At sites where aquatic vegetation is present, predation by exotic fish appears more likely to result in California tiger salamander extirpation than bullfrogs (Fisher and Shaffer 1996). Burrowing-mammal control programs are considered a threat to California tiger salamander populations. Rodent control through destruction of burrows and release of toxic chemicals into burrows can cause direct mortality to individual salamanders and may result in a decrease of available suitable habitat (U.S. Fish and Wildlife Service 2000). Vehicular-related mortality is an important threat to California tiger salamander populations (Barry and Shaffer 1994; Jennings and Hayes 1994). California tiger salamanders readily attempt to cross roads during migration, and roads that sustain heavy vehicle traffic or barriers that impede seasonal migrations may have impacted tiger salamander populations in some areas (Shaffer and Stanley 1992 in Jennings and Hayes 1992; Barry and Shaffer 1994). Therefore, establishing artificial barriers to movement or maintaining roads that support a considerable amount of vehicle traffic in areas that support California tiger salamander populations could severely degrade salamander habitat (Jennings and Hayes 1994).

F4.1.8 Recovery Planning

F4.1.8.1 State

The California tiger salamander recovery plan was authored in 2017 by USFWS. The three objectives of the recovery plan are to permanently protect habitat of self-sustaining populations of Central California tiger salamanders through the full range of the DPS, ameliorate or eliminate threats to the species, and restore or conserve a healthy ecosystem supportive of Central California tiger salamander populations. Recovery of this species can be achieved by addressing the conservation of remaining aquatic and upland habitat that provides essential connectivity, reducing fragmentation, and sufficiently buffers against encroaching development and intensive agricultural land uses. The recovery plan identifies 4 recovery units and 27 management areas; each recovery unit specifically prescribes the number of preserves necessary to recover the species. The recovery plan also defines the minimum preserve size as 3,398 acres with at least four ponds (U.S. Fish and Wildlife Service 2017).

F4.1.8.2 Study Area

The Central Coast Range Recovery Unit contains the following management units: Fort Ord, Carmel Valley, and Salinas Valley which overlap the study area. Within the three management units the goal is to protect nine preserves that total 30,583 acres (U.S. Fish and Wildlife Service 2017). The recovery plan (2017) states that most populations are not protected and have not been monitored for status and trends. Maintaining the native genetic integrity of Central California tiger salamanders within this recovery unit is a priority.

F4.1.9 Existing Conservation Actions in the Study Area

As noted in Section F3, *California Red-Legged Frog*, ESNERR is a concentration of protected lands to the north of the Salinas River Lagoon. ESNERR is a federal reserve where research and management focus on understanding and protecting ecosystem function and rare species habitat. In addition, there are several properties to the north, east, and south of the reserve that are also protected and managed to benefit ecosystem function. Freshwater restoration efforts target California tiger salamander, California red-legged frog, western pond turtle, and the Santa Cruz long-toed salamander.

To the south of the Salinas River lagoon, near the city of Seaside, the Fort Ord Reuse Authority is currently working on the FOHCP. Once established, the FOHCP will protect and manage a portion of lands within the former base at Fort Ord to benefit native flora and fauna, particularly the rare and endangered species that occur within the planning area including the California red-legged frog and California tiger salamander. However, even before the inception of the FOHCP, BLM has monitored the California tiger salamander on the Fort Ord National Monument (Fort Ord) to identify which resources are occupied by California tiger salamanders and/or nonnative salamanders. Genetic sampling efforts carried out by BLM and the Schaffer lab indicate that most ponds show some trace of nonnative genes. To date, the Fort Ord region is positioned at the northernmost edge of the hybrid swarm and the southernmost edge for the super-invasive only population.

As stated above under *Population Trends*, the hybrid tiger salamanders on Fort Hunter Liggett are considered a threat to native California tiger salamanders. There are no known native populations of tiger salamander on or adjacent to Fort Hunter Liggett, and eradication efforts would be resource intensive with unknown costs, effectiveness, and benefit. The current goal in the Fort Hunter Liggett Integrated Natural Resources Management Plan (INRMP) and Endangered Species Management Plan (ESMP) is to determine the cost and value of eradicating hybrid or nonnative tiger salamanders; this would provide valuable information for sites that have encroachment of nonnative tiger salamanders into native territories as well as for Fort Hunter Liggett (U.S. Army Garrison Fort Hunter Liggett 2012).

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Appendix F5
Least Bell's Vireo Species Account

F5.1 Least Bell's Vireo (*Vireo bellii pusillus*)

F5.1.1 Legal Status

F5.1.1.1 State

Least Bell's vireo is listed as endangered under the CESA.

F5.1.1.2 Federal

Least Bell's vireo was federally listed as endangered in 1986 (51 FR 16474). Least Bell's vireo is also protected under the Migratory Bird Treaty Act.

F5.1.1.3 Critical Habitat

Critical habitat for the vireo was designated in 1994 and is located in southern California (59 FR 4845).

F5.1.2 Taxonomy

There are four recognized subspecies of Bell's vireo (*Vireo bellii*): the eastern Bell's vireo (*V. b. belli),* the Texas Bell's vireo (*V. b. medius*)*,* the Arizona Bell's vireo (*V. b. arizonae*)*,* and the least Bell's vireo (*V. b. pusillus*). Least Bell's vireo constitutes all of the breeding birds in California apart from those in the vicinity of the Colorado River, which are the Arizona Bell's vireo (U.S. Fish and Wildlife Service 1998). The subspecies are isolated from one another during both the breeding and wintering seasons (Hamilton 1962). The first account of least Bell's vireo was written by J.G. Cooper based on two specimens collected near Manix in San Bernardino County, California (U.S. Fish and Wildlife Service 1998).

F5.1.3 Description

While all subspecies are similar in appearance, least Bell's vireos are mostly gray above and pale below, while easternmost birds are greenish above and yellowish below. Southwestern subspecies are intermediate in plumage characteristics.

F5.1.4 Distribution

F5.1.4.1 State

Historical

The Bell's vireo is a small migratory species that breeds in North America and overwinters primarily along the Pacific Coast in southern Mexico (Brown 1993). Breeding range for Bell's vireo is from north central to southwestern United States and into central Mexico (Brown 1993). Breeding has been documented from southwestern California and northwestern Baja California, Mexico, to central South Dakota, east to Illinois and northwestern Indiana, south to the Gulf Coast and into southern Sonora, Mexico (Brown 1993). Historically the breeding range of this species was broad in California and included both inland and coastal populations ranging from Red Bluff in Tehama County and from Santa Clara County to Baja California, Mexico (Grinnell and Miller 1944; U.S. Fish and Wildlife Service 1998). By the early 1980s, least Bell's vireo was extirpated from the northern portion of its range. Extant populations remained in counties south of Santa Barbara, with the most abundant populations found in San Diego County (U.S. Fish and Wildlife Service 1998).

Recent

Most breeding in California occurs in southwestern California and northwestern Baja California, Mexico (Brown 1993). Recently (2001) individuals have been reported during the breeding season as far north as Monterey County near San Juan Bautista (Roberson 2004; California Department of Fish and Wildlife 2018) and the San Joaquin Valley (U. S. Fish and Wildlife Service 2006), including Yolo County between 2010 and 2013 (California Department of Fish and Wildlife 2018). These sightings indicate this species may be expanding back into its historical range to the north of current populations.

F5.1.4.2 Study Area

In the study area, least Bell's vireo is a rare summer resident, occurring below 2,000 feet in willows (*Salix* spp.) and other valley foothill riparian habitat. Riparian vegetation communities within the study area provide moderate to highly suitable breeding, foraging, and cover habitat for least Bell's vireo. Sightings in recent years have been documented in southeastern Monterey County, but least Bell's vireo has not been observed in the Salinas River watershed since 1993 (McGraw and Boldero 2008).

F5.1.5 Natural History

F5.1.5.1 Habitat Requirements

Least Bell's vireo is an obligate riparian species which nest in California rose (*Rosa californica*), California grape (*Vitis californica*), poison oak (*Toxicodendron diversilobum*), giant creek nettle (*Urtica dioica L.* ssp*. holosericea*), and many other species, with an early-successional willow (Peterson et al. 2004) and Fremont's cottonwood (*Populus fremontii*) (Kus 2002) overstory. Suitable willow thickets are typically dense with well-defined vegetative strata or layers. The critical structural component of nesting habitat in California appears to be dense vegetation between 2 and 10 feet aboveground (Goldwasser 1981; Franzreb 1989; Brown 1993). According to USFWS (2001), the habitat elements essential for conservation of the taxon can be described as riparian woodland vegetation that generally contains both canopy and shrub layers and includes some associated upland habitats. Examples of suitable breeding habitat are broad cottonwood-willow woodlands with a dense shrubby understory and dense mule fat scrub. Most areas that support least Bell's vireo populations are areas where most woody vegetation is 5–10 years old (Gray and Greaves 1984; Franzreb 1989). Individuals occasionally forage and even nest in adjacent scrub or chaparral habitat. On its wintering grounds in southern Baja California, least Bell's vireo occurs primarily in mesquite scrub vegetation in arroyos (U. S. Fish and Wildlife Service 1998).

F5.1.5.2 Reproduction

Least Bell's vireos begin arriving on their breeding grounds in late March and begin nesting in early April (Kus 2002; Unitt 2004). Nesting is typically finished by the end of July (Kus 1999). Most pairs are monogamous during the breeding season (Brown 1993). Several factors may have an effect on breeding success, including development adjacent to riparian habitat, brown-headed cowbird *(Molothrus ater*) parasitism, and water management.

Female least Bell's vireos settle on male territories within 2 days of their arrival on the breeding grounds, and courtship begins immediately. In California, egg laying usually begins in April 1–2 days after nest construction is completed and lasts 4–5 days. Clutch size is usually 3–5 eggs; the mean clutch size of 196 California nests was 3.4 eggs (Franzreb 1989). Incubation begins once the first egg is laid and typically lasts 14 days.

Both sexes brood and feed the young, although females may brood more than males (Nolan 1960; Brown 1993). Young typically fledge 10–12 days after hatching. Therefore, the time to produce a successful brood is approximately 33–38 days. Most pairs in California produce one or two broods per season; however, up to four broods per season are occasionally produced (Franzreb 1989). When second broods are produced, a new nest is constructed immediately after the first brood has fledged or failed.

Many studies in California have reported the annual reproductive success of least Bell's vireos; in an area where the influence of brown-headed cowbird parasitism was manipulated, reproductive success ranged from 1.90 to 3.38 fledglings per breeding pair; however, where cowbirds were not manipulated, reproductive success ranged from 0.17 to 2.85 fledglings per breeding pair (Franzreb 1987).

F5.1.5.3 Movement

Least Bell's vireo is a neotropical migrant, leaving its breeding range in California to winter in Baja California, Mexico (Kus 2002). The species as a whole is known to be a nocturnal migrant (Brown 1993). Arrival of individuals in San Diego County ranges from mid-March to mid-April, with most departing from mid-August to late September (Unitt 2004). Least Bell's vireos exhibit strong territoriality. Males aggressively defend territories from neighboring birds by intensive singing or physical contact (Barlow 1962). Territories during the breeding season are mostly limited to areas within dense riparian corridors and are often linear in nature, following stream vegetation. Size of territories are dependent on the quality of breeding habitat available and the number of breeding individuals the area will support. Least Bell's vireo territory sizes vary considerably, and probably depend on habitat extent and quality, population density, and nesting stage. In California, reported territory sizes of least Bell's vireos are 0.5–4.0 acres (Gray and Greaves 1984) and 0.7–3.2 acres (U.S. Fish and Wildlife Service 1998 and referenced therein) (Table F5-1).

In California, fledglings have been reported to disperse 1 mile from their natal site by the time a second brood was produced (Gray and Greaves 1984) (Table F5-1). At a California study site, 15% of 312 nestlings or fledglings banded between 1979 and 1983 returned to breed the next year (Greaves 1987). At the same site, 18% of 203 nestlings or fledglings banded between 1987 and 1990 returned to breed the next year (Greaves 1991). Adult least Bell's vireos often exhibit strong breeding site fidelity, and nest sites are sometimes located within 3.3 feet of the previous year's nest (Greaves 1987).

Table F5-1. Documented Least Bell's Vireo Movement

Type	Distance/Area	Location of Study	Reference
Home range	0.5–4 acres	California	Gray and Greaves 1984
	1.5 acres	California	Collins et al. 1986
Dispersal	1 mile	California	Gray and Greaves 1984
	100–200 feet on day 14	Southern Indiana	Nolan 1960
Migration	From breeding grounds in Pacific Coast to southern Mexico	North America	Brown 1993

F5.1.5.4 Ecological Relationships

This species is dependent on dense, early-successional riparian corridors along watercourses for successful breeding. Riparian scrub habitats adjacent to these watercourses are equally important to the success of the species because they provide foraging opportunities as well as protection for nesting habitat. Brown-headed cowbirds have decimated least Bell's vireo populations throughout its breeding range. Dense riparian breeding habitat that is surrounded by agricultural lands or developed areas, like that of the study area, will facilitate brown-headed cowbird persistence and lower the breeding success of riparian nesting species like the least Bell's vireo.

F5.1.6 Population Status and Trends

F5.1.6.1 Population Trend

State

By the time least Bell's vireo was federally listed in 1986, the statewide population was estimated at 300 pairs. In 1996, the population had increased to 1,346 (U. S. Fish and Wildlife Service 1998); in 2000, the population had increased to 2,000 pairs (U.S. Fish and Wildlife Service 2001). The number of least Bell's vireo territories in 2005 was 2,968. The greatest population growth has been in San Diego and Riverside Counties, with smaller increases in Orange, Ventura, San Bernardino, and Los Angeles Counties and a sustained decrease in Santa Barbara County. The tremendous growth that most populations have experienced is attributed to an intensive brown-headed cowbird removal program that was initiated in some southern counties upon the listing of the species (U.S. Fish and Wildlife Service 2006). In addition to an increase in population size, it appears that least Bell's vireos are expanding their range and recolonizing sites that have been unoccupied for years. As populations continue to grow and disperse northward, they are beginning to reestablish in the central and northern portions of their historical breeding range (U.S. Fish and Wildlife Service 1998).

Study Area

The species was considered to be extirpated from Monterey County by around 1960, but three singing males were observed along the Salinas River near Bradley, upstream of the study area, in 1983; and in 1993, one singing male was observed near the same location (U.S. Fish and Wildlife Service 1998). In addition, habitat and least Bell's Vireo surveys were conducted along the Salinas River between the Highway 1 Bridge and Bradley between 1996 and 2001 as a component of the

2003 to 2008 Regional General Permit for the Salinas River Cooperative Monitoring Program (CMP). No least Bell's vireos were observed during this period.

F5.1.7 Threats

Habitat loss and degradation and nest parasitism by brown-headed cowbirds are identified as the biggest threats to least Bell's vireo populations (U. S. Fish and Wildlife Service 1998). Brown-headed cowbirds increased in the least Bell's vireo's range starting around 1915, presumably due to anthropogenic effects (e.g., residential development, agriculture, or grazing), with a decline in least Bell's vireo individuals beginning in the mid-1920s (Grinnell and Miller 1944). Nest parasitism by brown-headed cowbirds has greatly reduced nest success throughout most of its breeding range. In fragmented habitats, adult and nest predation by other nonnative predators (e.g., domestic cats, opossums, or Argentine ants) is an increased risk (U.S. Fish and Wildlife Service 2006). In the only formal study of nest predation on least Bell's vireo (Peterson et al. 2004), nearly half of predations were by the native California scrub-jay (*Aphelocoma californica*).

Least Bell's vireo habitat has decreased due to flood control, water impoundment and diversion, urban and rural development, agriculture and livestock grazing (U. S. Fish and Wildlife Service 2006). Riparian habitat connectivity has been improving along some major rivers in California, due to giant reed (*Arundo donax*) removal, restoration, and reductions in high impact activities (e.g. sand mining), but fragmentation is still occurring along lower order tributary streams due to urban development and flood control (U.S. Fish and Wildlife Service 2006). However, riparian areas are often along edges of agricultural or urban areas; vireo territories along these habitats are less productive than those adjacent to native upland habitats (U.S. Fish and Wildlife Service 2006).

F5.1.8 Recovery Planning

F5.1.8.1 State

According to the draft recovery plan for the least Bell's vireo (U.S. Fish and Wildlife Service 1998), the strategy for recovery of this species will involve (1) brown-headed cowbird control, (2) riparian habitat creation and restoration, (3) nest monitoring to remove brown-headed cowbird eggs and to obtain population data, and (4) control of nonnative predators (e.g., cats, Argentine ants). Additional management tools are shown below (U. S. Fish and Wildlife Service 1998, 2006).

- Water management on rivers.
- Control of nonnative vegetation.
- Grazing restrictions in and adjacent to riparian habitats.
- Habitat acquisition and management in perpetuity.

F5.1.8.2 Study Area

There have been no recovery-planning efforts for least Bell's vireo in the Salinas River watershed or in Monterey County (Zefferman pers. comm. 2018). USFWS collected census data from volunteer surveys conducted between 1983 and 1984, and other least Bell's vireo sightings have been documented incidentally during surveys for other listed species (California Department of Fish and Wildlife 2018).

More recently (as described below under *Existing Conservation Actions in the Study Area*), a giant reed removal project began along the Salinas River in 2014 and is expected to benefit least Bell's vireo (Zefferman pers. comm. 2018). Focused least Bell's vireo surveys conducted from King City to Soledad along the length of the Salinas River in 2017 did not identify any individuals; however, habitat conditions were suitable for the species throughout the survey area, although the taller cottonwoods were dead (Colibri Ecological Consulting 2017). General nesting bird surveys conducted annually in the same region since 2014 have not documented any least Bell's vireo (Zefferman pers. comm. 2018).

F5.1.9 Existing Conservation Actions in the Study Area

Efforts to control invasive giant reed and tamarisk (*Tamarix* spp.) and restore riparian habitat in the Salinas River watershed are underway in Monterey County, and major efforts are administered by the Resource Conservation District of Monterey County and implemented through partnerships with state and local agencies as well and private land-owners and non-profit organizations (Resource Conservation District of Monterey County 2018).

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Appendix F6
Western Snowy Plover Species Account

F6.1 Western Snowy Plover (*Charadrius alexandrinus nivosus)*

F6.1.1 Legal Status

F6.1.1.1 State

Western snowy plover is identified as a Species of Special Concern.

F6.1.1.2 Federal

The Pacific Coast population of western snowy plover was listed as federally threatened on March 5, 1993 (58 FR 12864). It is also protected under the Migratory Bird Treaty Act.

F6.1.1.3 Critical Habitat

Critical habitat was designated on June 19, 2012 (77 FR 36728). The closest critical habitat unit is in Elkhorn Slough, just north of the study area.

F6.1.2 Taxonomy

Two subspecies of snowy plover are recognized in North America: the western snowy plover (*Charadrius alexandrinus nivosus*) and the Cuban snowy plover (*C. a. tenuirostri*). The Cuban snowy plover breeds along the gulf coast south to the Caribbean (U.S. Fish and Wildlife Service 2007). The western snowy plover is a small shorebird native to North America in the family Charadriidae and consists of Pacific coastal and interior populations. Breeding data indicate that the Pacific coast population of western snowy plover are distinct from the population in the interior (U.S. Fish and Wildlife Service 2007). The Pacific Coast population was listed as threatened under the ESA in 1993 with the listed population defined as "those individuals that nest within 50 miles of the Pacific Ocean on the mainland coast, peninsulas, offshore islands, bays, estuaries, or rivers of the United States and Baja California, Mexico."

F6.1.3 Description

Western snowy plover is a small shorebird that is approximately 5.9–6.6 inches long. The plover's body is pale-gray brown above and white below, with a white hindneck collar. The bill and legs are blackish. In breeding plumage, the males have black markings and the females have dark brown markings on the head and breast (U.S. Fish and Wildlife Service 2007).

F6.1.4 Distribution

F6.1.4.1 State

Historical

Historically found along the entire California coast, western snowy plover was once more widely distributed and abundant throughout its range, especially in southern California (U.S. Fish and Wildlife Service 2007).

Recent

The current Pacific Coast breeding range of the western snowy plover extends from Damon Point, Washington, to Bahia Magdelena, Baja California, Mexico. The population is sparse in Washington, Oregon, and northern California. In 2006, estimated populations were 2,231 adults in coastal California and San Francisco Bay (71 FR 20,607–20,624). Eight geographic areas support over three-quarters of the California coastal breeding population: San Francisco Bay, Monterey Bay, Morro Bay, the Callendar Mussel Rock Dunes area, the Point Sal to Point Conception area, the Oxnard lowland, Santa Rosa Island, and San Nicolas Island (U.S. Fish and Wildlife Service 2007).

F6.1.4.2 Study Area

The study area includes 11 western snowy plover nesting areas spanning the length of the coast between Moss Landing and Sand City, which include the Salinas River National Wildlife Refuge and the Salinas River State Beach. These nesting sites are managed and monitored by the CDFW and/or USFWS.

F6.1.5 Natural History

F6.1.5.1 Habitat Requirements

Sparsely-vegetated sandy beaches and dunes, beaches at river and creek mouths, and salt pans at estuaries and lagoons provide the primary coastal nesting habitat for western snowy plover. Less commonly used nesting habitats include bluff-backed beaches, dredged material disposal sites, salt pond levees, dry salt ponds, and gravel bars (U.S. Fish and Wildlife Service 2007). Driftwood, kelp, and dune plants provide cover for chicks that crouch near objects to hide from predators. Invertebrates are often found near debris, so driftwood and kelp are also important for harboring western snowy plover food sources (Page et al. 2009).

F6.1.5.2 Reproduction

Breeding and nesting occurs March through September, and nests are found above the high tide line on sandy, open ground. Nests consist of a shallow scrape or depression, sometimes lined with beach debris (e.g., small pebbles, shell fragments, plant debris, and mud chips); nest lining increases as incubation progresses. They are monogamous by clutch and can have multiple clutches per year with two to six eggs per clutch. Both the male and female incubate the eggs. The young are precocial and will leave the nest within hours of hatching in search of food. Fledging is reached at approximately 1 month after hatching but the young will rarely remain in the nesting territory until fledging. Typically, males will continue to care for and feed the young while the female initiates a new nest. Western snowy plovers are highly sensitive to disturbance and may abandon their nests if disturbed (U.S. Fish and Wildlife Service 2007). Western snowy plovers maintain high site fidelity, returning to the same area to breed year after year.

F6.1.5.3 Movement

Some coastal populations of western snowy plovers remain in their breeding sites year-round, while others migrate north or south for the winter where they winter in coastal areas from southern Washington to Central America. The migrants vacate California coastal nesting areas primarily from late June to late October (Page et al. 2009). For those that remain, the majority concentrate on sand spits and dune-backed beaches. In winter, western snowy plovers are found on many of the beaches used for nesting, as well as some beaches where they do not nest. They also occur in human-made salt ponds and on estuarine sand and mud flats (U.S. Fish and Wildlife Service 2007).

Most western snowy plovers that nest inland migrate to the coast for the winter (Page et al. 2009). Thus, the flocks of non-breeding birds that begin forming along the U.S. Pacific Coast in early July are a mixture of adult and hatching-year birds from both coastal and interior nesting areas (U.S. Fish and Wildlife Service 2007).

F6.1.5.4 Ecological Relationships

Western snowy plovers forage on aquatic and terrestrial invertebrates in wet sand and wrack in the intertidal zone, in dry sand above the high tide line, on slat pans, spoil sites, and along the edges of lagoons, salt marshes, and salt ponds and occasionally glean insects from low growing vegetation. They have also been observed foraging in shallow water and foraging for flying insects (U.S. Fish and Wildlife Service 2007). Thus, the presence of kelp, stagnant water, or other insect-attractants along the coastline provide the western snowy plover with an important source of insect prey.

F6.1.6 Population Status and Trends

F6.1.6.1 Population Trend

State

By the late 1970s, nesting western snowy plovers were absent from 33 of 53 locations with breeding records prior to 1970 (U.S. Fish and Wildlife Service 2007). An estimated 1,593 adult western snowy plovers were seen during pioneer surveys. Western snowy plover populations in California have fluctuated between roughly 1,000 and 2,000 birds over the past 30 years. By 2000, populations had declined further to 71% of the late 1970s levels along the California coast (U.S Fish and Wildlife Service 2007).

Study Area

In the Monterey Bay region, the number of western snowy plovers is on the rise, and in 2017 there were an estimated 403 snowy plovers, which was a decrease from several previous years but an overall increase from 146 in 1999. The yearly average productivity in 2017 was 1.33 fledged chick per male (Neuman et al. 2018).

F6.1.7 Threats

Threats to western snowy plover include habitat degradation caused by human disturbance, urban development, introduced beachgrass (*Ammophila spp.),* and expanding predator populations including ravens and skunks (U.S. Fish and Wildlife Service 2007).

F6.1.8 Recovery Planning

F6.1.8.1 State

According to the 2007 USFWS recovery plan for the western snowy plover, the plan's objective is to remove the Pacific coast population of the western snowy plover from the List of Endangered and Threatened Wildlife and Plants by (1) increasing population numbers distributed across the range of the Pacific coast population of the western snowy plover; (2) conducting intensive ongoing management for the species and its habitat and developing mechanisms to ensure management in perpetuity; and (3) monitoring western snowy plover populations and threats to determine success of recovery actions and refine management actions.

The Pacific coast population will be considered for delisting when the following criteria have been met (U.S. Fish and Wildlife Service 2007).

- An average of 3,000 breeding adults has been maintained for 10 years.
- An average productivity of at least one fledged chick per male has been maintained in each recovery unit in the last 5 years prior to delisting.

Mechanisms are in place to ensure long-term protection and management of breeding, wintering, and migration areas to maintain the population size and productivity noted above.

F6.1.8.2 Study Area

To recover the species, the Sonoma to Monterey County recovery unit, which includes the study area, must maintain 400 breeding adults for 10 years (U.S. Fish and Wildlife Service 2007), as well as one fledged chick per male for 5 years.

F6.1.9 Existing Conservation Actions in the Study Area

There are a number of ongoing conservation actions that occur on federal and state land in the study area, including the following (U.S. Fish and Wildlife Service 2007).

- Exclosures and fencing.
- Law enforcement.
- Predator control.
- European beachgrass control.
- Annual population monitoring.
- Public education and outreach.
- Section 6 cooperative agreements.

In addition to the conservation actions listed above, a snowy plover rehabilitation program has been operating since 2000 at the Monterey Bay Aquarium (Monterey Bay Aquarium 2018) where egg incubation, bird rearing, rehabilitation, and banding take place. In addition, other groups are implementing habitat restoration and invasive plant removals to enhance and restore snowy plover habitat.

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Appendix F7
San Joaquin Kit Fox Species Account

F7.1 San Joaquin Kit Fox (*Vulpes macrotis mutica)*

F7.1.1 Legal Status

F7.1.1.1 State

San Joaquin kit fox is listed as threatened under the CESA.

F7.1.1.2 Federal

San Joaquin kit fox was listed as federally endangered in 1967 (32 FR 4001). Critical habitat has not been designated for San Joaquin kit fox.

F7.1.2 Taxonomy

The San Joaquin kit fox is a subspecies of the kit fox (*Vulpes macrotis*), the smallest member of the dog family in North America. Though there has been some debate as to the taxonomic relationship among North American arid land foxes, the San Joaquin kit fox remains a distinct subspecies due to its limited range in California. Descriptions of the species' physical characteristics can be found in U.S. Fish and Wildlife Service (1998).

F7.1.3 Description

San Joaquin kit fox is the largest subspecies of kit fox in terms of skeletal measurements, body size, and weight; males average 31.7 inches, and females average 30.3 inches. The average weight of adults is between 4.6 and 5 pounds. Kit foxes have a small, slim body; large ears set close together, and a long, bushy tail which tapers at the tip. The coloration of their coat varies but typically they are buff, tan, grizzled, or yellow-gray. The tail is distinctly black tipped (U.S. Fish and Wildlife Service 1998).

F7.1.4 Distribution

F7.1.4.1 Statewide

Historical

Historically, San Joaquin kit fox ranged throughout most of San Joaquin Central Valley from southern Kern County north to Tracy, San Joaquin County, on the west side, and near La Grange, Stanislaus County, on the east side.

Recent

Currently, the entire range of the kit fox appears to be similar to what it was at the time of USFWS's 1998 *Recovery Plan for Upland Species of the San Joaquin Valley*; however, population structure has become more fragmented, at least some of the resident satellite subpopulations are frequented by dispersers rather than resident animals (U.S. Fish and Wildlife Service 2010). San Joaquin kit foxes occur in some areas of suitable habitat on the floor of the San Joaquin Valley and in the surrounding foothills of the Coast Ranges, Sierra Nevada, and Tehachapi Mountains from Kern County north to Contra Costa, Alameda, and San Joaquin Counties (U.S. Fish and Wildlife Service 1998). There are known occurrences in Alameda, Contra Costa, Fresno, Kern, Kings, Madera, Merced, Monterey, San Benito, San Joaquin, San Luis Obispo, Santa Barbara, Santa Clara, Stanislaus, and Tulare Counties (California Department of Fish and Wildlife 2018). The largest extant populations of kit fox are in Kern County (Elk Hills and Buena Vista Valley) and San Luis Obispo County in the Carrizo Plain Natural Area (U.S. Fish and Wildlife Service 1998).

F7.1.4.2 Study Area

Extant

San Joaquin kit fox may be extirpated from San Joaquin County (U.S. Fish and Wildlife Service 2010). Past occurrences are concentrated along the Salinas Valley from Soledad southward and at Camp Roberts at the southern end of the study area, in addition to few occurrences documented in Fort Hunter Liggett (California Department of Fish and Wildlife 2018). USFWS considers the population at Fort Hunter Liggett to be extirpated (but with occasional sightings of presumed dispersers) and the population at Camp Roberts to be potentially extirpated (U.S. Fish and Wildlife Service 2010).

F7.1.5 Natural History

F7.1.5.1 Habitat Requirements

San Joaquin kit foxes favor shrublands and grasslands in dry arid climates and will also occupy areas with low to moderate oil and gas production activities and urban areas. Use of agricultural lands is limited to occasional foraging in irrigated crops and orchards, but only when such lands are adjacent to natural habitat. Kit foxes prefer habitats with well-drained sandy to loamy soils, which support their preferred prey (kangaroo rats) and allow for the excavation of dens (Cypher et al. 2012). Dens are generally located in open areas with grass or grass and scattered brush, and seldom occur in areas with thick brush. Preferred sites are located in relatively flat and well-drained terrain (U.S. Fish and Wildlife Service 1998; Roderick and Mathews 1999). They are seldom found in areas with shallow soils due to high water tables (McCue and O'Farrell 1981) or impenetrable bedrock or hardpan layers (O'Farrell and Gilbertson 1979). However, kit foxes may occupy soils with a high clay content where they can modify burrows dug by other animals, such as California ground squirrels (*Spermophilus beecheyi*), kangaroo rats, and badgers (Orloff et al. 1986; Cypher et al. 2012).

F7.1.5.2 Reproduction

Kit foxes can, but do not necessarily, breed their first year. Sometime between January and late March, one to six pups are born per litter, with an average of four (Cypher et al. 2012). The annual reproductive success for adults can range widely but is generally below 60% (Cypher et al. 2012);

immature reproductive success ranges between 14 and 21% (Clark et al. 2007). Kit fox pups emerge from dens at approximately 1 month of age and some disperse after 4–5 months, usually between July and September. The mean age of dispersal is 8 months and typically begins in June and peaks in July (Cypher et al. 2012).

Reproductive success is strongly influenced by food availability (Cypher et al. 2012). Population growth rates generally vary positively with reproductive success and kit fox density is often positively related to both current and the previous year's prey availability (Cypher et al. 2014). Prey abundance is generally strongly related to the previous year's effective (October to May) precipitation, which influences seed production for granivorous rodents (U.S. Fish and Wildlife Service 2010).

F7.1.5.3 Movement

Kit foxes may range up to 20 miles at night during the breeding season and somewhat less (6 miles) during the pup-rearing season (Koopman et al. 2000). The species can readily navigate a matrix of land use types. Home ranges vary from less than 1 square mile up to approximately 12 square miles (Knapp 1978; Spiegel and Bradbury 1992; White and Ralls 1993). The home ranges of pairs or family groups of kit foxes generally do not overlap (White and Ralls 1993). This behavior may be an adaptation to periodic drought-induced scarcity in prey abundance.

F7.1.5.4 Ecological Relationships

San Joaquin kit foxes prey upon a variety of small mammals, ground-nesting birds, and insects. They are in turn subject to predation by such species as coyote, nonnative red foxes, domestic dog, eagles, and large hawks (U.S. Environmental Protection Agency 2013). Standley and others (1992) determined that coyotes were responsible for 45.8% of kit fox deaths (11 of 24 individuals) during a 3-year study at Camp Roberts in southern Monterey County.

F7.1.6 Population Status and Trends

F7.1.6.1 Population Trend

State

In the 1983 recovery plan (U.S. Fish and Wildlife Service 1983), O'Farrell estimated that the range-wide population of adult kit fox prior to 1930 may have been between 8,667 and 12,134 animals, assuming an occupied range of 8,667 square miles, and assuming densities of 1.04 to 1.55 adult kit fox per square mile. Previously (1969–1975) various biologists had provided estimates of the total kit fox population that varied between 1,000 and 14,800 (Morrell 1975 in U.S. Fish and Wildlife Service 1983). In the 1983 recovery plan, O'Farrell adjusted Morrell's estimates to account for agricultural lands and provided a corrected population estimate for 1975 of 6,961 adult kit fox. When compared to the pre-1930 estimate, the change represented a possible population decline of 20–43%(U.S. Fish and Wildlife Service 1983).

Relatively recent population estimates are only available for the National Petroleum Reserves in California (NPRC) and the Carrizo Plain National Monument. Surveys on the 77,000-acre NPRC in western Kern County provided population estimates that ranged from 262 down to 74 in the period from 1981 to 1983 (Harris 1987 in U.S. Fish and Wildlife Service 2010), and that fluctuated between

46 and 363 adults from 1983 to 1995 (Warrick and Harris 2001 in U.S. Fish and Wildlife Service 2010). The only estimate for the Carrizo Plain provides is a population size of between 251 and 610 individuals, although the estimate may be high (Bean and White 2000 in U.S. Fish and Wildlife Service 2010). The Carrizo Plain is thought to have the largest kit fox population remaining in California (Carrizo Plains Conservancy 2018).

Study Area

Spatial distribution of the kit fox has become increasingly fragmented since listing. San Joaquin kit fox populations within the study area are considered to be satellite populations, which are areas that contain lower quality or more fragmented habitat and small or even intermittently present populations (Cypher et al. 2014). A total of 54 occurrences of San Joaquin kit fox have been documented in the study area since 1971 (California Department of Fish and Wildlife 2018). Although survey efforts have likely varied over the years in some areas, kit fox sightings have declined in areas with ongoing surveys; 30 of the 54 occurrence records in the study area were documented between 1970 and 1990 (California Department of Fish and Wildlife 2018). San Joaquin kit foxes have not been observed in the study area since 2007 and are considered potentially extirpated (California Department of Fish and Wildlife 2018).

F7.1.7 Threats

Habitat loss, fragmentation, and degradation from agricultural, urban, and industrial development continue to be the primary threats to San Joaquin kit fox throughout its range. Livestock grazing is not thought to be detrimental to kit foxes (Orloff et al. 1986; U.S. Fish and Wildlife Service 2010), but it may affect the number of prey species available, depending on the intensity of grazing (U.S. Fish and Wildlife Service 1998). In some areas, livestock grazing may benefit kit foxes by reducing shrub cover and maintaining grassland habitat.

F7.1.8 Recovery Planning

F7.1.8.1 Statewide

USFWS approved a federal recovery plan for the San Joaquin kit fox in 1983, and in 1998 it approved an updated multi-species recovery plan that includes the San Joaquin kit fox. The goal of the 1998 recovery plan for San Joaquin kit fox is to work towards the establishment of a viable complex kit fox population on private and public lands throughout its historic range. Conserving a number of populations of various sizes in strategic locations will be a necessary foundation for recovery. The recovery strategy for San Joaquin kit fox hinges on the protection and management of three geographically-distinct populations in the Carrizo Plains National Monument in San Luis Obispo County, natural lands of western Kern County, and the Ciervo-Panoche Natural Area of western Fresno and eastern San Benito Counties (U.S. Fish and Wildlife Service 1998). USFWS (1998) also states that in addition to land retirement and habitat restoration and management, research is also another important component of the recovery plan for San Joaquin kit fox. Habitat acquisition, large-scale habitat surveys, research into the ecology of the kit fox population, and public education have occurred for the recovery of San Joaquin kit fox since the time of its listing as endangered (U.S. Fish and Wildlife Service 1998).

F7.1.8.2 Study Area

Mitigation in the form of management and research was granted to both the Army National Guard (Camp Roberts) and Department of Defense (Fort Hunter Liggett) within the study area (U.S. Fish and Wildlife Service 1998).

Fort Hunter Liggett has an INRMP to manage natural resources within the army base to mitigate negative impact and enhance positive effects on the regional ecosystems, which includes an ESMP for San Joaquin kit fox. The goals of the INRMP and ESMP are to implement a San Joaquin kit fox management plan that (1) minimizes the potential for take of kit foxes while allowing the Fort Hunter Liggett base operations and military training to meet current and future mission standards and (2) establishes a protocol for monitoring for presence of kit foxes and red foxes on Fort Hunter Liggett (U.S. Army Garrison Fort Hunter Liggett 2012).

EG&G Energy Measurements has conducted research on San Joaquin kit fox on Camp Roberts for over 20 years. Reproduction (Spencer et al. 1992) and mortality (Standley et al. 1992) are two examples of studies that have been performed within Camp Roberts to further the goal of additional ecological research for kit fox in the USFWS recovery plan.

F7.1.9 Existing Conservation Actions in the Study Area

The INRMP and ESMP include the following current actions that take place on the Fort Hunter Liggett for San Joaquin kit fox (U.S. Army Garrison Fort Hunter Liggett 2012).

- Monitor predator indices of abundance in kit fox habitat biannually by means of night-time spotlighting and scent stations.
- If a kit fox is sighted within the past 12 months, conduct pre-activity surveys prior to ground-disturbing activities in the valley in which the sighting occurred.
- Conduct pre-activity surveys prior to poisoning of ground squirrels.
- Annually monitor artificial kit fox dens.
- Update GIS data for kit fox and red fox observations.
- Manage vegetation by implementing yellow-star thistle control and conducting prescribed burns.

Spotlight and scent station surveys have been conducted two or three times per year since 1998. Pre-activity surveys are regularly conducted prior to construction or use of rodenticide in potential habitat; however, no San Joaquin kit fox dens have been found (U.S. Army Garrison Fort Hunter Liggett 2012).

The INRMP also includes a list of future actions.

- Keep abreast of many factors affecting satellite populations of San Joaquin kit fox by attending local resource agency meetings and coordinating with USFWS, and adapt management and monitoring as needed to address new information (U.S. Army Garrison Fort Hunter Liggett 2012).

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Appendix F8
Monterey Spineflower Species Account

F8.1 Monterey Spineflower (*Chorizanthe pungens* var. *pungens*)

F8.1.1 Legal Status

F8.1.1.1 State

Monterey spineflower is not listed under the CESA. At the local level, Monterey spineflower is categorized as a California Native Plant Society Rank 1B.2 species.[1]

F8.1.1.2 Federal

Monterey spineflower was listed as federally threatened by USFWS in 1994 (59 FR 5499).

F8.1.1.3 Critical Habitat

Critical habitat for the Monterey spineflower was designated in 2002 (67 FR 37497) and 2008 (73 FR 1525) and is located within the northwestern portion of the study area.

F8.1.2 Taxonomy

The Monterey spineflower was originally described by George Bentham in 1836 based on a specimen collected by David Douglas from Monterey in 1833. It was originally classified as *Chorizanthe douglasii* var. *albens* by Charles Parry in 1889. Parry's classification was reassigned to *C. pungens* decades later, which was further reduced to two varieties: *C. pungens* var. *pungens and C. pungens* var. *hartwegiana*. *C. pungens* var. *pungens* differs from *C. p.* var. *hartwegiana* by having involucral lobe margins that are white rather than dark pink or purple (U.S. Fish and Wildlife Service 1998; Reveal and Rosatti 2013).

F8.1.3 Description

Monterey spineflower is an annual prostrate herb in the buckwheat family (*Polygonaceae*). It has linear, alternate leaves and the inflorescence is characterized by hooked involucre awns. Monterey spineflower blooms from April through July and can self-pollinate as well as outcross. It produces small seeds that are dropped or shaken by wind from their capsule and may then be dispersed with blowing sand or by fur-bearing animals to which the spiny fruits may attach and be carried. The species colonizes open sandy sites and tends to invade roadsides and firebreaks (U.S. Fish and Wildlife Service 1998).

[1] *1B* means rare, threatened, or endangered in California and elsewhere; *.2* means fairly endangered in California.

F8.1.4 Distribution

F8.1.4.1 State

Historical

The Monterey spineflower is known from the mountains of Santa Cruz County south to the coastline of Monterey and inland to the coastal plain of the Salinas Valley. Historically, this species occurred farther south near San Lucas in southern Monterey County and near San Simeon along the coast of northern San Luis Obispo County as indicated in early collections by Keck and Stockwell in 1935 and Gambel in 1842, respectively (California Native Plant Society 2018a, 2018b). Historical occurrences in the Salinas Valley have been extirpated primarily because of conversion of natural habitat to agricultural land.

Recent

The range of the Monterey spineflower is now limited to the interior of Santa Cruz County south along the coastal areas of the Monterey Peninsula, as well as the inland coastal plain of the Salinas Valley. There are 36 extant occurrences known from the region (California Department of Fish and Wildlife 2018). The northernmost population is known from the Santa Cruz Mountains between Scotts Valley and Felton, and the southernmost population is located on the south side of the Salinas River levee approximately 2.5 miles southeast of the town of Soledad. The largest population occurs on the Fort Ord on the coast of the Monterey Bay and the furthest inland population is located in Manzanita Park near Prunedale (U.S. Fish and Wildlife Service 1998; 73 FR 1525; California Department of Fish and Wildlife 2018; California Native Plant Society 2018a, 2018b).

F8.1.4.2 Study Area

Twenty eight occurrences of Monterey spineflower are known from within the study area, most of which have been documented in the last 40 years (California Department of Fish and Wildlife 2018; California Native Plant Society 2018b). They are found predominately along the coastline on protected lands of Salinas River State Beach, Salinas River National Wildlife Refuge, and Marina State Beach and on Fort Ord. Only 2 of the 28 occurrences are reported from the coastal plain of the Salinas Valley: one historic location sighted by Jepson in 1920 and annotated by Reveal in 1987 in the Arroyo Seco riverbed just upstream of the confluence with the Salinas River near Mission Soledad (California Department of Fish and Wildlife 2018); and another more recent occurrence along the Salinas River levee approximately 2.5 miles southeast of Soledad reported in 1994 and 2013 (California Department of Fish and Wildlife 2018; California Native Plant Society 2018b).

F8.1.5 Natural History

F8.1.5.1 Habitat Requirements

Monterey spineflower is found in maritime chaparral, coast live oak woodland, coastal scrub, grassland, and coastal dune habitats. It tends to colonize open sandy sites with little to no vegetative cover and has been found in firebreaks, along roadsides, and within sandy openings between shrubs. This species can tolerate some disturbance, such as scraping of roads and firebreaks, which can reduce the competition from other herbaceous species and consequently provide favorable conditions for Monterey spineflower. The largest population on Fort Ord is reported from sandy areas that were frequently disturbed by military training activities. Occurrences range in elevation from 7 to 2,300 feet. Species primarily associated with Monterey spineflower include beach-bur (*Ambrosia chamissonis*), coastal sagewort (*Atremisia pycnocephala*), and mock heather (*Ericameria ericioides*).

F8.1.5.2 Reproduction

Monterey spineflower plants produce one seed per flower successfully via self- or cross-pollination. In plants with high vigor, this can equate to dozens of seeds produced in one blooming season per plant. Plants typically germinate soon after winter rains, flowering occurs in the spring, and seed is set in the summer. Dispersal is either by wind or animal. Seeds are characterized by hooked spines that help facilitate successful attachment to local animals during the late summer and early fall (U.S. Fish and Wildlife Service 2009). A study regarding the seed bank of Monterey spineflower found that density of a population is directly related to the previous year's seed set (Fox et al. 2006), suggesting that the species does not create an extensive, reliable seed bank. However, there have been recent studies documenting Monterey spineflower reappearing after iceplant removal activities, indicating that a stable seed bank exists in some locations and can substantially repopulate a site after several years of dormancy (U.S. Fish and Wildlife Service 2009).

F8.1.5.3 Ecological Relationships

Potential breeding system studies on Monterey spineflower have yet to be conducted; however, a pollination ecology study was conducted on a closely related species: robust spineflower (*Chorizanthe robusta var. robusta*), a species from Santa Cruz County that is federally listed as endangered. The pollination study compared the pollination ecology of coastal and inland populations (U.S. Fish and Wildlife Service 2009) and found that although the species may self-pollinate, pollinator access to flowers significantly increased seed set. Because these two taxa occur in close proximity to each other at several locations (Sunset and Manresa State Beaches), inhabit similar niches, and are closely related (U.S. Fish and Wildlife Service 2009), the results of this study should be considered relevant for the recovery of Monterey spineflower. The study also found that a high diversity of potential pollinators, including sweat bees (Halictidae), bumblebees (*Bombus* sp.), wasps (Sphecidae), honeybees (*Apis mellifera*), and soft-winged flower beetles (Dasytidae), were reported to transport pollen efficiently. Pollinator diversity also correlated with variation in habitat conditions, including slope; proximity to the coast; and the structure, composition, and density of the surrounding vegetation (U.S. Fish and Wildlife Service 2009). These results indicated that pollinator diversity and habitat protection is important to the recovery of the *Chorizanthe* taxa.

F8.1.6 Population Status and Trends

F8.1.6.1 Population Trend

State

As an annual species, Monterey spineflower is highly dependent upon annual precipitation, resulting in large fluctuations across populations each year. At the time of listing in 1994, Monterey spineflower was known from a dozen scattered populations along the Monterey Peninsula, at Manzanita Park near Prunedale, in the coastal terraces of Fort Ord, and from historical collections described from Watsonville and Soledad in the Salinas Valley. Since its listing, additional populations have been discovered in the Prunedale Hills of Monterey County and interior areas of Santa Cruz County. Many populations support large numbers of individuals (thousands or tens of thousands of plants) scattered in openings among the dominant perennial vegetation (California Department of Fish and Wildlife 2018).

Study Area

All 28 occurrences within the study area are presumed to be extant (California Department of Fish and Wildlife 2018) except for one occurrence in the coastal plain of the Salinas Valley observed by Jepson in 1920. The majority of occurrences are on protected lands managed by California State Parks or the Fort Ord Reuse Authority, therefore declines in populations and contraction of the range are not anticipated. Historically, this species occurred farther south near San Lucas in southern Monterey County and near San Simeon along the coast of northern San Luis Obispo County, as indicated in early collections by Keck and Stockwell in 1935 and Gambel in 1842.

F8.1.7 Threats

At the time of listing, several threats to Monterey spineflower habitat were identified: industrial and residential development, recreational use including horseback riding, and road improvements. Urban development in coastal cities, and to a lesser extent in the study area, has resulted in the loss of large portions of the range of Monterey spineflower. In addition, introduction of invasive African ice plant (*Carpobrotus edulis*) and European beach grass (*Ammophila arenaria*) for dune stabilization has altered typical Monterey spineflower habitat and made conditions unsuitable for the species. In the USFWS 5-year review for the species, newly identified threats to the species include climate change and sea level rise; however, the extent of these threats is unknown (U.S. Fish and Wildlife Service 2009).

F8.1.8 Recovery Planning

F8.1.8.1 State

According to the *Seven Coastal Plants and the Myrtle's Silverspot Butterfly Recovery Plan* (U.S. Fish and Wildlife Service 1998), the strategy for recovery of the Monterey Spineflower will involve (1) habitat restoration via eradication of invasive plant species; (2) protection of occupied habitat; (3) species reintroduction to historical range, and (4) long term monitoring and management.

F8.1.8.2 Study Area

Long term restoration and monitoring for Monterey spineflower has been ongoing within the study area on Marina State Beach, Sunset State Beach, and Asilomar State Park lands, which are protected in perpetuity (U.S. Fish and Wildlife Service 1998). In addition, populations at Fort Ord are also being protected and monitored by several agencies including USFWS (2009) and U.S. Army Corps of Engineers as part of the *Installation-Wide Multispecies Habitat Management Plan for Former Fort Ord, California* (U.S. Army Corps of Engineers 1997, 2013).

F8.1.9 Existing Conservation Actions in the Study Area

Currently Marina State Beach, Sunset State Beach, and Asilomar State Park are removing invasive plant species, planting native species, protecting known occurrences of the species, and conducting annual monitoring activities per the recovery plan (U.S. Fish and Wildlife Service 1998). Habitat reserve areas established on Fort Ord that contain Monterey spineflower are also being monitored and managed as part of the habitat management plan established for the base closure (U.S. Army Corps of Engineers 1997). In addition, the FOHCP is expected to be completed by 2020, and activities associated with this plan will protect and monitor the known occurrences of Monterey spineflower at Fort Ord.

Efforts to control invasive giant reed and tamarisk and restore riparian habitat in the Salinas River watershed are underway in Monterey County, and major efforts are administered by the Resource Conservation District of Monterey County and implemented through partnerships with state and local agencies as well and private land-owners and non-profit organizations (Resource Conservation District of Monterey County 2018). These efforts will also benefit the two known occurrences of Monterey spineflower in the watershed as well as restore historical localities where the species once occurred and may be reintroduced as suggested by USFWS (2009) in the 5-year review.

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Appendix F9
Sand Gilia Species Account

F9.1 Sand Gilia (*Gilia tenuiflora* ssp. *arenaria*)

F9.1.1 Legal Status

F9.1.1.1 State

Sand gilia is listed as threatened under the CESA. At the local level, sand gilia is categorized as a California Native Plant Society Rank 1B.2 species.[1]

F9.1.1.2 Federal

Sand gilia was listed as federally endangered in 1992 by the USFWS (57 FR 27848).

F9.1.1.3 Critical Habitat

No critical habitat has been designated for sand gilia.

F9.1.2 Taxonomy

The sand gilia (also commonly referred to as Monterey gilia) was originally described by George Bentham in 1833 based on a specimen collected by David Douglas from Monterey in the early 1800s. In 1943, Willis Linn Jepson reduced it to a variety of *G. tenuiflora* based on corolla diameter width and color. In 1956, Verne Grant and Alva Day Grant further reduced the taxon to a subspecies based on the taxon’s slightly exerted stamens and distinctly large fruit capsules (U.S. Fish and Wildlife Service 1998; Porter 2013).

F9.1.3 Description

Sand gilia is an annual herb in the phlox family (*Polemoniaceae*) less than 7 inches tall. It is characterized by a central erect stem supported by a basal rosette of leaves with several other stems spreading out from the base densely covered with glandular hairs, giving the plant a cobwebby appearance near the base. The taxon has white and purple funnel-shaped flowers with narrow petal lobes. Sand gilia typically germinates from December to February and blooms from April to June. It is able to self-pollinate as well as outcross, and fruit is set by the end of May (U.S. Fish and Wildlife Service 1998). It produces small seeds that are dropped or shaken from their capsules and are then dispersed, likely by gravity or wind. The plant occurs along trails and roadsides, on the cut banks of sandy ephemeral drainages, in recently burned chaparral, and in other disturbed patches. It appears to do well on sites that have undergone recent substrate disturbance. Most populations are small and localized (U.S. Fish and Wildlife Service 2008). Many of the populations of sand gilia found in the study area support individuals with characteristics intermediate with sand gilia and the related

[1] 1B means rare, threatened, or endangered in California and elsewhere; .2 means fairly endangered in California.

subspecies slender-flowered gilia (*Gilia tenuiflora* ssp. *tenuiflora*) (Dorrell-Canepa 1994). Slender-flowered gilia is an inland subspecies known to occur in the study area in sandy washes of woodlands in the Salinas Valley. It is possible that the study area is a zone of intergradation between these two subspecies.

F9.1.4 Distribution

F9.1.4.1 State

Historical

Sand gilia is historically known from the Monterey Peninsula dune complexes. Five historical occurrences were distributed in discontinuous coastal populations from the dunes north of Marina to Monterey. Before 1990, this species was presumed to be extinct; historical occurrences were extirpated primarily because of conversion of natural habitat to development, agriculture, or military uses. Two inland locations within the Salinas Valley and Santa Lucia Mountains are also known from historical collections but are likely extirpated as well (California Native Plant Society 2018a, 2018b; Consortium of California Herbaria 2018).

Recent

Currently, sand gilia is known from 27 extant occurrences in a relatively small geographic range spanning approximately 22 miles of the California coast from Sunset Beach State Park in Santa Cruz County to Pacific Grove in Monterey County. Of these coastal populations, some are scattered inland throughout Fort Ord (California Department of Fish and Wildlife 2018). Approximately half of the potentially extant coastal occurrences occur on State, federal, and local agency protected lands with the remaining half on private lands (U.S. Fish and Wildlife Service 2008).

F9.1.4.2 Study Area

All but six of the statewide sand gilia occurrences are known from the study area, most of which have been documented in the last 50 years (California Department of Fish and Wildlife 2018; California Native Plant Society 2018b). The 21 extant occurrences are predominately found along the coastline or on Fort Ord. None of the occurrences are reported from the Salinas Valley.

F9.1.5 Natural History

F9.1.5.1 Habitat Requirements

Sand gilia is found on rear dunes, near the dune summit in level areas, and on depressions or slopes in wind-sheltered openings in low-growing dune scrub vegetation. On ancient dune soils, which extend inland between 6 and 8 miles from the coast, it occurs in openings among maritime chaparral, coastal sage scrub, oak woodlands, grasslands, and where other vegetative cover is low. At least half of the species' range occurs in the study area, where extensive suitable habitat is found in dunes of Salinas River State Beach, Salinas River National Wildlife Refuge, Marina State Beach as well as in maritime chaparral of Fort Ord. Occurrences range in elevation from 0 to 100 feet. Species associated with sand gilia include dune bush lupine (*Lupinus chamissonis*), coastal buckwheat (*Eriogonum latifolium*), beach-bur, coastal sagewort, and mock heather.

F9.1.5.2 Reproduction

Sand gilia plants produce three to many seeds per flower successfully primarily by self-pollination. Plants typically germinate soon after winter rains, flowering occurs in the early spring, and seed is set in the late spring or early summer. Dispersal is by wind (Dorrell-Canepa 1994). A study conducted in the early 2000s showed that sand gilia have long-lived seeds, which contribute to the taxon's persistent soil seed bank (Fox et al. 2006). After a large burn on Fort Ord, the subspecies emerged even in areas where it had not been observed pre-burn (U.S. Fish and Wildlife Service 2008). Furthermore, where burns occurred in occupied habitat, density of individuals increased and overall plant size was larger for at least 2 years following the prescribed burn event. This study indicated that sand gilia can tolerate variable climatic conditions and that successful germination events may only occur in years with very specific conditions.

F9.1.5.3 Ecological Relationships

Sand gilia requires semi-open areas of sandy soil to germinate and to thrive. The taxon is generally found in sparse dune scrub and maritime chaparral communities and does not compete well in denser vegetation communities (Dorrell-Canepa 1994). Most populations of sand gilia either seem to have a high cover of invasive plants already established or are being encroached upon. Due to the presence of invasive species throughout the geographic range of the subspecies, invasive plant species management will be required in long-term planning for recovery (U.S. Fish and Wildlife Service 2008).

F9.1.6 Population Status and Trends

F9.1.6.1 Population Trend

State

Sand gilia is an annual species and like most annual taxa, it can go through large changes in number of individuals from year to year depending on rainfall amounts and timing of events (U.S. Army Corps of Engineers 2013). For instance, rainfall events later in the growing season can significantly affect population trends observed in sand gilia (Dorrell-Canepa 1994). Surveys at Marina State Beach have reported fluctuations from a low of 5,000 individuals in 1987 to a high of 25,000 individuals in 1993; the number of individuals at Salinas River State Beach has fluctuated from a low of 1,665 individuals in 1987 to a high of 13,500 individuals in 1993 (U.S. Fish and Wildlife Service 2008). A more recent monitoring effort on Fort Ord was conducted between 2010 and 2013 with fluctuations in number of individuals markedly related to precipitation amounts and timing. Average population numbers ranged from 10 to 50 individuals in drought years with up to 1,000 plants in reference plots during an average rainfall year (U.S. Army Corps of Engineers 2013). With increasing pressures and stressors on existing populations, long term monitoring and management is necessary to generate data useful in determining future sand gilia trends or causalities in the remaining populations.

Study Area

All 21 occurrences of sand gilia within the study area are presumed to be extant (California Department of Fish and Wildlife 2018). The majority of occurrences are on protected lands managed by California State Parks, USFWS, or the Fort Ord Reuse Authority, therefore declines in populations and contraction of the range are not anticipated.

F9.1.7 Threats

At the time of listing, several threats to sand gilia habitat were identified: industrial and residential development, recreational use including horseback riding, introduction of invasive species, road improvements, and herbivory. Urban development in coastal cities, and to a lesser extent in the study area, has resulted in the loss of large portions of the range of sand gilia (Dorrell-Canepa 1994). Introduction of invasive African ice plant, jubata grass (*Cortaderia jubata*) and ripgut brome (*Bromus diandrus*) has altered typical sand gilia habitat and made conditions unsuitable for the species. In the 5 year review for the sand gilia, newly identified threats to the species included vegetation management on Fort Ord, including poorly timed prescribed fires, pre-fire treatments that may introduce invasive species into sand gilia habitat, and mechanical vegetation removal in which chipped vegetation is left behind on the soil surface (U.S. Fish and Wildlife Service 2008). Vegetation management activities on Fort Ord are being addressed through the habitat conservation planning process that will guide future management of the inland sites of sand gilia.

F9.1.8 Recovery Planning

F9.1.8.1 State

According to the *Seven Coastal Plants and the Myrtle's Silverspot Butterfly Recovery Plan* (U.S. Fish and Wildlife Service 1998), the strategy for recovery of the sand gilia will involve (1) habitat restoration via eradication of invasive plant species, (2) protection of occupied habitat, (3) species reintroduction to historical range, and (4) long term monitoring and management (U.S. Fish and Wildlife Service 1998, 2008).

F9.1.8.2 Study Area

Long-term restoration and monitoring for sand gilia has been established within the study area on Sunset State Beach, Salinas River State Beach, Marina State Beach, and Asilomar State Park lands, which are protected in perpetuity (U.S. Fish and Wildlife Service 1998). In addition, populations at Fort Ord are also being protected and monitored by several agencies including USFWS (2008) and U.S. Army Corps of Engineers as part of the *Installation-Wide Multispecies Habitat Management Plan for Former Fort Ord, California* (U.S. Army Corps of Engineers 1997, 2013). Transplantation attempts for mitigation at the Spanish Bay Golf Course failed; however, artificial augmentation seeding has been attempted at other sites and has been successful (U.S. Fish and Wildlife Service 2008).

F9.1.9 Existing Conservation Actions in the Study Area

Currently Sunset State Beach, Salinas River State Beach, Marina State Beach, and Asilomar State Park are removing invasive plant species, planting native species, protecting known occurrences of the species, and conducting annual monitoring activities per the recovery plan (U.S. Fish and Wildlife Service 1998). Habitat reserve areas established on Fort Ord that contain sand gilia are also being monitored and managed as part of the habitat management plan established for the base closure (U.S. Army Corps of Engineers 1997). In addition, the FOHCP is expected to be complete by 2020, and activities associated with this plan will also protect and monitor the known occurrences of sand gilia on Fort Ord.

F9.1.10 Literature Cited

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