APPENDIX C

Status of the Species & Environmental Baseline for LAA Species and CH

This Appendix describes the range-wide status of the species and environmental baseline for all the Covered Species and any associated critical habitat in the PBO (see Table 1). We describe factors, such as life history, distribution, and population size and trends, which help determine the likelihood of both survival and recovery of the species. For a majority of the Covered Species and Critical Habitat, the Status of the Species and Environmental Baseline are the same due to the species occurring within only California.

The information in this Appendix provides additional information used for the Jeopardy and Adverse Modification analyses in the PBO.

Table 1: Species and CH Analyzed in PBO

Species Common Name	Species Latin Name	ESA Status	Critical Habitat
Amphibians			
arroyo toad	*Anaxyrus californicus*	E	Yes
California red-legged frog	*Rana draytonii*	T	Yes
California tiger salamander – Central California DPS	*Ambystoma californiense*	T	Yes
California tiger salamander – Santa Barbara County DPS	*Ambystoma californiense*	E	Yes
foothill yellow-legged frog – Central Coast DPS	*Rana boylii*	T	PCH
foothill yellow-legged frog – North Feather DPS	*Rana boylii*	T	PCH
foothill yellow-legged frog – South Coast DPS	*Rana boylii*	E	PCH
foothill yellow-legged frog – Southern Sierra DPS	*Rana boylii*	E	PCH
mountain yellow-legged frog – northern California DPS	*Rana muscosa*	E	Yes
Santa Cruz long-toed salamander	*Ambystoma macrodactylum croceum*	E	N/A
Sierra Nevada yellow-legged frog	*Rana sierrae*	E	Yes
western spadefoot – Northern DPS	*Spea hammondii*	PT	N/A
western spadefoot – Southern DPS	*Spea hammondii*	PT	N/A
Yosemite toad	*Anaxyrus canorus*	T	Yes
Reptiles			
Alameda whipsnake	*Masticophis lateralis euryxanthus*	T	Yes
giant garter snake	*Thamnophis gigas*	T	N/A
northwestern pond turtle	*Actinemys marmorata*	PT	N/A

San Francisco garter snake	*Thamnophis sirtalis tetrataenia*	E	N/A
southwestern pond turtle	*Actinemys pallida*	PT	N/A
Birds			
California least tern	*Sterna antillarum browni*	E	N/A
California Ridgway's rail	*Rallus obsoletus obsoletus*	E	N/A
California spotted owl – Coastal-Southern California DPS	*Strix occidentalis occidentalis*	PE	N/A
California spotted owl – Sierra Nevada DPS	*Strix occidentalis occidentalis*	PT	N/A
coastal California gnatcatcher	*Polioptila californica*	T	Yes
least Bell's vireo	*Vireo bellii pusillus*	E	Yes
light-footed Ridgway's rail	*Rallus obsoletus levipes*	E	N/A
marbled murrelet	*Brachyramphus marmoratus*	T	Yes
northern spotted owl	*Strix occidentalis caurina*	T	Yes
western snowy plover – Pacific Coast DPS	*Anarhynchus nivosus* ssp. *nivosus*	T	Yes
Mammals			
riparian woodrat	*Neotoma fuscipes riparia*	E	N/A
riparian brush rabbit	*Sylvilagus bachmani riparius*	E	N/A
salt marsh harvest mouse	*Reithrodontomys raviventris*	E	N/A
San Bernardino kangaroo rat (*Critical Habitat Only*)	*Dipodomys merriami parvus*	E	Yes
Invertebrates			
California freshwater shrimp	*Syncaris pacifica*	E	N/A
conservancy fairy shrimp	*Branchinecta conservatio*	E	Yes
longhorn fairy shrimp	*Branchinecta longiantenna*	E	Yes
Mount Hermon June beetle	*Polyphylla barbata*	E	N/A
Riverside fairy shrimp	*Streptocephalus woottoni*	E	Yes
San Diego fairy shrimp	*Branchinecta sandiegonensis*	E	Yes
Smith's blue butterfly	*Euphilotes enoptes smithi*	E	N/A
valley elderberry longhorn beetle	*Desmocerus californicus dimorphus*	T	Yes
vernal pool fairy shrimp	*Branchinecta lynchi*	T	Yes
vernal pool tadpole shrimp	*Lepidurus packardi*	E	Yes
Fish			
Delta smelt	*Hypomesus transpacificus*	T	Yes
Lahontan cutthroat trout	*Oncorhynchus clarkii henshawi*	T	N/A
longfin smelt – San Francisco Bay-Delta DPS	*Spirinchus thaleichthys*	E	PCH
tidewater goby	*Eucyclogobius newberryi*	E	Yes
unarmored threespine stickleback	*Gasterosteus aculeatus williamsoni*	E	N/A
Non-vernal pool Plant Species			
Ben Lomond spineflower	*Chorizanthe pungens* var. *hartwegiana*	E	N/A
California seablite	*Suaeda californica*	E	N/A

La Graciosa thistle	*Cirsium loncholepis*	E	Yes
marsh sandwort	*Arenaria paludicola*	E	N/A
salt marsh bird's-beak	*Chloropyron maritimum* subsp. *maritimum*	E	N/A
Ventura marsh milk-vetch	*Astragalus pycnostachyus* var. *lanosissimus*	E	Yes
Vernal Pool Plant Species			
Butte County meadowfoam	*Limnanthes floccosa* ssp. *californica*	E	Yes
California Orcutt grass	*Orcuttia californica*	E	N/A
Contra Costa goldfields	*Lasthenia conjugens*	E	Yes
few-flowered navarretia	*Navarretia leucocephala* ssp. *pauciflora*	E	N/A
fleshy owl's-clover	*Castilleja campestris* ssp. *succulenta*	T	Yes
hairy Orcutt grass	*Orcuttia pilosa*	E	Yes
Hoover's spurge	*Chamaesyce hooveri*	T	Yes
Otay Mesa-mint	*Pogogyne nudiuscula*	E	N/A
Sacramento Orcutt grass	*Orcuttia viscida*	E	Yes
San Diego ambrosia	*Ambrosia pumila*	E	Yes
San Diego button-celery	*Eryngium aristulatum* var. *parishii*	E	N/A
San Joaquin Orcutt grass	*Orcuttia inaequalis*	T	Yes
slender Orcutt grass	*Orcuttia tenuis*	T	Yes
spreading navarretia	*Navarretia fossalis*	T	Yes
thread-leaved brodiaea	*Brodiaea filifolia*	T	Yes

E = Federally Endangered under the ESA T = Federally Threatened under the ESA
PE = Proposed Endangered under the ESA PT = Proposed Threatened under the ESA PCH = Proposed Critical Habitat under the ESA

Amphibians

Arroyo Toad [*Anaxyrus californicus* (*Bufo microscaphus c.*)] and its Critical Habitat

Listing Status

The Service federally listed the arroyo toad as endangered on December 16, 1994 (Service 1994), On February 9, 2011, the Service designated approximately 98,366 acres of critical habitat for the arroyo toad (Service 2011). At the time of listing, the primary threats to arroyo toads were urban development, agricultural conversion, operations of dams and water flow, roads and road maintenance, recreational activities, introduced predators, and droughts.

Life History and Habitat

The arroyo toad is a small, light-olive green or gray to tan toad with dark spots and warty skin. Arroyo toads are terrestrial for much of the year and can range widely into upland habitat for foraging and burrowing, but use aquatic habitat for breeding. Breeding occurs in shallow, slow-moving stream systems and may occur from January to July. Breeding tends to occur earlier in coastal areas than inland areas (Service 1999).

Population Status

Thirty-five populations of arroyo toad are distributed from Monterey County, California, in the United States south to Baja California, Mexico (Service 2015). Urbanization, agriculture, and dams are the main reasons for the decline of arroyo toad and are also current threats. Other threats include water management activities and diversions; road construction, maintenance, and use; grazing; mining; recreation; and nonnative plants and animals (Service 1999). Decline in number of populations of arroyo toads has already occurred (Jennings and Hayes 1994, p. 57 in Service 2015), and new data indicate that the species has continued to decline in numbers and in area occupied within its current range (Hancock 2007–2014, entire; Hollingsworth in litt. 2014; USGS in litt. 2014; Sweet 2015, pers. comm.; USGS 2015, pers. comm., all In Service 2015).

Critical Habitat

This critical habitat occurs in 21 units within Santa Barbara, Ventura, Los Angeles, San Bernardino, Riverside, Orange, and San Diego counties, California. The physical and biological features of designated critical habitat for the arroyo toad are:

1. Rivers or streams with hydrologic regimes that supply water to provide space, food, and cover needed to sustain eggs, tadpoles, metamorphosing juveniles, and adult breeding arroyo toads. Breeding pools must persist for a minimum of 2 months for the completion of larval development. However, due to the dynamic nature of southern California riparian systems and flood regimes, the location of suitable breeding pools may vary from year to year. Specifically, the conditions necessary to allow for successful reproduction of arroyo toads are: (a) breeding pools that are less than 6 inches deep; (b) areas of flowing water with current velocities less than 1.3 feet per second; and (c) surface water that lasts for a minimum of 2 months during the breeding season (a sufficient wet period in the spring months to allow arroyo toad larvae to hatch, mature, and metamorphose).
2. Riparian and adjacent upland habitats, particularly low-gradient (typically less than 6 percent) stream segments and alluvial streamside terraces with sandy or fine gravel substrates that support the formation of shallow pools and sparsely vegetated sand and gravel bars for breeding and rearing of tadpoles and juveniles; and adjacent valley bottomlands that include areas of loose soil where arroyo toads can burrow underground, to provide foraging and living areas for juvenile and adult arroyo toads.

3. A natural flooding regime, or one sufficiently corresponding to natural, that: (a) is characterized by intermittent or near-perennial flow that contributes to the persistence of shallow pools into at least mid-summer; (b) maintains areas of open, sparsely vegetated, sandy stream channels and terraces by periodically scouring riparian vegetation; and (c) also modifies stream channels and terraces and redistributes sand and sediment, such that breeding pools and terrace habitats with scattered vegetation are maintained.
4. Stream channels and adjacent upland habitats that allow for movement to breeding pools, foraging areas, overwintering sites, upstream and downstream dispersal, and connectivity to areas that contain suitable habitat.

Recovery Plan Information

A recovery plan for the species was published in 1999 (Service 1999). The recovery strategy for the arroyo toad is focused on providing sufficient breeding and upland habitat to maintain self-sustaining populations of arroyo toads throughout the historic range of the species in California, and minimizing or eliminating impacts and threats to arroyo toad populations. The recovery strategy for the arroyo toad consists of five parts: 1) stabilize and maintain populations throughout the range of the arroyo toad in California by protecting sufficient breeding and nonbreeding habitat, 2) monitor the status of existing populations to ensure recovery actions are successful, 3) identify and secure, by appropriate management and monitoring, additional suitable arroyo toad habitat and populations, 4) conduct research to determine the population dynamics and ecology of the species to guide management efforts and determine the best methods for reducing threats, and 5) develop and implement an outreach program.

Environmental Baseline

The species only occurs within the State of California, extending into Baja California, Mexico. Please refer to the above information regarding the species environmental baseline.

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California Red-legged Frog (*Rana draytonii*) and its Critical Habitat

Listing Status

The California red-legged frog was listed as a threatened species on May 23, 1996 (USFWS 1996). Critical habitat was designated for this species on April 13, 2006 (USFWS 2006), with revisions to the critical habitat designation published on March 17, 2010 (USFWS 2010). At that time, the Service recognized the taxonomic change from *Rana aurora draytonii* to *Rana draytonii* (Shaffer et al. 2010).

Life History and Habitat

Habitat

The California red-legged frog generally breeds in still or slow-moving water associated with emergent vegetation, such as cattails, tules (hardstem bulrush), or overhanging willows (Storer 1925; Fellers 2005). Aquatic breeding habitat predominantly includes permanent water sources such as streams, marshes, and natural and manmade ponds in valley bottoms and foothills (Jennings and Hayes 1994; Bulger et al. 2003; Stebbins 2003). Since the 1850's, manmade ponds may actually supplement stream pool breeding habitat and can be capable of supporting large populations of this species. Breeding sites may hold water only seasonally, but sufficient water must persist at the beginning of the breeding season and into late summer or early fall for tadpoles to successfully complete metamorphosis. Breeding habitat does not include deep lacustrine water habitat (e.g., deep lakes and reservoirs 50 acres or larger in size) (USFWS 2010). Within the coastal lagoon habitats, salinity is a significant factor on embryonic mortality or abnormalities (Jennings and Hayes 1990). Jennings and Hayes (1990) conducted laboratory studies and field observations concluding salinity levels above 4.5 parts per thousand detrimentally affected the California red-legged frog embryos. Aquatic breeding habitat does not need to be available every year, but it must be available at least once within the frog's lifespan for breeding to occur (USFWS 2010).

Non-breeding aquatic habitat consists of shallow (non-lacustrine) freshwater features not suitable as breeding habitat, such as seasonal streams, small seeps, springs, and ponds that dry too quickly to support breeding. Non-breeding aquatic and riparian habitat is essential for providing the space, food, and cover necessary to sustain the California red-legged frog. Riparian habitat consists of vegetation growing nearby, but not typically in, a body of water on which it depends, and usually extends from the bank of a pond or stream to the margins of the associated floodplain (USFWS 2010). Adult California red-legged frogs may avoid coastal habitat with salinity levels greater than 6.5 parts per thousand (Jennings and Hayes 1990).

Cover and refugia are important habitat characteristic preferences for the species (Halstead and Kleeman 2017). Refugia may include vegetation, organic debris, animal burrows, boulders, rocks, logjams, industrial debris, or any other object that provides cover. Agricultural features such as watering troughs, spring boxes, abandoned sheds, or haystacks may also be utilized by the species. Incised stream channels with portions narrower and depths greater than 18 inches may also provide important summer sheltering habitat. During periods of high water flow, California red-legged frogs are rarely observed; individuals may seek refuge from high flows in pockets or small mammal burrows beneath banks stabilized by shrubby riparian growth (Jennings and Hayes 1994). Accessibility to cover habitat is essential for the survival of California red-legged frogs within a watershed and can be a factor limiting frog population numbers and survival.

Breeding

In the Coast Range and at lower elevations, the California red-legged frog typically breeds between November and April (Storer 1925; Jennings and Hayes 1994; Fellers 2005). However, breeding phenology varies by location and across years, largely based on differences in climatic conditions (McHarry et al. 2019). At sites that routinely experience winter temperatures below freezing, the beginning of breeding is generally corresponded with the onset of spring's warmer air temperatures, such as in the Sierra Nevada where breeding typically occurs in late February and March (McHarry et al. 2019). Dependent on weather conditions, breeding in the Sierra Nevada can occur into late April (Barry 2002).

Females deposit their egg masses on emergent vegetation, floating on or near the surface of the water. The California red-legged frog is often a prolific breeder, laying eggs during or shortly after large rainfall events. Egg masses containing 300-4,000 eggs hatch after six to fourteen days (Storer 1925; Jennings and Hayes 1994; Fellers 2005). Historically, the California red-legged frog in the Sierra Nevada likely bred within stream pools, which tend to be small with limited forage, constraining the size and number of populations (Barry and Fellers 2013).

California red-legged frog tadpoles undergo metamorphosis three to seven months following hatching. Most males reach sexual maturity in two years, while it takes approximately three years for females (Jennings and Hayes 1985; Fellers 2005). Under favorable conditions, California red-legged frogs may live eight to ten years (Jennings et al. 1992). Of the various life stages, tadpoles likely experience the highest mortality rates; only one percent of each egg mass completes metamorphosis (Jennings et al. 1992).

Diet

The California red-legged frog has a variable diet that changes with each of its life history stages. The feeding habits of the early stages are likely similar to other ranids, whose tadpoles feed on algae, diatoms, and detritus by grazing on the surface of rocks and vegetation (Fellers 2005). Hayes and Tennant (1985) found invertebrates to be the most common food items of adult California red-legged frogs collected in southern California; however, they speculated that this was opportunistic and varied based on prey availability. Vertebrates, such as Pacific tree frogs (*Pseudacris regilla*) and California mice (*Peromyscus californicus*), represented over half of the prey mass eaten by larger frogs, although invertebrates were the most numerous food items. Feeding typically occurs along the shoreline and on the surface of the water; juveniles appear to forage during both daytime and nighttime, whereas adults appear to feed at night (Hayes and Tennant 1985).

Movement

California red-legged frogs do not have a distinct breeding migration (Fellers 2005), rather they may move seasonally from non-breeding pools or refugia to breeding pools. Some individuals remain at breeding sites year-round while others disperse to neighboring water features or moist upland sites when breeding is complete and/or when breeding pools dry (USFWS 2002; Bulger et al. 2003; Fellers and Kleeman 2007; Tatarian and Tatarian 2008; Tatarian 2008). Studies in the several San Francisco Bay counties showed movements are typically along riparian corridors (Fellers and Kleeman 2007; Tatarian 2008). Although, some individuals, especially on rainy nights and in more mesic areas, travel without apparent regard to topography, vegetation type, or riparian corridors, and can move directly from one site to another through normally inhospitable habitats such as heavily grazed pastures or oak-grassland savannas (Bulger et al 2003).

California red-legged frogs show high site fidelity (Tatarian and Tatarian 2008) and typically do not move significant distances from breeding sites (Bulger et al. 2003; Fellers and Kleeman 2007; Tatarian and Tatarian 2008; Tatarian 2008). When traveling between aquatic sites, California red-legged frogs typically travel less than 0.31 miles (Fellers and Kleeman 2007; Tatarian and Tatarian 2008), although they have been documented to move more than two miles in Santa Cruz County (Bulger et al. 2003). Various studies have found that the frogs typically do not make terrestrial forays further than 200 feet from aquatic habitat (Bulger et al. 2003; Fellers and Kleeman 2007; Tatarian and Tatarian 2008; Tatarian 2008). Upland movements are typically associated with precipitation events and usually last for one to four days (Tatarian 2008).

Population Status

Rangewide Status of the Species

The historical range of the California red-legged frog extended from central Mendocino County and western Tehama County south in the California Coast Range to northern Baja California, Mexico, and in the Sierra Nevada/Cascade Ranges from Shasta County south to Madera County (Jennings and Hayes 1994). The species historically occurred from sea level to elevations of about 5,200 feet in 46 counties; however, currently the taxon is extant in 238 streams or drainages within only 22 counties, representing a loss of 70 percent of its former range (USFWS 2002). Isolated populations persist in several Sierra Nevada foothill locales and in Riverside County (Barry and Fellers 2013; Backlin et al. 2017; CDFW 2017; Gordon, R. and J. Bennett, pers. comm., 2017). The species is no longer considered extant in California's Central Valley due to significant declines caused by habitat modifications and exotic species (Fisher and Shaffer 1996). Currently, the California red-legged frog is widespread in the San Francisco Bay nine-county area (CDFW 2017). They are still locally abundant within the California coastal counties from Mendocino County to Los Angeles County and presumed extirpated in Orange and San Diego counties (CDFW 2017; Yang, D. and J. Martin, pers. comm., 2017; Gordon, R. and J. Bennett, pers. comm., 2017). Baja California represents the southernmost edge of the species' current range (Peralta-García et al. 2016).

Barry and Fellers (2013) conducted a comprehensive study to determine the current range of the California red-legged frog in the Sierra Nevada, concluding that it differs little from its historical range; however, the current Sierra Nevada populations appear to be small and tend to fluctuate. Since 1991, eleven California red-legged frog populations have been discovered or confirmed, including eight probable breeding populations (Barry and Fellers 2013; Mabe, J., pers. comm., 2017). Microsatellite and mitochondrial DNA analysis by Richmond et al. (2014) confirmed the Sierra Nevada populations of the California red-legged frog are genetically distinct from each other, as well as from other populations throughout the range of this species. The research concluded that the Sierra Nevada populations are persisting at low levels of genetic diversity and no contemporary gene flow across populations exist. On a larger geographic scale, range contraction has left a substantial gap between Sierra Nevada and Coast Range populations, similar to the gap separating the Southern California and Baja California populations (Richmond et al. 2014).

Population Summary

Number of distinct occurrences (subpopulations) is unknown but probably is at least several dozen. According to USFWS (2000), the species occurs in about 238 streams or drainages. In the mid-1990s, most of the occupied habitat was in Monterey, San Luis Obispo, and Santa Barbara counties; the species occurred in only 5 sites south of the Tehachapi Mountains (80+ historic sites) (USFWS 1996). Aggregations including more than 350 adults were known only from Pescadero Marsh Natural Preserve in coastal San Mateo County, Point Reyes National Seashore in Marin County, and Rancho San Carlos in Monterey County (USFWS 1996). More than 120 breeding sites exist in Marin County (Fellers 2005). In California, south of Los Angeles, a single population is known from the Santa Rosa Plateau in Riverside County (Shaffer et al. 2004). Only two populations are known to exist south of Santa Barbara

(Fellers 2005). In the Sierra Nevada, *Rana draytonii* is now represented by only about a half dozen populations, only one of which is known to have more than 10 breeding adults (Shaffer et al. 2004).

Over the long term, extent of occurrence, area of occupancy, number of subpopulations, and population size have undergone a major decline. The species has been extirpated from much of its former range in California (Hayes and Jennings 1988, Shaffer et al. 2004). Range has been reduced by 70% (USFWS 1996, USFWS 2000). Total adult population size is unknown but undoubtedly exceeds 10,000. The species is still locally abundant in portions of the San Francisco Bay area and the central coast (USFWS 2000). Breeding sites in Marin County include several thousand adults (Fellers 2005).

Threats

Factors associated with declining populations of the California red-legged frog throughout its range include degradation and loss of habitat through agriculture, urbanization, mining, overgrazing, recreation, timber harvesting, non-native species, impoundments, water diversions, erosion and siltation altering upland and aquatic habitat, degraded water quality, use of pesticides, and introduced predators (USFWS 2002, USFWS 2010). Urbanization often leaves isolated habitat fragments and creates barriers to frog dispersal.

Non-native species pose a major threat to the recovery of California red-legged frogs. Several researchers have noted the decline and eventual local disappearance of California and northern red-legged frogs in systems supporting bullfrogs (Jennings and Hayes 1990; Twedt 1993), red swamp crayfish, signal crayfish, and several species of warm water fish including sunfish, goldfish, common carp, and mosquitofish (Moyle 1976; Barry 1992; Hunt 1993; Fisher and Shaffer 1996). The decline of the California red-legged frog due to these non-native species has been attributed to predation, competition, and reproduction interference (Twedt 1993; Bury and Whelan 1984; Storer 1933; Emlen 1977; Kruse and Francis 1977; Jennings and Hays 1990; Jennings 1993).

Chytridiomycosis, an infectious disease caused by the chytrid fungus, *Batrachochytrium dendrobatidis* (*Bd*), has been found to adversely affect amphibians globally (Davidson et al. 2003; Lips et al. 2006). While *Bd* prevalence in wild amphibian populations in California is unknown (Fellers et al. 2011), chytrid is expected to be widespread throughout much of the California red-legged frog's range. The chytrid fungus has been documented within the California red-legged frog populations at Point Reyes National Seashore, two properties in Santa Clara County, Yosemite National Park, Hughes Pond, Sailor Flat, Big Gun Diggings, and Spivey Pond (Padgett-Flohr and Hopkins 2010; Tatarian and Tatarian 2010; Fellers et al. 2011; Barry and Fellers 2013). However, no chytrid-related mortality has been reported in these populations, suggesting that California red-legged frogs are less vulnerable to the pathogenic effects of chytrid infection than other amphibian species (Tatarian and Tatarian 2010; Barry and Fellers 2013; Fellers et al. 2017). While chytrid infection may not directly lead to mortality in California red-legged frogs, Padgett-Flohr (2008) states that this infection may reduce overall fitness and could lead to long-term effects. Therefore, it is difficult to estimate the full extent and risk of chytridiomycosis to the California red-legged frog populations.

Five-Year Status Review

On December 16, 2022, the U.S. Fish and Wildlife Service completed a five-year status review of the California red-legged frog, and concluded that this species' threatened status would remain unchanged (USFWS 2022).

Critical Habitat

Critical habitat was designated for this species on April 13, 2006 (USFWS 2006), with revisions to the critical habitat designation published on March 17, 2010 (USFWS 2010). In total, approximately 1,636,609 acres (ac) (662,312 hectares (ha)) of critical habitat in 27 California counties fall within the boundaries of the final revised critical habitat designation.

The PCEs of critical habitat for the California red-legged frog are the habitat components that provide:

1) Aquatic Breeding Habitat. Standing bodies of fresh water (with salinities less than 4.5 ppt), including natural and manmade (e.g., stock) ponds, slow-moving streams or pools within streams, and other ephemeral or permanent water bodies that typically become inundated during winter rains and hold water for a minimum of 20 weeks in all but the driest of years (USFWS 2010).
2) Aquatic Non-Breeding Habitat. Freshwater pond and stream habitats, as described above, that may not hold water long enough for the species to complete its aquatic lifecycle but which provide for shelter, foraging, predator avoidance, and aquatic dispersal of juvenile and adult California red-legged frogs. Other wetland habitats considered to meet these criteria include, but are not limited to: plunge pools within intermittent creeks, seeps, quiet water refugia within streams during high water flows, and springs of sufficient flow to withstand short-term dry periods (USFWS 2010).
3) Upland Habitat. Upland areas adjacent to or surrounding breeding and non-breeding aquatic and riparian habitat up to a distance of 1 mi (1.6 km) in most cases (i.e., depending on surrounding landscape and dispersal barriers) including various vegetational types such as grassland, woodland, forest, wetland, or riparian areas that provide shelter, forage, and predator avoidance for the California red-legged frog. Upland features are also essential in that they are needed to maintain the hydrologic, geographic, topographic, ecological, and edaphic features that support and surround the aquatic, wetland, or riparian habitat. These upland features contribute to: (1) Filling of aquatic, wetland, or riparian habitats; (2) maintaining suitable periods of pool inundation for larval frogs and their food sources; and (3) providing nonbreeding, feeding, and sheltering habitat for juvenile and adult frogs (e.g., shelter, shade, moisture, cooler temperatures, a prey base, foraging opportunities, and areas for predator avoidance). Upland habitat should include structural features such as boulders, rocks and organic debris (e.g., downed trees, logs), small mammal burrows, or moist leaf litter (USFWS 2010).
4) Dispersal Habitat. Accessible upland or riparian habitat within and between occupied or previously occupied sites that are located within 1 mi (1.6 km) of each other, and that support movement between such sites. Dispersal habitat includes various natural habitats, and altered habitats such as agricultural fields, that do not contain barriers (e.g., heavily traveled roads without bridges or culverts) to dispersal. Dispersal habitat does not include moderate- to high-density urban or industrial developments with large expanses of asphalt or concrete, nor does it include large lakes or reservoirs over 50 ac (20 ha) in size, or other areas that do not contain those features identified in PCE 1, 2, or 3 as essential to the conservation of the species (USFWS 2010).

Recovery Plan Information

The Service's *Recovery Plan for the California red-legged frog (Rana aurora draytonii)* (Recovery Plan) was published for the California red-legged frog on September 12, 2002 (USFWS 2002). The Recovery Plan identifies eight recovery units (USFWS 2002). The goal of the Recovery Plan is to protect the long-term viability of all extant populations within each recovery unit. Within each recovery unit, delineated core areas, designed to protect metapopulations, represent contiguous areas of moderate to high California red-legged frog densities. The management strategy identified within this Recovery Plan will allow for the recolonization of habitats within and adjacent to core areas naturally subjected to periodic localized extinctions, thus assuring the long-term survival and recovery of California red-legged frogs.

Environmental Baseline

The California red-legged frog and its designated critical habitat only occur in California. Please refer to information above for the environmental baseline.

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California Tiger Salamander (*Ambystoma californiense*), Central California DPS and its Critical Habitat

Listing Status

The California tiger salamander, Central California DPS was listed as threatened on August 4, 2004 (69 FR 47212). Critical habitat was designated for the California tiger salamander, Central California DPS on August 23, 2005 (70 FR 49380).

Life History and Habitat

Habitat Requirements

Egg: California tiger salamanders breed in deeper vernal pools and wetlands that have sufficiently long periods of inundation to prevent stranding/desiccation. Eggs are attached to a substrate such as twigs, grass stems, or other vegetation or debris (USFWS 2014).

Larvae: Ponding duration is an important factor for breeding success. Wetlands must have a long enough ponding duration for California tiger salamander larvae to mature into juveniles capable of dispersing from the aquatic breeding site to suitable terrestrial habitat. This typically takes 3 months or more, and will vary depending on factors such as water temperature and the depth of the breeding ponds (USFWS 2014).

Adult: California tiger salamander populations are strongly correlated with small burrowing mammal communities, particularly California ground squirrel (*Otospermophilus beecheyi*) and Botta's pocket gopher (*Thommomys bottae*). Adult California tiger salamanders spend roughly 90 percent of any given year underground. Most evidence suggests that California tiger salamanders remain active in their underground dwellings. California tiger salamanders appear to have high site fidelity, returning to their natal pond as adults. After breeding, they commonly return to the same terrestrial habitat areas (USFWS 2014). Although California tiger salamanders are adapted to natural vernal pools and ponds, they now frequently use livestock ponds and other modified ephemeral and permanent ponds surrounded by large tracts of land dominated by grassland, oak savanna, or oak woodland. California tiger salamanders breed in deeper vernal pools and wetlands that have sufficiently long periods of inundation. Breeding pools typically have moderate to high levels of turbidity; California tiger salamanders rarely use ponds with clear water. This species is not known to breed in streams or rivers; however, breeding populations have been reported in ditches that contain seasonal wetlands, and have been documented in sewage treatment ponds in Calaveras County. There has been a shift in habitat use from vernal pools on valley floors to livestock ponds and other artificial wetlands in the foothills (USFWS 2014). Geographic barriers include heavily traveled roads, especially at night during salamander breeding season, so that salamanders almost never successfully traverse the road; roads with a barrier that is impermeable to salamanders; wide, fast rivers; and areas of intensive development dominated by buildings and pavement (NatureServe 2015).

Dispersal/Migration

Peak periods for metamorphs to leave their natal ponds have been reported from May to July. Once metamorphosis occurs, juveniles often depart their natal ponds at night and enter into terrestrial habitat in search of underground burrows. Although wet conditions are more favorable for upland travel, metamorphs typically travel during dry weather because summer rain events seldom occur as metamorphosis is completed and ponds begin to dry. However, if a rain event does occur, it is likely that it will trigger a mass emergence from the natal pond (USFWS 2014). The mean distance that juveniles travel before settling in a burrow is 26 m (85 ft.); dispersal into terrestrial habitat occurs randomly with respect to direction (USFWS 2014). After breeding events, adults and juveniles disperse from the breeding pond in search of small burrowing mammal communities, particularly California ground squirrel (*Otospermophilus beecheyi*) and Botta's pocket gopher (*Thommomys bottae*), or in their absence (especially as recent metamorphs), soil cracks (USFWS 2014). The average dispersal distance is estimated to be 562 m (1,844 ft.). The mean distance adults travel before settling into a burrow is 35.9 m (118 ft.). During the breeding season, rainstorms precede major migrations to breeding sites, with most migrations occurring on rainy nights. Adult California tiger salamanders migrate up to about 2 km (1.25 mi.) between terrestrial habitat and breeding pond (NatureServe 2015; USFWS 2014). However, estimates suggest California tiger salamanders are physiologically capable of migrating up to 1.5 mi. (2.4 km) during a breeding season, and an estimated 95 percent of California tiger salamander populations are thought to occur within 1.86 km (1.16 mi.) of a breeding pond (USFWS 2014).

Reproduction

Egg: Females attach their eggs singly or, in rare circumstances, in groups of two to four (68 FR 28648). After deposition, California tiger salamander eggs hatch in 10 to 28 days; the amount of time for hatching is likely related to water temperatures (USFWS 2014).

Adult: With the onset of the breeding season, typically from November through April (although migrating adults can be observed as early as October and as late as May), adult salamander leave their refugia during rain and storm events in search of breeding ponds (e.g., ephemeral/vernal or perennial water). Males typically arrive before the females, generally remaining in the ponds longer (average of 44.7 days) than the females (average of 11.8 days). The male deposits a spermatophore on the bottom of the pond, which the female picks up and uses to fertilize her eggs internally. Females then attach their eggs to twigs, grass stems, or other vegetation or debris (USFWS 2014). Breeding adults usually range from 1 (rare) or 2 years (typical) old, up to 4 to 5 years of age; females breed an estimated 1.4 times in their lifetime (up to 10 years or more). Given that an estimated 8.5 young survive to metamorphosis per reproductive event, a female's reproductive capacity averages roughly 12 metamorphic offspring over its lifetime (USFWS 2014).

Feeding

Larvae: The California tiger salamander larvae is an opportunistic invertivore/carnivore, and is among the top aquatic predators in the seasonal pool ecosystems. The larvae prey on zooplankton, small crustaceans, and aquatic insects, moving toward larger prey such as the tadpoles of Sierran tree frog (*Pseudacris sierra*), western spadefoot (*Spea hammondii*), and California red-legged frogs (*Rana draytonii*) as they grow in size (USFWS 2014). The larvae often rest on the bottom in shallow water, but also may be found at different layers in the water column in deeper water. The young salamanders are wary; when approached by potential predators, they will dart into vegetation on the bottom of the pool (68 FR 28648). Typical competitors include nonnative and hybrid tiger salamanders and western mosquitofish (*Gambusia affinis*), which can outcompete larvae when they occur (USFWS 2014). Larvae feed for about 6 to 8 weeks after hatching, after which they switch to larger prey (USFWS 2014). The larval stage of the California tiger salamander usually lasts 3 to 6 months, with metamorphosis beginning in late spring or early summer (USFWS 2014). Larvae develop faster in smaller, more rapidly drying pools. The developmental period is prolonged in colder weather and in larger pools; larvae development (time from eggs laid to larvae leaving the pond) has been observed taking from 74 days to 94 days (USFWS 2014).

Adult: The California tiger salamander adult is an opportunistic invertivore/carnivore, foraging predominantly underground during the dry summer months. Invertebrate prey items found in adult salamander stomachs include aphids (Aphididae), wood cockroaches (Blattellidae), ground beetles (Carabidae), springtails (Collembola), centipedes (Cryptopidae, Lithobiidae, and Scolopendra), true weevils (Curculionidae), webspinners (Embioptera), wasps/bees/ants (Hymenoptera), woodlice (Isopoda), silverfish (Lepismatidae), wolf spiders (Lycosidae), owlet moths (Noctuidae), harvestmen (Opiliones), crickets (Rhaphidophoridae), scarab beetles (Scarabaeidae), and crane flies (Tipula). Most evidence suggests that California tiger salamanders remain active in their underground dwellings during the summer months, making frequent underground movements in burrow systems of less than 33 ft. (10 m), but otherwise remaining underground until the onset of rain and the winter months (USFWS 2014).

Population Status

Rangewide Status of the Species

Historically, California tiger salamanders were endemic to the San Joaquin-Sacramento river valleys, bordering foothills, and coastal valleys of Central California. Although the historical distribution of California tiger salamanders is not known in detail, their current distribution suggests that they may have been continuously distributed along the low-elevation grassland-oak woodland plant communities of the valleys and foothills. In this area, the species is known from sites on the Central Valley floor near sea level, up to a maximum elevation of roughly 1,200 meters (m) (3,940 feet [ft.]) in the Coast Ranges and 500 m (1,640 ft.) in the Sierra Nevada foothills (USFWS 2014).

The California tiger salamander – Central California DPS is currently restricted to the Central Valley and Inner Coast Range, from Tulare and San Luis Obispo counties in the south to Sacramento and Yolo counties in the north, and including Alameda, Amador, Calaveras, Contra Costa, Fresno, Kern, Kings, Madera, Mariposa, Merced, Monterey, San Benito, San Mateo, San Joaquin, Santa Clara, Santa Cruz, Stanislaus, Solano, and Tuolumne counties (68 FR 28648). However, along the Central Valley floor, urbanization and intensive agriculture has eliminated virtually all valley grassland and oak savanna habitat from the Central Valley floor; grasslands and, consequently, Central California tiger salamanders are now distributed primarily in a ring around the Central Valley. Likewise, there has also been a significant increase in elevation of localities, suggesting that low-elevation breeding sites have been eliminated where valley floor habitat has been lost (USFWS 2014).

As of 2017, 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 (USFWS 2017).

Population Summary

Both the California tiger salamander (Central California DPS) population levels and the overall California tiger salamander species are decreasing; the total adult population size is unknown, but certainly exceeds 10,000 and likely is at least several 10,000s (NatureServe 2015). The correlation between declining California tiger salamander numbers and surrounding urban and agricultural land uses has been well documented. As of 2002, there was a 20.7 percent loss of known Central California DPS records as a result of habitat loss and degradation. However, because the species spends a majority of its life underground and may not breed every year (= low detectability), it is difficult to determine the exact number of California tiger salamander populations that have been lost due to habitat conversion (USFWS 2014). Although the number of individual extant occurrences of California tiger salamander (Central California DPS) have increased from 638 to 867 since the DPS was first listed in 2004, these do not necessarily correlate with an improvement in status or a reduction in threats to the California tiger salamander; many of these ponds (occurrences) are likely threatened by development, or may have already been destroyed or degraded as a result of development projects. 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. As of 2012, general occurrence data derived from the California Natural Diversity Data Base indicate that there are 257 extant, 18 extirpated, and 12 possibly extirpated occurrences in the Bay Area population; 439 extant, 18 extirpated, and 17 possibly extirpated occurrences in the Central Valley population; 73 extant, 8 extirpated, and 7 possibly extirpated occurrences in the Southern Jan Joaquin Valley population; and 98 extant, 2 extirpated, and 2 possibly extirpated occurrences in the Central Coast Range population (USFWS 2014). The total adult population size is unknown, but certainly exceeds 10,000 and likely is at least several 10,000s (NatureServe 2015).Given the species' comparatively widespread distribution across the landscape, their ecological diversity/variation across their range, and their sensitivity to environmental changes, the species shows a moderate resilience to withstand stochastic events, has a moderate representation to adapt to changing environmental conditions across the landscape, a moderate redundancy to withstand catastrophic events, a low resistance to disease, and low adaptability.

<u>*Threats*</u>

Threats to this species include:

- Urban impacts include development activities such as building and maintenance of housing, commercial, and industrial developments; construction and widening of roads and highways; golf course construction and maintenance; landfill operation and expansion; operation of gravel mines and quarries; and dam building and inundation of habitat by reservoirs (USFWS 2014).
- Agricultural impacts include the conversion of native habitat by discing and deep-ripping; and cultivation, planting, and maintenance of row crops, orchards, and vineyards. Conversion of grasslands to intensive agricultural uses, such as vineyards, orchards, and row crops, has led to the direct loss of Central California tiger salamander populations (USFWS 2014).
- For example, ranavirus diseases such as *Ambystoma tigrinum* virus (ATV) and regina ranavirus (RRV) are known to cause die-offs of other *Ambystoma* species, and although not yet documented to occur in California tiger salamander in the Central California DPS, such diseases are lethal to the species in experimental conditions. If introduced (i.e., by way of nonnative tiger salamanders sold as fishing bait), such diseases could spread from a single pond to an entire metapopulation (USFWS 2014). California tiger salamanders are also susceptible to infection by Chytrid fungus (*Batrachochytrium dedrobatidis*), which causes infected individuals to molt (slough) their entire skin every 2 to 3 days (rather than the typical once every 1 to 2 weeks); this may help prevent mortality, but also requires more energy and reduces individual fitness (USFWS 2014).
- In addition to native predators (amphibians, snakes, turtles, birds, and small mammals), nonnative and exotic predators include bullfrogs (*Rana catesbeiana*); nonnative and hybrid tiger salamanders; western mosquitofish (*Gambusia affinis*) and other introduced fishes like largemouth bass (*Micropterus salmoides*) and blue gill (*Lepomis macrochirus*); nonnative crayfish species (*Pacifastacus*, *Oronectes*, and *Procambarus* sp.), all of which can prey on either the larval or adult (or both) stages of the California tiger salamander (USFWS 2014).
- The primary cause of the decline of the Central California tiger salamander is the loss, degradation, and fragmentation of habitat that results from human activities. There are several state and federal laws and regulations that are pertinent to the protection of Central California tiger salamanders; however, federal, state, and local laws have not been sufficient to prevent past and ongoing losses of the California tiger salamander and its habitat (USFWS 2014).
- The California tiger salamander – Central California DPS has been heavily affected by hybridization. The large-scale introduction of barred tiger salamander was first reported in the Salinas Valley about 60 years ago, when many tens of thousands of barred tiger salamander (*Ambystoma mavortium*) were introduced in support of the bass-bait industry (USFWS 2014).
- Sources of chemical pollution that may adversely affect California tiger salamander (Central

California DPS) include hydrocarbon and other contaminants from oil production and road runoff; the application of chemicals for agricultural production and urban/suburban landscape maintenance; and increased nitrogen levels in aquatic habitats. Amphibians in general are extremely sensitive to contaminants, due to their highly permeable skin. Exposure to pesticides can increase their susceptibility to parasitic or bacterial infections, alter their rates of metamorphosis, lead to growth abnormalities, reduce their overall fitness, and lead to increased mortality (USFWS 2014).

- Because ground squirrels and pocket gophers are critical for burrow construction and maintenance, and therefore critical to the California tiger salamander, rodent population control efforts are a potential threat to California tiger salamanders. Eradication techniques include the application of poisoned grains; fumigant rodenticide; gases (including aluminum phosphide, carbon monoxide, and methyl bromide) introduced into burrows through cartridges, pellets, and other methods; and combustible gas injected into burrow complexes and then ignited (USFWS 2014).
- The distribution of the California tiger salamander (Central California DPS) spans a considerable range in climatic conditions (including annual variation), and it is uncertain how the various sub-populations of the Central California tiger salamander might differ in their responses to climate change (USFWS 2014).

Five-Year Status Review

There have been two five-year status reviews for this species: one on October 21, 2014 and one on August 10, 2023. The latest five-year status review concluded the Central California population of the California tiger salamander continues to meet the definition of threatened and would remain a threatened species (USFWS, 2014, USFWS 2024).

Critical Habitat

On August 23, 2005, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the Central population of the California tiger salamander pursuant to the Endangered Species Act of 1973, as amended (70 FR 49380). In total, approximately 199,109 acres (ac) (80,576 hectares (ha)) fall within the boundaries of the critical habitat designation. The critical habitat is located within 19 counties in California.

The critical habitat designation for *Ambystoma californiense* includes 31 units totaling 199,109 acres in four geographic regions in California. The four regions containing critical habitat are: (1) The Central Valley Region; (2) the Southern San Joaquin Valley Region; (3) the East Bay Region (including Santa Clara Valley area); and (4) the Central Coast Region.

The PCEs of critical habitat for the Central population of the California tiger salamander are the habitat components that provide:

(i) Standing bodies of fresh water (including natural and manmade (e.g., stock)) ponds, vernal pools, and other ephemeral or permanent water bodies which typically support inundation during winter rains and hold water for a minimum of 12 weeks in a year of average rainfall;
(ii) Upland habitats adjacent and accessible to and from breeding ponds that contain small mammal burrows or other underground habitat that CTS depend upon for food, shelter, and protection from the elements and predation; and
(iii) Accessible upland dispersal habitat between occupied locations that allow for movement between such sites.

Recovery Plan Information

On June 6, 2017, the Recovery Plan for the Central California DPS of the California tiger salamander was issued (USFWS 2017).

Recovery Actions

- Reduce Road Mortality: Coordinate with transportation agencies to incorporate wildlife tunnels in design plans for new roads and road improvement projects to decrease Central California tiger salamander road mortality (USFWS 2017).
- Reduce road mortality. Upgrade existing roads to include wildlife tunnels to decrease Central California tiger salamander road mortality (USFWS 2017).
- Reduce the risk of introduction of diseases (e.g., ranaviruses, chytrid fungi, or other pathogens) within preserves. Monitor breeding sites to detect disease outbreaks. Monitoring should be conducted during the breeding season to detect rapid die-offs of larvae, which may be the result of ranavirus, chytrid or other pathogens (USFWS 2017).
- Reduce the risk of introduction of diseases (e.g., ranaviruses, chytrid fungi, or other pathogens) within preserves. Determine the cause of die-offs. If a rapid die-off is detected, tests for ranaviruses, chytrid fungi, or other pathogens should be conducted immediately. Land managers should coordinate with the Service and CDFW to determine the appropriate next steps (USFWS 2017).
- Reduce the risk of introduction of diseases (e.g., ranaviruses, chytrid fungi, or other pathogens) within preserves. Develop contingency plans. Contingency plans should be incorporated into all management plans to ensure that a population infected with a ranavirus, chytrid fungus, or other pathogen is quickly isolated and the disease does not spread to uncontaminated populations (USFWS 2017).
- Reduce the risk of introduction of diseases (e.g., ranaviruses, chytrid fungi, or other pathogens) within preserves. Develop measures to sterilize field equipment to minimize disease transmission (USFWS 2017).
- Reduce levels of non-native predator species within preserves. Reduce populations of non- native predators to a level where they are determined to not decrease Central California tiger salamander populations (USFWS 2017).
- Reduce levels of non-native predator species within preserves. Identify sites within each preserve that require non-native predator eradication or control. As a short-term method, physical removal of these non-native species may be most beneficial. However, proactive means of reducing the conditions in which these non-native species thrive is a long-term priority (see action 1.2.2 for a description of optimal breeding habitat to reduce non-native predators) (USFWS 2017).
- Reduce levels of non-native predator species within preserves. Prohibit introduction of fish species to breeding habitat or within any aquatic system that has the potential to convey non-native fish to breeding habitat (USFWS 2017).
- Develop and implement adaptive management and monitoring plans for protected habitat counted toward recovery. All preserves (as described in recovery criteria A/1 through A/4) should have management and monitoring plans. These plans should specifically target management and monitoring of Central California tiger salamander breeding and upland habitat to maintain habitat suitability in perpetuity. The plans may include, but are not limited to, actions to identify and reduce: harmful contaminants, non-native predator species, road mortality, and non-native tiger salamanders and hybrids. Management plans should describe grazing management and disease prevention strategies. Plans should be updated based on feedback from land managers and adaptive to climate change and other variables (USFWS 2017).
- Develop and implement adaptive management and monitoring plans for protected habitat counted toward recovery. Secure funding in perpetuity for habitat management and monitoring either through an endowment or other funding mechanism (USFWS 2017).

- Develop and implement adaptive management and monitoring plans for protected habitat counted toward recovery. Management plans should be developed to ensure high quality upland and breeding habitat is available for the Central California tiger salamander in perpetuity (USFWS 2017).
- Monitor trends to gain a better understanding of population health, trends in habitat loss, and other information that will help to guide conservation planning for the Central California tiger salamander.
    1. Establish and maintain a database that tracks the amount of incidental take authorized through section 7 and 10 of the Act.
    2. Monitor habitat land use change. Utilize GIS land use cover data to determine amount of suitable habitat that has been lost.
    3. Survey lands for Central California tiger salamander in areas that have not been well surveyed. The following management units have not been well surveyed: Dunnigan Hills, Central Valley West Side, Farmington, Oakdale/Waterford, Northeast Diablo Range, and Southeast Diablo. Other areas will likely require surveys as well.
    4. Conduct population viability analyses for Central California tiger salamander metapopulations throughout the range of the DPS. Population viability analyses are tools that can identify populations in need of recovery actions, as opposed to those that may be viable over the long-term without intervention.
    5. Research should be conducted to determine the effectiveness of standard avoidance and minimization measures (e.g., exclusion fencing, burrow excavation, and seasonal work windows) to ensure the most successful measures are being used during implementation of projects that may impact Central California tiger salamanders and their habitat.
    6. Conduct research on the effects of contaminants.
        - 6.1. Conduct investigations on effects of contaminants on Central California tiger salamander (or a surrogate salamander species if determined appropriate).
        - 6.2. Conduct research that determines which pesticides and other contaminants are commonly used on agriculture lands within the range of the Central California tiger salamander.
        - 6.3. Conduct research on the effects of mosquito abatement chemicals on Central California tiger salamander populations.
    7. Conduct genetic research.
        - 7.1. Monitor projects designed to increase native species genomes and limit hybridization. These studies should occur within a variety of geographic areas (e.g., Salinas Valley floor, foothill areas to the north and east of Salinas Valley, and Bay Area) to determine the most effective strategies in various geographic areas.
        - 7.2. Conduct focused research on SI alleles to determine how each non-native gene is physically expressed and the subsequent ecological impact of these genes.
        - 7.3. Conduct landscape genomic research and climate change modeling to identify genetic variability that may provide resiliency to climate change and identify areas of climate refugia.
    8. Conduct research on small burrowing mammal communities.
        - 8.1. Conduct research to determine burrow requirements for Central California tiger salamander populations (i.e., what burrow densities are optimal for Central California tiger salamanders, and how many small burrowing mammals are required to maintain these densities?).
        - 8.2. Conduct research to determine optimum grazing regimes to increase small mammal burrowing communities (USFWS 2017).
- Develop and implement participation plans for each Recovery Unit. Participation plans will assist in the realization of recovery goals by facilitating commitments from participating agencies and

stakeholders to implement recovery actions, where feasible (USFWS 2017).

Environmental Baseline

The Central California DPS of the California tiger salamander and its designated critical habitat occur in the Central Valley and Inner Coast Range, California. Please refer to information above for the environmental baseline.

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California Tiger Salamander (*Ambystoma californiense*), Santa Barbara County DPS and its Critical Habitat

Listing Status

On September 21, 2000, the Service emergency listed the Santa Barbara County DPS of the California tiger salamander as endangered (65 Federal Register (FR) 3096). In 2004, the Service designated critical habitat for the Santa Barbara County DPS of the California tiger salamander (69 FR 68568). At the time of publication of the emergency listing rule in January 2000, the Santa Barbara County California tiger salamander was known from 14 ponds. The emergency and final listing rules acknowledged that other potential breeding ponds or pond complexes may exist, but could not be surveyed at that time due to restricted access.

Life History and Habitat

Historically, the Santa Barbara County California tiger salamander inhabited low-elevation (below 475 meters (1,500 ft)) seasonal ponds and associated grassland, oak savannah, and coastal scrub plant communities of the Santa Maria, Los Alamos, and Santa Rita Valleys in the northwestern area of Santa Barbara County (Shaffer et al. 1993, p. 4). California tiger salamanders spend the majority of their lives in upland habitats and cannot persist without them (Trenham and Shaffer 2005, p. 1165). The upland component of California tiger salamander habitat consists of grassland savannah, but includes grasslands with scattered oak trees, and scrub or chaparral habitats (Shaffer et al. 1993, p. 4; 65 FR 3096). Juvenile and adult California tiger salamanders spend the dry summer and fall months of the year in the burrows of small mammals, such as California ground squirrels (*Otospermophilus beecheyi*) and Botta's pocket gopher (*Thomomys bottae*) (Loredo et al. 1996b, p. 283; Cook et al. 2006, p. 216). In general, studies show that adults can move 2 kilometers (1.2 miles) to more than 2.2 kilometers (1.4 miles) from breeding ponds (Trenham et al. 2001, p. 3526; Orloff 2011, p, 270).

Population Status

Currently, there are approximately 60 known extant California tiger salamander breeding ponds in Santa Barbara County (Service 2009, p. 9) distributed across the six metapopulations. Since listing, Service and CDFW developed guidance for protocol survey efforts (Service and Department 2003), and this guidance has aided in the detection of additional breeding ponds discovered post-listing. Several of the additional ponds were discovered as a result of surveys conducted as a part of proposed development or land conversion projects. The Santa Barbara County DPS of the California tiger salamander is threatened primarily by the destruction, degradation, and fragmentation of upland and aquatic habitats, primarily resulting from the conversion of these habitats by urban, commercial, and intensive agricultural activities (Service 2016). Additional threats to the species include hybridization with introduced nonnative barred tiger salamanders (*A. tigrinum mavortium*) (Service 2016, p. I-16), destructive rodent-control techniques (e.g., deep-ripping of burrow areas, use of fumigants) (Service 2016, p. I-10), reduced survival due to the presence of mosquitofish (*Gambusia affinis*) (Leyse and Lawlor 2000, p. 76), and mortality on roads due to vehicles (65 FR 3096).

Critical Habitat

A total of 4,523 hectares (11,180 acres) in six separate units are designated as critical habitat for the California tiger salamander in Santa Barbara County. Per the final critical habitat designation, the PCEs within the defined area that are essential to the conservation of the species include:

1. Standing bodies of fresh water, including natural and man-made (e.g., stock) ponds, vernal pools, and dune ponds, and other ephemeral or permanent water bodies that typically become inundated during winter rains and hold water for a sufficient length of time (i.e., 12 weeks) necessary for the species to complete the aquatic portion of its lifecycle;

2. Barrier-free uplands adjacent to breeding ponds that contain small mammal burrows. Small mammals are essential in creating the underground habitat that adult California tiger salamanders depend upon for food, shelter, and protection from the elements and predation; and
3. Upland areas between breeding locations (PCE 1) and areas with small mammal burrows (PCE 2) that allow for dispersal among such sites (69 FR 6858).

Recovery Plan Information

The goal of the recovery plan for the Santa Barbara County DPS of California tiger salamander (Service 2016) is to reduce the threats to the population to ensure its long-term viability in the wild, and allow for its removal from the list of threatened and endangered species. The interim goal is to recover the population to the point that it can be downlisted from endangered to threatened status. The overall objectives of the recovery plan are to (1) protect and manage sufficient habitat within the metapopulation areas to support long-term viability of the Santa Barbara County DPS of the California tiger salamander and (2) reduce or remove other threats to the Santa Barbara County DPS of the California tiger salamander.

Five-Year Status Review

There have been two five-year status reviews for this species: one on November 16, 2009 and one on July 19, 2022. Both five-year status reviews concluded the Santa Barbara County population of the California tiger salamander continues to meet the definition of endangered and would remain an endangered species (Service, 2009; Service 2022).

Environmental Baseline

The species only occurs within the State of California, please refer to the above information regarding the species environmental baseline.

Literature Cited

Cook, D.G, P.C. Trenham, and P.T. Northen. 2006. Demography and breeding phenology of the California tiger salamander (*Ambystoma californiense*) in an urban landscape. Northwestern Naturalist 87(3):215-224.Trenham et al. 2001, p. 3526.

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Foothill Yellow-legged Frog (*Rana boylii*), Central Coast DPS

Listing Status

The Central Coast DPS of the foothill yellow-legged frog was proposed for listing as threatened on December 28, 2021 (86 FR 73914). After review of the best scientific and commercial information available, the Service published a final rule to list the Central Coast DPS of the foothill yellow-legged frog as threatened, with a Section 4(d) rule, on August 29, 2023 (88 FR 59698). The Service proposed designation of critical habitat for the Central Coast DPS of the foothill yellow-legged frog on January 15, 2025 (90 CFR 3412). Critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Life History and Habitat

The foothill yellow-legged frog is a small- to medium-sized stream-dwelling frog with fully webbed feet and rough pebbly skin. The foothill yellow-legged frog is a stream-obligate species. Stream habitat for the species is highly variable and keyed on flow regimes. Habitat within the stream includes rocky substrate mostly free of sediments with interstitial spaces to allow for predator avoidance. Stream morphology is a strong predictor of breeding habitat because it creates the microhabitat conditions required for successful oviposition (i.e., egg-laying), hatching, growth, and metamorphosis (86 FR 73914).

Population Status

The Central Coast DPS extends south from the San Francisco Bay through the Diablo Range and through the Coast Range (Santa Cruz Mountains and Gabilan Mountains) east of the Salinas Valley. It is unknown whether foothill yellow-legged frogs historically occupied San Francisco County (CDFW 2019, p. 38 in Service 2021). On average, the Central Coast DPS receives the least amount of annual precipitation of all the DPSs (PRISM Climate Group 2012, 30-year climate dataset in Service 2021). Ecoregions that are unique to the Central Coast DPS include those associated with the Diablo Range (6r, 6x, and 6z), Santa Cruz Mountains (1n), San Mateo Coastal Hills (1o), Eastern Hills (6aa), Bay Terraces/Lower Santa Clara Valley (6t), Upper Santa Clara Valley (6v), and Livermore Hills and Valleys (6u) (Environmental Protection Agency Level IV Ecoregions (Omerick and Griffith 2014, entire; Griffith *et al.* 2016, entire, all in Service 2021)). Although the mountain ranges of the Central Coast DPS are geologically unique and separated from those of the South Coast DPS by the Salinas Valley, there are several attributes that are similar between the two DPSs. For example, there are similarities in mountain elevation range, elevation grade, and some vegetation types (Griffith et al. 2016, entire in Service 2021). The Central Coast and South Coast DPSs are both warm and dry (PRISM Climate Group 2012, 30-year climate dataset in Service 2021) and their waterways are similar in terms of hydrological properties to the South Coast DPS in they tend to have flashier flows, more ephemeral channels, and a higher degree of intermittency because of the region’s more variable, and lower amount of, precipitation (Storer 1925, pp. 257– 258; Gonsolin 2010, p. 54; Adams et al. 2017, p. 10227, all in Service 2021).

Critical Habitat

The Service proposed designation of critical habitat for the Central Coast DPS of the foothill yellow-legged frog. However, critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The Central Coast DPS only occurs in California, please refer to the information above regarding the species environmental baseline.

Literature Cited

Adams, A.J., A.P. Pessier, and C.J. Briggs. 2017. Rapid Extirpation of a North American Frog Coincides with an Increase in Fungal Pathogen Prevalence: Historical Analysis and Implications for Reintroduction. Ecology and Evolution 7(23):10216–10232. DOI 10.1002/ece3.3468.

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88 FR 59698. Endangered and Threatened Wildlife and Plants; Foothill Yellow-Legged Frog; Threatened Status with Section 4(d) Rule for Two Distinct Population Segments and Endangered for Two Distinct Population Segments. Final Rule. Vol 88, No. 166. Federal Register 59698. August 29, 2023.

90 CFR 3412. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for Four Distinct Population Segments of the Foothill Yellow-Legged Frog, Proposed Rule. Vol. 90, No. 8. Federal Register 3412. January 14, 2025.

Foothill Yellow-legged Frog (*Rana boylii*), North Feather DPS

Listing Status

The North Feather DPS of the foothill yellow-legged frog was proposed for listing as threatened on December 28, 2021 (86 FR 73914). After review of the best scientific and commercial information available, the Service published a final rule to list the North Feather DPS of the foothill yellow-legged frog as threatened, with a Section 4(d) rule, on August 29, 2023 (88 FR 59698). The Service proposed designation of critical habitat for the North Feather DPS of the foothill yellow-legged frog on January 15, 2025 (90 CFR 3412). Critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Life History and Habitat

The foothill yellow-legged frog is a small- to medium-sized stream-dwelling frog with fully webbed feet and rough pebbly skin. The foothill yellow-legged frog is a stream-obligate species. Stream habitat for the species is highly variable and keyed on flow regimes. Habitat within the stream includes rocky substrate mostly free of sediments with interstitial spaces to allow for predator avoidance. Stream morphology is a strong predictor of breeding habitat because it creates the microhabitat conditions required for successful oviposition (i.e., egg-laying), hatching, growth, and metamorphosis (86 FR 73914).

Population Status

The North Feather DPS is located primarily in Plumas and Butte counties. This DPS occupies the transition zone between the northern Sierra Nevada, Southern Cascades Foothills, and Tuscan Flows ecoregions. The Tuscan Flows is an ecoregion that is geologically related to the Cascades but has similarities to the Sierra Nevada Foothills ecoregion (Environmental Protection Agency Level IV Ecoregions (Omerick and Griffith 2014, entire; Griffith *et al.* 2016, entire, all in Service 2021)). The North Feather DPS differs from the surrounding watersheds in terms of geology and aspect (Peek *et al.* 2019, p. 4638 in Service 2021), and is the only known area where the foothill yellow-legged frog and Sierra Nevada yellow-legged frog currently coexist (Peek *et al.* 2019, p. 4637 in Service 2021). As expected by its position at the northern end of the Sierra Nevada Range, the North Feather DPS averages cooler and wetter than the DPSs to the south (PRISM Climate Group 2012, 30-year climate dataset in Service 2021).

Critical Habitat

The Service proposed designation of critical habitat for the North Feather DPS of the foothill yellow-legged frog. However, critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The North Feather DPS only occurs in California, please refer to the information above regarding the species environmental baseline.

Literature Cited

Griffith, G.E., J.M. Omernik, D.W. Smith, T.D. Cook, E. Tallyn, K. Moseley, and C.B. Johnson. 2016. Ecoregions of California (poster): U.S. Geological Survey Open-File Report 2016– 1021, with map, scale 1:1,100,000, http://dx.doi.org/10.3133/ofr20161021.

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90 CFR 3412. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for Four Distinct Population Segments of the Foothill Yellow-Legged Frog, Proposed Rule. Vol. 90, No. 8. Federal Register 3412. January 14, 2025.

Foothill Yellow-legged Frog (*Rana boylii*), South Coast DPS

Listing Status

The South Coast DPS of the foothill yellow-legged frog was proposed for listing as endangered on December 28, 2021 (86 FR 73914). After review of the best scientific and commercial information available, the Service published a final rule to list the South Coast DPS of the foothill yellow-legged frog as endangered on August 29, 2023 (88 FR 59698). The Service proposed designation of critical habitat for the South Coast DPS of the foothill yellow-legged frog on January 15, 2025 (90 CFR 3412). Critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Life History and Habitat

The foothill yellow-legged frog is a small- to medium-sized stream-dwelling frog with fully webbed feet and rough pebbly skin. The foothill yellow-legged frog is a stream-obligate species. Stream habitat for the species is highly variable and keyed on flow regimes. Habitat within the stream includes rocky substrate mostly free of sediments with interstitial spaces to allow for predator avoidance. Stream morphology is a strong predictor of breeding habitat because it creates the microhabitat conditions required for successful oviposition (i.e., egg-laying), hatching, growth, and metamorphosis (86 FR 73914).

Population Status

The South Coast unit extends along the coastal Santa Lucia Range and the Sierra Madre Mountains. This unit is also believed to include an isolated, historical population in the San Gabriel Mountains (Los Angeles County), which is 77 km (48 mi) from the closest foothill yellow-legged frog population in record (Zweifel 1955, p. 239 in Service 2021). Ecoregions that are unique to the South Coast unit include those associated with the Santa Lucia Range (6ag–6aj), Western Transverse Range (8a–8b), and Southern California Lower Montane Shrub and Woodland (8e) (Environmental Protection Agency Level IV Ecoregions (Omerick and Griffith 2014, entire; Griffith et al. 2016, entire, all in Service 2021)).While the streams and rivers in the South Coast unit are different from those in most other parts of the foothill yellow-legged frog range, they share similarities to many waterways in the Central Coast unit. Waterways in the South Coast and Central Coast units tend to have flashier flows, more ephemeral channels, and a higher degree of intermittency because of the region’s more variable, and lower amount of, precipitation (Storer 1925, pp. 257– 258; Gonsolin 2010, p. 54; Adams et al. 2017, p. 10227, all in Service 2021). The South Coast and Central Coast units receive the least amount of annual precipitation and average the warmest temperatures within the species’ range (Table 3; PRISM Climate Group 2012, 30-year climate dataset in Service 2021).

Critical Habitat

The Service proposed designation of critical habitat for the South Coast DPS of the foothill yellow-legged frog. However, critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The Central Coast, North Feather, South Coast, and Southern Sierra DPS only occurs in California, please refer to the information above regarding the species environmental baseline.

Literature Cited

Adams, A.J., A.P. Pessier, and C.J. Briggs. 2017. Rapid Extirpation of a North American Frog Coincides with an Increase in Fungal Pathogen Prevalence: Historical Analysis and Implications for Reintroduction. Ecology and Evolution 7(23):10216–10232. DOI 10.1002/ece3.3468

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Zweifel, R.G. 1955. Ecology, Distribution, and Systematics of Frogs of the *Rana boylei* Group. University of California Publications in Zoology 54(4):207–292.

88 FR 73914. Endangered and Threatened Wildlife and Plants; Foothill Yellow-Legged Frog; Threatened Status with Section 4(d) Rule for Two Distinct Population Segments and Endangered Status for Two Distinct Population Segments, Proposed Rule. Vol 86, No. 246. Federal Register 73914. December 28, 2021.

88 FR 59698. Endangered and Threatened Wildlife and Plants; Foothill Yellow-Legged Frog; Threatened Status with Section 4(d) Rule for Two Distinct Population Segments and Endangered for Two Distinct Population Segments. Final Rule. Vol 88, No. 166. Federal Register 59698. August 29, 2023.

90 CFR 3412. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for Four Distinct Population Segments of the Foothill Yellow-Legged Frog, Proposed Rule. Vol. 90, No. 8. Federal Register 3412. January 14, 2025.

Foothill Yellow-legged Frog (*Rana boylii*), South Sierra DPS

Listing Status

The South Sierra DPS of the foothill yellow-legged frog was proposed for listing as endangered on December 28, 2021 (86 FR 73914). After review of the best scientific and commercial information available, the Service published a final rule to list the South Sierra DPS of the foothill yellow-legged frog as endangered on August 29, 2023 (88 FR 59698). The Service proposed designation of critical habitat for the South Sierra DPS of the foothill yellow-legged frog on January 15, 2025 (90 CFR 3412). Critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Life History and Habitat

The foothill yellow-legged frog is a small- to medium-sized stream-dwelling frog with fully webbed feet and rough pebbly skin. The foothill yellow-legged frog is a stream-obligate species. Stream habitat for the species is highly variable and keyed on flow regimes. Habitat within the stream includes rocky substrate mostly free of sediments with interstitial spaces to allow for predator avoidance. Stream morphology is a strong predictor of breeding habitat because it creates the microhabitat conditions required for successful oviposition (i.e., egg-laying), hatching, growth, and metamorphosis (86 FR 73914).

Population Status

The South Sierra DPS extends from the South Fork American River sub-basin to the transition zone between the Sierra Nevada and the Tehachapi Mountains that border the south end of the California Central Valley. This DPS largely includes ecoregions that are unique to the southern and central Sierra Nevada Range including the Southern Sierra Mid-Montane Forests (5m), Southern Sierra Lower Montane Forest and Woodland (5n), Southern Sierran Foothills (6c), Tehachapi Mountains (5o), and Tehachapi Foothills (6ae) (Environmental Protection Agency Level IV Ecoregions (Omerick and Griffith 2014, entire; Griffith *et al.* 2016, entire, all in Service 2021)). The South Sierra DPS also shares an ecoregion transition zone with the North Sierra DPS (Omerick and Griffith 2014, entire; Griffith *et al.* 2016, entire, all in Service 2021). Average precipitation and temperature in the South Sierra DPS is fairly dry and warm (PRISM Climate Group 2012, 30-year climate dataset in Service 2021).

Critical Habitat

Critical habitat has not been designated for this species.The Service proposed designation of critical habitat for the South Sierra DPS of the foothill yellow-legged frog. However, critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The South Sierra DPS only occurs in California, please refer to the information above regarding the species environmental baseline.

Literature Cited

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Mountain Yellow-legged Frog (*Rana muscosa*), Northern DPS and its Critical Habitat

Listing Status

The mountain yellow-legged frog was listed as endangered, effective on June 30, 2014 (79 FR 24256). Critical habitat was designated for the mountain yellow-legged frog on August 26, 2016 (81 FR 59045).

Life History and Habitat

Mountain yellow-legged frogs currently exist in montane regions of the Sierra Nevada of California in lakes, ponds, marshes, meadows, and streams at elevations ranging from 1,370 to 3,660 m (4,500 to 12,000 feet ft.). Mountain yellow-legged frogs are highly aquatic, are generally not found more than 1 m (3.3 ft.) from water (78 FR 24471; CDFG 2011), and display strong site fidelity, returning to the same overwintering and summer habitats from year to year (78 FR 24471). Both adult and tadpole mountain yellow-legged frogs overwinter for up to 9 months in the bottoms of lakes that are at least 1.7 m (5.6 ft.) deep; however, overwinter survival may be greater in lakes that are at least 2.5 m (8.2 ft.) deep (78 FR 24471). Where water depths range from 0.2 m (0.7 ft.) to 1.5 m (5 ft.), the availability of rock crevices, holes, and ledges near shore offer protection to overwintering frogs when water bodies freeze over completely (78 FR 24471).

Mountain yellow-legged frogs are highly aquatic; they are generally not found more than 1 m (3.3 ft.) from water (78 FR 24471; CDFG 2011). Adults typically are found sitting on rocks along the shoreline, usually where there is little or no vegetation. Although mountain yellow-legged frogs may use a variety of shoreline habitats, both tadpoles and adults are less common at shorelines that drop abruptly to a depth of 60 cm (2 ft.) than at open shorelines that gently slope up to shallow waters of only 5 to 8 cm (2 to 3 in) in depth (78 FR 24471). At lower elevations within their historical range, these species are known to be associated with rocky streambeds and wet meadows surrounded by coniferous forest. Streams used by adults vary from streams having high gradients and numerous pools, rapids, and small waterfalls; to streams with low gradients and slow flows, marshy edges, and sod banks. Aquatic substrates vary from bedrock to fine sand, rubble (rock fragments), and boulders. Mountain yellow-legged frogs appear absent from the smallest creeks, probably because these creeks have insufficient depth for adequate refuge and overwintering habitat. Sierra Nevada yellow-legged frogs do use stream habitats, especially the remnant populations in the northern part of their range. At higher elevations, these species occupy lakes, ponds, tarns (small steep banked mountain lake or pool), and streams. Mountain yellow-legged frogs in the Sierra Nevada are most abundant in high-elevation lakes and slow-moving portions of streams. The borders of alpine (above the tree line) lakes and mountain meadow streams used by mountain yellow-legged frogs are frequently grassy or muddy. This differs from the sandy or rocky shores inhabited by mountain yellow- legged frogs in lower elevation streams. Both adult and tadpole mountain yellow-legged frogs overwinter for up to 9 months in the bottoms of lakes that are at least 1.7 m (5.6 ft.) deep; however, overwinter survival may be greater in lakes that are at least 2.5 m (8.2 ft.) deep (78 FR 24471). Where water depths range from 0.2 m (0.7 ft.) to 1.5 m (5 ft.), the availability of rock crevices, holes, and ledges near shore offer protection to overwintering frogs when water bodies freeze over completely (78 FR 24471).

Adults emerge from overwintering sites at spring thaw or snowmelt and commence breeding soon thereafter—between April and May at lower elevations and progressively later (June and July) at higher elevations (CDFG 2011). Eggs are deposited underwater in the shallows of ponds or in inlet streams in clusters, which they attach to rocks, gravel, or vegetation, or which they deposit under banks. Because tadpoles must overwinter multiple years before metamorphosis, successful breeding sites are located in (or connected to) lakes and ponds that do not dry out in the summer, and also are deep enough that they do not completely freeze or become oxygen depleted (anoxic) in winter. The eggs are deposited in globular clumps, which are often somewhat flattened and roughly 2.5 to 5 cm (1 to 2 in.) in diameter (78

FR 24471; CDFG 2011). Clutch size varies from 15 to 350 eggs per egg mass. Egg hatching time ranges from 16 to 21 days at temperatures of 5 to 13.5 °C (41 to 56°F). The time required to reach reproductive maturity in mountain yellow-legged frogs is thought to vary between 3 and 4 years post-metamorphosis. In combination with the extended amount of time as a tadpole before metamorphosis, it may take 5 to 8 years for mountain yellow-legged frogs to begin reproducing (78 FR 24471; CDFG 2011). Longevity of adults is unknown, but adult survivorship from year to year is very high under normal circumstances. Mountain yellow-legged frogs are presumed to be long-lived amphibians (78 FR 24471; CDFG 2011).

<u>Juvenile</u>: Mountain yellow-legged frogs are omnivorous, feeding as tadpoles on algae, diatoms, and detritus. Tadpoles forage for prey at the bottoms of lakes, ponds, and streams, in shallow waters. During winter, tadpoles remain in warmer water below the thermocline; in the spring, when warmer days raise surface water temperatures, they move to shallow, near-shore water, retreating during the late afternoon and evening to offshore waters that are less subject to night cooling (78 FR 24471; CDFG 2011). Tadpoles may take more than 1 year, and often require 2 to 4 years, to reach metamorphosis (transformation from tadpoles to frogs), depending on local climate conditions and site-specific variables (78 FR 24471; CDFG 2011).

<u>Adult</u>: Mountain yellow-legged frogs are omnivorous, feeding in adulthood on a diet of terrestrial and aquatic insects and macro invertebrates, other amphibians, and the occasional cannibalism of eggs and tadpole/adult carcasses. Adults forage for prey at the bottoms of lakes, ponds, and streams; in shallow waters; and onshore. As adults, frogs maximize body temperatures during a majority of the day by basking in the sun, moving between water and land, and concentrating in the warmer shallows along the shoreline. As temperatures decrease in the fall, frogs become less active and move to overwintering habitats (78 FR 24471; CDFG 2011). With the widespread introduction of nonnative trout, nearly all large, deep lakes that could provide suitable overwintering habitat for frogs are now occupied by introduced trout. In addition to their role as predators of mountain yellow-legged frogs, trout are also competitors for the same invertebrate species that frogs rely on for food. The direct impacts of trout predation on invertebrates can have a negative effect on frogs via competition for invertebrate prey; and can alter lake nutrient cycles, resulting in negative impacts to frogs and other native species (CDFG 2011).

Population Status

<u>Rangewide Status of the Species</u>

Mountain yellow-legged frogs were historically abundant across much of the higher elevations of the Sierra Nevada. The precise historical ranges of the Sierra Nevada yellow-legged frog and the mountain yellow-legged frog are difficult to determine, because projections must be inferred from museum collections that do not reflect systematic surveys; and historic survey information is very limited. Sierra Nevada yellow-legged frogs occupy the western Sierra Nevada north of the Monarch Divide (in Fresno County) and the eastern Sierra Nevada (east of the crest) in Inyo and Mono counties. The northern DPS of the mountain yellow-legged frog extends in the western Sierra Nevada from south of the Monarch Divide in Fresno County through portions of the Kern River drainage; the southern DPS of the mountain yellow-legged frog occupies the canyons of the Transverse Ranges in southern California. The ranges of the two frog species in the mountain yellow-legged complex therefore meet each other roughly along the Monarch Divide to the north, and along the crest of the Sierra Nevada to the east (78 FR 24471; CDFG 2011).

Currently, mountain yellow-legged frogs exist in montane regions of the Sierra Nevada of California at elevations ranging from 1,370 to 3,660 meters (m) (4,500 to 12,000 feet [ft.]). Sierra Nevada yellow-legged frogs occupy the western Sierra Nevada north of the Monarch Divide (in Fresno County) and the eastern Sierra Nevada (east of the crest) in Inyo and Mono counties.

Researchers have reported disappearances of these species from a large fraction of their historical ranges in the Sierra Nevada, with their distributions currently restricted primarily to publicly-managed lands at high elevations, including streams, lakes, ponds, and meadow wetlands in National Forests and National Parks. The most pronounced declines in the mountain yellow-legged frog complex have occurred north of Lake Tahoe in the northernmost 125-kilometer (km) (78-mile [mi.]) portion of the range (Sierra Nevada yellow-legged frog) and south of Sequoia and Kings Canyon National Parks in Tulare County, in the southernmost 50-km (31- mi.) portion, where only a few populations of the northern DPS of the mountain yellow-legged frog remain. Mountain yellow-legged frog populations have persisted in greater density in the National Parks of the Sierra Nevada than in the surrounding U.S. Forest Service (USFS) lands, and the populations that do occur in the National Parks generally exhibit greater abundances than those on USFS lands. Currently, the northern DPS of the mountain yellow-legged frog is discrete from the southern DPS because it is separated from the southern frogs by a 140-mi. (225-km) barrier of unsuitable habitat (78 FR 24471; 79 FR 24255; CDFG 2011).

<u>*Population Summary*</u>

Monitoring efforts and research studies have documented substantial declines of mountain yellow-legged frog populations in the Sierra Nevada. The number of extant populations has declined greatly over the last few decades. Remaining populations are patchily scattered throughout the historical range. Documented extirpations appear to occur nonrandomly across the landscape, are typically spatially clumped, and involve the disappearance of all or nearly all of the mountain yellow-legged frog populations in a watershed. Over the available historical record, estimates range from losses between 69 to 93 percent. Range-wide reduction has diminished the number of watersheds that support mountain yellow-legged frogs (*R. sierrae*), at a conservative estimate of 59 percent. Remaining populations are much smaller than historical norms, and the density of populations per watershed has declined substantially; as a result, many watersheds currently support single metapopulations at low abundances. Remaining populations are generally very small, and available information indicates that the rates of population decline have not abated, and they have likely accelerated during the 1990s into the 2000s (79 FR 24255). Southern DPS: Southern *Rana muscosa*, which historically was widely distributed in at least 166 known populations across four mountain ranges in southern California, are currently considered to be extant in 10 small populations distributed disproportionately across three mountain ranges. Most populations are isolated in the headwaters of streams or tributaries due to the extensive distribution of predatory nonnative trout in historical habitat; thus, it exists in a highly fragmented environment. Such isolation and fragmentation followed by the prevention of successful recolonization increases the potential for extirpation of the remaining populations (USFWS 2018).

<u>*Threats*</u>

Threats to this species include:

- Recreational foot traffic in naturally stressed Sierra Nevada ecosystems like riparian areas tramples the vegetation, compacts the soils, and can physically damage the streambanks (78 FR 24471).
- The presence of trout from current and historical stocking for the maintenance of a sport fishery is documented to have a significant detrimental impact to mountain yellow-legged frog populations. This anthropogenic activity has community-level effects and constitutes the primary detrimental impact to mountain yellow-legged frog habitat and species viability.
- Numerous reservoirs, dams, and water diversions have been constructed within the ranges of the

mountain yellow-legged frog complex and altered aquatic habitats in the Sierra Nevada. The combination of these features has reduced habitat suitability within the range of the species by creating migration barriers and altering local hydrology.

- Grazing reduces the suitability of habitat for mountain yellow-legged frogs by reducing its capability to sustain frogs and facilitate dispersal and migration, especially in stream areas.
- The impact of this stressor to mountain yellow-legged frogs is ongoing, but of relatively low importance as a limiting factor on extant populations, although this stressor may have played a greater role historically.
- Packstock grazing is the only grazing currently permitted in the National Parks of the Sierra Nevada (78 FR 24471).
- Activities that alter the terrestrial environment (such as road construction and timber harvest) may impact amphibian populations in the Sierra Nevada (78 FR 24471).
- Mountain yellow-legged frogs are generally found at high elevations in wilderness areas and National Parks where vegetation is sparse and fire suppression activities are infrequently implemented. Where such activities may occur, potential impacts to the species resulting from fire management activities include habitat degradation through water drafting (taking of water) from occupied ponds and lakes; erosion and siltation of habitat from construction of fuel breaks; and contamination by fire retardants from chemical fire suppression.
- The most prominent predator of mountain yellow-legged frogs is introduced trout, whose significance is well-established because it has been repeatedly observed that nonnative fishes and frogs rarely coexist; and it is known that introduced trout can and do prey on all frog life stages.
- Over roughly the last 2 decades, pathogens have been associated with amphibian population declines, mass die-offs, and even extinctions worldwide. One pathogen strongly associated with dramatic declines on all five continents is the chytrid fungus, *Batrachochytrium dendrobatidis* (Bd) (78 FR 24471).
- In the Sierra Nevada ecoregion, climate models predict temperature change (warming), which would result in warmer winters, earlier spring snowmelt, and higher summer temperatures; this in turn would lead to higher winter streamflows, earlier runoff, and reduced spring and summer streamflows, with increasing severity in the southern Sierra Nevada. Climate change represents a substantial future threat to the persistence of mountain yellow-legged frog populations (78 FR 24471).
- Remaining populations for both the Sierra Nevada yellow-legged frog and the mountain yellow-legged frog are small in many localities (78 FR 24471).

Five-Year Status Review

No five-year status review has been assessed for the mountain yellow-legged frog, northern DPS. On January 27, 2020, the U.S. Fish and Wildlife Service issued a notice of initiation of 5-year status reviews of 66 species in California and Nevada under the Endangered Species Act, which includes the mountain yellow-legged frog.

Critical Habitat

On August 6, 2016, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the Sierra Nevada yellow-legged frog (*Rana sierrae*), the northern distinct population segment (DPS) of the mountain yellow-legged frog (*Rana muscosa*), and the Yosemite toad (*Anaxyrus canorus*) under the Endangered Species Act of 1973, as amended (Act) (81 FR 59045). There is significant overlap in the critical habitat designations for these three species. The designated area, taking into account overlap in the critical habitat designations for these three species, is in total approximately 733,357 hectares (ha) (1,812,164 acres (ac)) in Alpine, Amador, Calaveras, El Dorado, Fresno, Inyo, Lassen, Madera, Mariposa, Mono, Nevada, Placer, Plumas, Sierra, Tulare, and Tuolumne counties, California (81 FR 59045). All critical habitat units and subunits are occupied by the respective species.

Two units and seven subunits are designated as critical habitat for the northern DPS of the mountain yellow-legged frog. Units are named after the major genetic clades (Vredenburg et al. 2007, p. 361), of which three exist rangewide for the mountain yellow-legged frog, and two are within the northern DPS of the mountain yellow-legged frog in the Sierra Nevada. Distinct units within each clade are designated as subunits.

Unit 4: Northern DPS of the Mountain Yellow-Legged Frog Clade 4. This unit represents a significant portion of the northern DPS of the mountain yellow-legged frog's range and reflects a core conservation area comprising the most robust remaining populations at higher densities (closer proximity) across the species' range. Unit 4, including all subunits, is an essential component to the entirety of this critical habitat designation due to the unique genetic and distributional area this unit encompasses. The frog populations within Clade 4 of the northern DPS of the mountain yellow-legged frog distribution face significant threats from habitat fragmentation. The critical habitat within the unit is necessary to sustain viable populations within Clade 4 northern DPS of the mountain yellow-legged frog, which are at very low abundances. Unit 4 is crucial to the species for range expansion and recovery. In addition, Clade 4 includes the only remaining basins with high-density, lake-based populations that are not infected with Bd, and Bd will likely invade these uninfected populations in the near future unless habitat protections and special management considerations are implemented. It is necessary to broadly protect remnant habitat across the range of Clade 4 to facilitate species persistence and recovery.

Subunit 4A: Frypan Meadows. The Frypan Meadows subunit consists of approximately 1,585 ha (3,917 ac), and is located in Fresno County, California, approximately 4.3 km (2.7 mi) northwest of Highway 180. The Frypan Meadows subunit consists entirely of Federal land, located predominantly within the boundaries of the Kings Canyon National Park, with some overlap into the Monarch Wilderness within the Sequoia National Forest. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Frypan Meadows subunit may require special management considerations or protection due to fish persistence.

Subunit 4B: Granite Basin. The Granite Basin subunit consists of approximately 1,777 ha (4,391 ac), and is located in Fresno County, California, approximately 3.2 km (2 mi) north of Highway 180. The Granite Basin subunit consists entirely of Federal land, located within the boundaries of the Kings Canyon National Park. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Granite Basin subunit may require special management considerations or protection due to fish persistence.

Subunit 4C: Sequoia Kings. The Sequoia Kings subunit consists of approximately 67,566 ha (166,958 ac), and is located in Fresno, Inyo and Tulare counties, California, approximately 18 km (11.25 mi) west of Highway 395 and 4.4 km (2.75 mi) southeast of Highway 180. The Sequoia Kings subunit consists entirely of Federal land, all within Sequoia and Kings Canyon National Parks. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Sequoia Kings subunit may require special management considerations or protection due to the presence of introduced fishes and fish persistence.

Subunit 4D: Kaweah River. The Kaweah River subunit consists of approximately 3,663 ha (9,052 ac), and is located in Tulare County, California, approximately 2.8 km (1.75 mi) east of Highway 198. The Kaweah River subunit consists entirely of Federal land, all within Sequoia National Park. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Kaweah River subunit may require special management considerations or protection due to fish persistence.

Unit 5: Northern DPS of the Mountain Yellow-Legged Frog Clade 5. This unit represents the southern portion of the species' range and reflects unique ecological features within the range of the species because it comprises populations that are stream-based. Unit 5, including all subunits, is an essential component of the entirety of this critical habitat designation due to the unique genetic and distributional area this unit encompasses. The frog populations within Clade 5 of the northern DPS of the mountain yellow-legged frog's distribution are at very low numbers and face significant threats from habitat fragmentation. The critical habitat within the unit is necessary to sustain viable populations within Clade 5 of the northern DPS of the mountain yellow-legged frog, which are at very low abundances. Unit 5 is crucial to the species for range expansion and recovery.

Subunit 5A: Blossom Lakes. The Blossom Lakes subunit consists of approximately 2,069 ha (5,113 ac), and is located in Tulare County, California, approximately 0.8 km (0.5 mi) northwest of Silver Lake. The Blossom Lakes subunit consists entirely of Federal land, located within Sequoia National Park and Sequoia National Forest. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Blossom Lakes subunit may require special management considerations or protection due to fish persistence.

Subunit 5B: Coyote Creek. The Coyote Creek subunit consists of approximately 9,802 ha (24,222 ac), and is located in Tulare County, California, approximately 7.5 km (4.7 mi) south of Moraine Lake. Land ownership within this subunit consists of approximately 9,792 ha (24,197 ac) of Federal land and 10 ha (24 ac) of private land. The Coyote Creek subunit is predominantly within Sequoia National Park and Sequoia and Inyo National Forests, including area within the Golden Trout Wilderness. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Coyote Creek subunit may require special management considerations or protection due to the presence of introduced fishes and recreational activities.

Subunit 5C: Mulkey Meadows. The Mulkey Meadows subunit consists of approximately 3,175 ha (7,846 ac), and is located in Tulare and Inyo counties, California, approximately 10 km (6.25 mi) west of Highway 395. The Mulkey Meadows subunit consists entirely of Federal land, all within the Inyo

National Forest, including area within the Golden Trout Wilderness. This subunit is considered to be within the geographical area occupied by the species at the time of listing, and it contains the physical or biological features essential to the conservation of the species, is currently functional habitat sustaining frogs, and is needed to provide for core surviving populations and their unique genetic heritage. The physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog in the Mulkey Meadows subunit may require special management considerations or protection due to the presence of introduced fishes, inappropriate grazing activity, and recreational activities.

<u>*Primary Constituent Elements*</u>

Critical habitat units are designated for Fresno, Inyo and Tulare counties, California. Within these areas, the primary constituent elements of the physical or biological features essential to the conservation of the northern DPS of the mountain yellow-legged frog consist of:

(i) Aquatic habitat for breeding and rearing. Habitat that consists of permanent water bodies, or those that are either hydrologically connected with, or close to, permanent water bodies, including, but not limited to, lakes, streams, rivers, tarns, perennial creeks (or permanent plunge pools within intermittent creeks), pools (such as a body of impounded water contained above a natural dam), and other forms of aquatic habitat. This habitat must: (A) For lakes, be of sufficient depth not to freeze solid (to the bottom) during the winter (no less than 1.7 meters (m) (5.6 feet (ft)), but generally greater than 2.5 m (8.2 ft), and optimally 5 m (16.4 ft) or deeper (unless some other refuge from freezing is available)). (B) Maintain a natural flow pattern, including periodic flooding, and have functional community dynamics in order to provide sufficient productivity and a prey base to support the growth and development of rearing tadpoles and metamorphs. (C) Be free of introduced predators. (D) Maintain water during the entire tadpole growth phase (a minimum of 2 years). During periods of drought, these breeding sites may not hold water long enough for individuals to complete metamorphosis, but they may still be considered essential breeding habitat if they provide sufficient habitat in most years to foster recruitment within the reproductive lifespan of individual adult frogs. (E) Contain: (1) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel, cobble, rock, and boulders (for basking and cover); (2) Shallower microhabitat with solar exposure to warm lake areas and to foster primary productivity of the food web; (3) Open gravel banks and rocks or other structures projecting above or just beneath the surface of the water for adult sunning posts; (4) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, or rocks and vegetation to provide cover from predators; and (5) Sufficient food resources to provide for tadpole growth and development.

(ii) Aquatic nonbreeding habitat (including overwintering habitat). This habitat may contain the same characteristics as aquatic breeding and rearing habitat (often at the same locale), and may include lakes, ponds, tarns, streams, rivers, creeks, plunge pools within intermittent creeks, seeps, and springs that may not hold water long enough for the species to complete its aquatic lifecycle. This habitat provides for shelter, foraging, predator avoidance, and aquatic dispersal of juvenile and adult mountain yellow-legged frogs. Aquatic nonbreeding habitat contains: (A) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel, cobble, rock, and boulders (for basking and cover); (B) Open gravel banks and rocks projecting above or just beneath the surface of the water for adult sunning posts; (C) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, or rocks and vegetation to provide cover from predators; (D) Sufficient food resources to support juvenile and adult foraging; (E) Overwintering refugia, where thermal properties of the microhabitat protect hibernating life stages from winter freezing, such as crevices or holes within bedrock, in and near shore; and/or (F) Streams, stream reaches, or wet meadow habitats that can function as corridors for movement between aquatic habitats used as breeding or foraging sites.

(iii) Upland areas. (A) Upland areas adjacent to or surrounding breeding and nonbreeding aquatic habitat that provide area for feeding and movement by mountain yellow-legged frogs. (1) For stream habitats, this area extends 25 m (82 ft) from the bank or shoreline. (2) In areas that contain riparian habitat and upland vegetation (for example, mixed conifer, ponderosa pine, montane conifer, and montane riparian woodlands), the canopy overstory should be sufficiently thin (generally not to exceed 85 percent) to allow sunlight to reach the aquatic habitat and thereby provide basking areas for the species. (3) For areas between proximate (within 300 m (984 ft)) water bodies (typical of some high mountain lake habitats), the upland area extends from the bank or shoreline between such water bodies. (4) Within mesic habitats such as lake and meadow systems, the entire area of physically contiguous or proximate habitat is suitable for dispersal and foraging. (B) Upland areas (catchments) adjacent to and surrounding both breeding and nonbreeding aquatic habitat that provide for the natural hydrologic regime (water quantity) of aquatic habitats. These upland areas should also allow for the maintenance of sufficient water quality to provide for the various life stages of the frog and its prey base.

Recovery Plan Information

There is no Recovery Plan for the Sierra Nevada yellow-legged frog at this time.

Recovery Actions

Need to develop recovery actions and Recovery Plan.

Environmental Baseline

The mountain yellow-legged frog and its designated critical habitat occur in the Sierra Nevada, California. Please refer to information above for the environmental baseline.

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Santa Cruz Long-toed Salamander (*Ambystoma macrodactylum croceum*)

Listing Status

The Santa Cruz long-toed salamander was federally listed as endangered on March 11, 1967, under the Endangered Species Preservation Act of 1966 (Service 1967).

Life History and Habitat

The Santa Cruz long-toed salamander utilizes terrestrial and aquatic habitats during the course of its lifecycle. Terrestrial habitats include upland mesic coastal scrub and woodland areas of coast live oak (*Quercus agrifolia*) or Monterey pine (*Pinus radiata*) and riparian vegetation, such as arroyo willows (*Salix lasiolepis*). The Santa Cruz long-toed salamander spends most of its life underground in burrows of small mammals, under leaf litter, rotten logs, fallen branches, and among the root systems of trees. Santa Cruz long-toed salamanders breed in shallow, usually ephemeral, freshwater ponds. Some breeding sites are ephemeral, while others contain water throughout the year (Boone et al. 2002).

Population Status

Prior to large-scale urbanization and conversion of lands for agricultural uses, it is probable that suitable upland sheltering and dispersal habitats were more widespread and contiguous in Santa Cruz and Monterey counties. Similarly, freshwater marshes and vernal pools likely occurred in greater abundance, in comparison to the present. Terrestrial and aquatic habitats suitable for Santa Cruz long-toed salamanders have been removed and altered due to urbanization and agricultural activities, and barriers to dispersal have been created, resulting in subpopulations which are isolated from each other. The likelihood of recolonization from other sites if a local extinction occurs is low because of habitat fragmentation. Additionally, population studies have been completed only sporadically since the time of listing, and only at 11 of the known breeding locations. The lack of population and genetic studies at the majority of these locations leaves little knowledge on breeding and recruitment success at each site, as well as whether genetic exchange between subpopulations is occurring (Service 2009).

Critical Habitat

Critical habitat has not been designated for this species.

Recovery Plan Information

The Draft Revised Recovery Plan for the Santa Cruz Long-Toed Salamander was published by the Service in April of 1999 (Service 1999). As stated in the recovery plan, due to the salamander's limited distribution, relatively small population sizes, and the dynamic nature of its habitats, all populations warrant protection and appropriate management. The goal of the recovery plan is to protect and enhance the long-term viability of all extant populations.

Environmental Baseline

The species only occurs within the State of California, please refer to the information above regarding the species environmental baseline.

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Sierra Nevada Yellow-legged Frog (*Rana sierrae*) and its Critical Habitat

Listing Status

The Sierra Nevada yellow-legged frog was listed as endangered, effective on June 30, 2014 (79 FR 24256). Critical habitat was designated for the Sierra Nevada yellow-legged frog on August 26, 2016 (81 FR 59045).

Life History and Habitat

Sierra Nevada yellow-legged frogs currently exist in montane regions of the Sierra Nevada of California in lakes, ponds, marshes, meadows, and streams at elevations ranging from 1,370 to 3,660 m (4,500 to 12,000 ft.). Sierra Nevada yellow-legged frogs are highly aquatic, are generally not found more than 1 m (3.3 ft.) from water (78 FR 24471; CDFG 2011), and display strong site fidelity, returning to the same overwintering and summer habitats from year to year (78 FR 24471). Both adult and tadpole Sierra Nevada yellow-legged frogs overwinter for up to 9 months in the bottoms of lakes that are at least 1.7 m (5.6 ft.) deep; however, overwinter survival may be greater in lakes that are at least 2.5 m (8.2 ft.) deep (78 FR 24471). Where water depths range from 0.2 m (0.7 ft.) to 1.5 m (5 ft.), the availability of rock crevices, holes, and ledges near shore offer protection to overwintering frogs when water bodies freeze over completely (78 FR 24471).

Adults are typically found sitting on rocks along the shoreline, usually where there is little or no vegetation. Although Sierra Nevada yellow-legged frogs may use a variety of shoreline habitats, both tadpoles and adults are less common at shorelines that drop abruptly to a depth of 60 cm (2 ft.) than at open shorelines that gently slope up to shallow waters of only 5 to 8 cm (2 to 3 in.) in depth (78 FR 24471). At lower elevations within their historical range, these species are known to be associated with rocky streambeds and wet meadows surrounded by coniferous forest. Streams used by adults vary from streams having high gradients and numerous pools, rapids, and small waterfalls to streams with low gradients and slow flows, marshy edges, and sod banks. Aquatic substrates vary from bedrock to fine sand, rubble (rock fragments), and boulders. Sierra Nevada yellow-legged frogs to be appear absent from the smallest creeks, probably because these creeks have insufficient depth for adequate refuge and overwintering habitat. Sierra Nevada yellow-legged frogs do use stream habitats, especially the remnant populations in the northern part of their range. At higher elevations, these species occupy lakes, ponds, tarns (small steep banked mountain lakes or pools), and streams. Sierra Nevada yellow-legged frogs in the Sierra Nevada are most abundant in high-elevation lakes and slow-moving portions of streams. The borders of alpine (above the tree line) lakes and mountain meadow streams used by mountain yellow-legged frogs are frequently grassy or muddy. This differs from the sandy or rocky shores inhabited by Sierra Nevada yellow-legged frogs in lower elevation streams.

Movements are typically localized, consisting of dispersal between selected breeding, feeding, and overwintering habitats during the course of a year, but can also lead to the re-colonization of sites where frogs have been extirpated previously. In aquatic habitats of high mountain lakes, Sierra Nevada yellow-legged frog adults typically move only a few hundred meters (few hundred yards), but single-season distances of up to 3.3 km (2.05 mi.) have been recorded along streams (78 FR 24471). Regular overland movements of more than 66 m (217 ft.) have been recorded, with individuals ranging as far 400 m (1,300 ft.) from water. During the overwintering period, adults have been observed along stream habitats more than 22 m (71 ft.) from the water (78 FR 24471; CDFG 2011). Regionally, Sierra Nevada yellow-legged frogs are thought to exhibit a metapopulation structure; metapopulations are spatially separated population subunits within migratory distance of one another, allowing individuals to interbreed among subunits and populations to become reestablished if they are extirpated (78 FR 24471).

Adults emerge from overwintering sites at spring thaw or snowmelt and commence breeding soon thereafter—between April and May at lower elevations and progressively later (June and July) at higher elevations (CDFG 2011). Eggs are deposited underwater in the shallows of ponds or in inlet streams in clusters, and are attached to rocks, gravel, or vegetation, or deposited under banks. Because tadpoles must overwinter multiple years before metamorphosis, successful breeding sites are located in (or connected to) lakes and ponds that do not dry out in the summer, and also are deep enough that they do not completely freeze or become oxygen-depleted (anoxic) in winter. The eggs are deposited in globular clumps, which are often somewhat flattened and roughly 2.5 to 5 cm (1 to 2 in.) in diameter. Clutch size varies from 15 to 350 eggs per egg mass. Egg hatching time ranges from 16 to 21 days at temperatures of 5 to 13.5 degrees Celsius (41 to 56 degrees Fahrenheit). The time required to reach reproductive maturity in Sierra Nevada yellow-legged frogs is thought to vary between 3 and 4 years post-metamorphosis. In combination with the extended amount of time as a tadpole before metamorphosis, it may take 5 to 8 years for Sierra Nevada yellow-legged frogs to begin reproducing (78 FR 24471; CDFG 2011). The longevity of adults is unknown, but adult survivorship from year to year is very high under normal circumstances. Sierra Nevada yellow- legged frogs are presumed to be long-lived amphibians (78 FR 24471; CDFG 2011).

<u>Juvenile</u>: Sierra Nevada yellow-legged frogs are omnivorous, feeding as tadpoles on algae, diatoms, and detritus. Tadpoles forage for prey at the bottoms of lakes, ponds, and streams, in shallow waters. During winter, tadpoles remain in warmer water below the thermocline; in the spring, when warmer days raise surface water temperatures, they move to shallow, near-shore water, retreating during the late afternoon and evening to offshore waters that are less subject to night cooling (78 FR 24471; CDFG 2011). Tadpoles may take more than 1 year, and often require 2 to 4 years, to reach metamorphosis (transformation from tadpoles to frogs), depending on local climate conditions and site-specific variables (78 FR 24471; CDFG 2011).

<u>Adult</u>: Sierra Nevada yellow-legged frogs are omnivorous; adult diet consists of terrestrial and aquatic insects and macro invertebrates, other amphibians, and the occasional cannibalism of eggs and tadpole/adult carcasses. Adults forage for prey at the bottoms of lakes, ponds, and streams; in shallow waters; and onshore. As adults, frogs maximize body temperatures during a majority of the day by basking in the sun, moving between water and land, and concentrating in the warmer shallows along the shoreline. As temperatures decrease in the fall, frogs become less active and move to overwintering habitats (78 FR 24471; CDFG 2011). With the widespread introduction of nonnative trout, nearly all large, deep lakes that could provide suitable overwintering habitat for frogs are now occupied by introduced trout. In addition to their role as predators of Sierra Nevada yellow-legged frogs, trout are competitors for the same invertebrate species that frogs rely on for food. The direct impacts of trout predation on invertebrates can have a negative effect on frogs via competition for invertebrate prey; and can alter lake nutrient cycles, resulting in negative impacts to frogs and other native species (CDFG 2011).

Population Status

<u>Rangewide Status of the Species</u>

Sierra Nevada yellow-legged frogs were historically abundant across much of the higher elevations of the Sierra Nevada. The precise historical ranges of the Sierra Nevada yellow-legged frog and the mountain yellow-legged frog are difficult to determine, because projections must be inferred from museum collections that do not reflect systematic surveys; and historic survey information is very limited. Sierra Nevada yellow-legged frogs occupy the western Sierra Nevada north of the Monarch Divide (in Fresno County) and the eastern Sierra Nevada (east of the crest) in Inyo and Mono counties. The northern DPS of the mountain yellow-legged frog extends in the western Sierra Nevada from south of the Monarch Divide in Fresno County through portions of the Kern River drainage; the southern DPS of the mountain yellow-legged frog occupies the canyons of the Transverse Ranges in southern California. The ranges of the two frog species in the mountain yellow-legged complex therefore meet each other roughly along the Monarch Divide to the north, and along the crest of the Sierra Nevada to the east (78 FR 24471; CDFG 2011).

Currently, the mountain yellow-legged frog complex exists in montane regions of the Sierra Nevada of California at elevations ranging from 1,370 to 3,660 meters (m) (4,500 to 12,000 feet [ft.]). Sierra Nevada yellow-legged frogs occupy the western Sierra Nevada north of the Monarch Divide (in Fresno County) and the eastern Sierra Nevada (east of the crest) in Inyo and Mono counties. Researchers have reported disappearances of these species from a large fraction of their historical ranges in the Sierra Nevada, with their distributions currently restricted primarily to publicly-managed lands at high elevations, including streams, lakes, ponds, and meadow wetlands in National Forests and National Parks. The most pronounced declines in the mountain yellow-legged frog complex have occurred north of Lake Tahoe in the northernmost 125-kilometer (km) (78-mile [mi.]) portion of the range (Sierra Nevada yellow-legged frog), and south of Sequoia and Kings Canyon National Parks in Tulare County, in the southernmost 50-km (31- mi.) portion, where only a few populations of the northern DPS of the mountain yellow-legged frog remain. Mountain yellow-legged frog populations have persisted in greater density in the National Parks of the Sierra Nevada than in the surrounding U.S. Forest Service (USFS) lands, and the populations that do occur in the National Parks generally exhibit greater abundances than those on USFS lands. Currently, the northern DPS of the mountain yellow-legged frog is discrete from the southern DPS because it is separated from the southern frogs by a 225-km (140-mi.) barrier of unsuitable habitat (78 FR 24471; 79 FR 24255; CDFG 2011).

Population Summary

Monitoring efforts and research studies have documented substantial declines of populations of the mountain yellow-legged frog complex in the Sierra Nevada. The number of extant populations has declined greatly over the last few decades. Remaining populations are patchily scattered throughout the historical range. Documented extirpations appear to occur nonrandomly across the landscape, are typically spatially clumped, and involve the disappearance of all or nearly all of the populations of the mountain yellow-legged frog complex in a watershed. Over the available historical record, estimated losses range from 69 to 93 percent. Range-wide reduction has diminished the number of watersheds that support mountain yellow-legged frogs (*R. sierrae*), at a conservative estimate of 59 percent. Remaining populations are much smaller than historical norms, and the density of populations per watershed has declined substantially; as a result, many watersheds currently support single metapopulations at low abundances. Remaining populations are generally very small, and available information indicates that the rates of population decline have not abated, and they have likely accelerated during the 1990s into the 2000s (79 FR 24255). Extensive surveys between 1995 and 2005 yielded only 11 occupied sites, and population size estimates range from 1,000 to 10,000 individuals (NatureServe 2015).

Threats

Threats to this species include:

- Recreational foot traffic in naturally stressed Sierra Nevada ecosystems like riparian areas tramples the vegetation, compacts the soils, and can physically damage the streambanks (78 FR 24471).
- Trout both compete for limited resources and directly prey on Sierra Nevada yellow-legged frog tadpoles and adults. These fish decimate frog populations through competition and predation,

leading to the isolation of populations and preventing recolonization by frogs. Fundamentally, this has prevented deeper lakes from serving as Sierra Nevada yellow-legged frog habitat at a landscape scale (78 FR 24471).
- Numerous reservoirs, dams, and water diversions have been constructed within the ranges of the Sierra Nevada yellow-legged frog complex and altered aquatic habitats in the Sierra Nevada. The combination of these features has reduced habitat suitability within the range of the species by creating migration barriers and altering local hydrology (78 FR 24471).
- Grazing reduces the suitability of habitat for Sierra Nevada yellow-legged frogs by reducing its capability to sustain frogs and facilitate dispersal and migration, especially in stream areas. The impact of this stressor to Sierra Nevada yellow-legged frogs is ongoing, but of relatively low importance as a limiting factor on extant populations, although this stressor may have played a greater role historically (78 FR 24471).
- Packstock grazing is the only grazing currently permitted in the National Parks of the Sierra Nevada (78 FR 24471).
- Activities that alter the terrestrial environment (such as road construction and timber harvest) may impact amphibian populations in the Sierra Nevada (78 FR 24471).
- Sierra Nevada yellow-legged frogs are generally found at high elevations in wilderness areas and National Parks where vegetation is sparse and fire suppression activities are infrequently implemented. Where such fire management activities occur, potential impacts that may result include habitat degradation through water drafting (taking of water) from occupied ponds and lakes; erosion and siltation of habitat from construction of fuel breaks; and contamination by fire retardants from chemical fire suppression (78 FR 24471).
- The most prominent predator of Sierra Nevada yellow-legged frogs is introduced trout, whose significance is well-established because it has been repeatedly observed that nonnative fishes and frogs rarely coexist; and it is known that introduced trout can and do prey on all frog life stages (78 FR 24471).
- Over roughly the last 2 decades, pathogens have been associated with amphibian population declines, mass die-offs, and even extinctions worldwide. One pathogen strongly associated with dramatic declines on all five continents is the chytrid fungus, *Batrachochytrium dendrobatidis* (Bd) (78 FR 24471).
- In the Sierra Nevada ecoregion, climate models predict temperature change (warming), which would result in warmer winters, earlier spring snowmelt, and higher summer temperatures; this in turn would lead to higher winter streamflows, earlier runoff, and reduced spring and summer streamflows, with increasing severity in the southern Sierra Nevada. Climate change represents a substantial future threat to the persistence of Sierra Nevada yellow-legged frog populations (78 FR 24471).
- Remaining populations for the Sierra Nevada yellow-legged frog are small in many localities. Small population size is currently a significant threat to most populations of Sierra Nevada yellow-legged frogs across the range of the species (78 FR 24471).

Five-Year Status Review

Currently, there are no five-year status reviews for this species. On February 10, 2020, the USFWS initiated a 5-year status reviews of 66 species in California and Nevada, including the Sierra Nevada yellow-legged frog.

Critical Habitat

On September 26, 2016, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the Sierra Nevada yellow-legged frog (*Rana sierrae*), the northern distinct population segment (DPS) of the mountain yellow-legged frog (*Rana muscosa*), and the Yosemite toad (*Anaxyrus canorus*) under the Endangered Species Act of 1973, as amended (Act) (81 FR 59045). There is significant overlap in the critical habitat designations for these three species. The designated area, taking into account overlap in the critical habitat designations for these three species, is in total approximately 733,357 hectares (ha) (1,812,164 acres) in Alpine, Amador, Calaveras, El Dorado, Fresno, Inyo, Lassen, Madera, Mariposa, Mono, Nevada, Placer, Plumas, Sierra, Tulare, and Tuolumne counties, California (81 FR 59045). All critical habitat units and subunits are occupied by the respective species.

437,929 ha (1,082,147 acres) are designated as critical habitat for the Sierra Nevada yellow-legged frog. This area represents approximately 18 percent of the historical range of the species as estimated by Knapp (unpublished data). All subunits designated as critical habitat are considered occupied (at the subunit level) and include lands within Lassen, Plumas, Sierra, Nevada, Placer, El Dorado, Amador, Calaveras, Alpine, Tuolumne, Mono, Mariposa, Madera, Fresno, and Inyo counties, California. Three units encompassing 24 subunits are designated as critical habitat for the Sierra Nevada yellow-legged frog (81 FR 59045).

Critical habitat units are designated for Lassen, Plumas, Sierra, Nevada, Placer, El Dorado, Amador, Alpine, Calaveras, Tuolumne, Mono, Mariposa, Madera, Fresno, and Inyo counties, California (81 FR 59045). Within these areas, the primary constituent elements of the physical or biological features essential to the conservation of the Sierra Nevada yellow-legged frog consist of:

(i) Aquatic habitat for breeding and rearing. Habitat that consists of permanent water bodies, or those that are either hydrologically connected with, or close to, permanent water bodies, including, but not limited to, lakes, streams, rivers, tarns, perennial creeks (or permanent plunge pools within intermittent creeks), pools (such as a body of impounded water contained above a natural dam), and other forms of aquatic habitat. This habitat must: (A) For lakes, be of sufficient depth not to freeze solid (to the bottom) during the winter (no less than 1.7 meters (m) (5.6 feet (ft)), but generally greater than 2.5 m (8.2 ft), and optimally 5 m (16.4 ft) or deeper (unless some other refuge from freezing is available)). (B) Maintain a natural flow pattern, including periodic flooding, and have functional community dynamics in order to provide sufficient productivity and a prey base to support the growth and development of rearing tadpoles and metamorphs. (C) Be free of introduced predators. (D) Maintain water during the entire tadpole growth phase (a minimum of 2 years). During periods of drought, these breeding sites may not hold water long enough for individuals to complete metamorphosis, but they may still be considered essential breeding habitat if they provide sufficient habitat in most years to foster recruitment within the reproductive lifespan of individual adult frogs. (E) Contain: (1) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel, cobble, rock, and boulders (for basking and cover); (2) Shallower microhabitat with solar exposure to warm lake areas and to foster primary productivity of the food web; (3) Open gravel banks and rocks or other structures projecting above or just beneath the surface of the water for adult sunning posts; (4) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, or rocks and vegetation to provide cover from predators; and (5) Sufficient food resources to provide for tadpole growth and development.

(ii) Aquatic nonbreeding habitat (including overwintering habitat). This habitat may contain the same characteristics as aquatic breeding and rearing habitat (often at the same locale), and may include lakes, ponds, tarns, streams, rivers, creeks, plunge pools within intermittent creeks, seeps, and springs that may not hold water long enough for the species to complete its aquatic lifecycle. This habitat provides for shelter, foraging, predator avoidance, and aquatic dispersal of juvenile and adult mountain yellow-legged frogs. Aquatic nonbreeding habitat contains: (A) Bank and pool substrates consisting of varying percentages of soil or silt, sand, gravel, cobble, rock, and boulders (for basking and cover); (B) Open gravel banks and rocks projecting above or just beneath the surface of the water for adult sunning posts; (C) Aquatic refugia, including pools with bank overhangs, downfall logs or branches, or rocks and vegetation to provide cover from predators; (D) Sufficient food resources to support juvenile and adult foraging; (E) Overwintering refugia, where thermal properties of the microhabitat protect hibernating life stages from winter freezing, such as crevices or holes within bedrock, in and near shore; and/or (F) Streams, stream reaches, or wet meadow habitats that can function as corridors for movement between aquatic habitats used as breeding or foraging sites.

(iii) Upland areas. (A) Upland areas adjacent to or surrounding breeding and nonbreeding aquatic habitat that provide area for feeding and movement by mountain yellow-legged frogs. (1) For stream habitats, this area extends 25 m (82 ft) from the bank or shoreline. (2) In areas that contain riparian habitat and upland vegetation (for example, mixed conifer, ponderosa pine, montane conifer, and montane riparian woodlands), the canopy overstory should be sufficiently thin (generally not to exceed 85 percent) to allow sunlight to reach the aquatic habitat and thereby provide basking areas for the species. (3) For areas between proximate (within 300 m (984 ft)) water bodies (typical of some high mountain lake habitats), the upland area extends from the bank or shoreline between such water bodies. (4) Within mesic habitats such as lake and meadow systems, the entire area of physically contiguous or proximate habitat is suitable for dispersal and foraging. (B) Upland areas (catchments) adjacent to and surrounding both breeding and nonbreeding aquatic habitat that provide for the natural hydrologic regime (water quantity) of aquatic habitats. These upland areas should also allow for the maintenance of sufficient water quality to provide for the various life stages of the frog and its prey base.

Recovery Plan Information

There is no Recovery Plan for the Sierra Nevada yellow-legged frog at this time.

Recovery Actions

Need to develop recovery actions and Recovery Plan.

Environmental Baseline

The Sierra Nevada yellow-legged frog and its designated critical habitat occur in the Sierra Nevada, California. Please refer to information above for the environmental baseline.

Literature Cited

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and the Yosemite Toad. Final Rule. Vol 81, No. 166. Federal Register 59045. August 26, 2016. Available online at: https://www.govinfo.gov/content/pkg/FR-2016-08-26/pdf/2016-20352.pdf#page=1

Western Spadefoot (*Spea hammondii*), Northern DPS

Listing Status

The western spadefoot is currently under federal review for listing under the Act. On December 4, 2023, the Service proposed to list two distinct population segments of the western spadefoot as threatened, the northern distinct population segment and the southern distinct population segment (88 FR 84252). A species status assessment was completed in May 2023 (Service 2023a), compiling biological information and conditions on both distinct population segments.

Life History and Habitat

The western spadefoot ranges in size from 1.5 to 2.5 inches snout to vent length (Stebbins and McGinnis 2012, p. 156). They are dusky green or gray on their backs and often have four irregular light-colored stripes, with the central pair of stripes sometimes distinguished by a dark, hourglass-shaped area; furthermore, spadefoot have yellow eyes with vertical pupils. Adult western spadefoot forage on a variety of small invertebrate prey. Stomach content examinations have found food that includes grasshoppers, true bugs, moths, ground beetles, predaceous diving beetles, ladybird beetles, click beetles, flies, ants, and earthworms (Morey and Guinn 1992, p. 155). The diet of western spadefoot larvae is unknown. However, the larvae of plains spadefoot (*Scaphiopus bombifrons*) consume planktonic organisms and algae, fairy shrimp, and will scavenge dead organisms, including other spadefoot larvae (Bragg 1962, p. 144; Bragg 1964, pp. 17–23). Adult, juvenile, and presumably larval western spadefoot consume food items that are also used by other co-occurring amphibians including: pacific tree frog (*Pseudacris regilla*), California tiger salamander (*Ambystoma californiense*), and western toad (*Anaxyrus boreas*) (Morey and Guinn 1992, p. 155).

Western spadefoots are primarily terrestrial and inhabit underground burrows. Western spadefoots can burrow up to 3 feet below ground (during the dry season to avoid temperature extremes and desiccation (Stebbins and McGinnis 2012, p. 157). Western spadefoots emerge from their burrows to forage and breed following seasonal rains in winter and spring (Dimmitt and Ruibal 1980, p. 21; Jennings and Hayes 1994, p. 94). Emergence is likely related to soil moisture at a depth of 10cm, air moisture, or a sound or vibration cue from the rain (Dimmitt and Ruibal 1980, p. 26; Halstead 2021). Most western spadefoot surface activity is nocturnal to reduce water loss. Little is known regarding the land surface types western spadefoot can traverse or the distances that western spadefoot may travel from aquatic resources for dispersal. A study looking at movement of western spadefoot individuals in an Orange County population found that the mean distance moved away from breeding pools was 131.36 feet (Baumberger 2013, p. 14), with the longest movement of an individual being 1,985 feet (Baumberger et al. 2020, p. 7).

Western spadefoot habitat is primarily open treeless grasslands, scrub, or mixed woodland and grassland where aquatic breeding habitat is available (Stebbins and McGinnis 2012, p. 157). Western spadefoot requires both aquatic and terrestrial habitat components in proximity to meet all life history requirements. Western spadefoots are primarily terrestrial and require upland habitats for feeding and for constructing burrows for long dry-season dormancy (Stebbins and McGinnis 2012, pp. 154–158).

Western spadefoots use aquatic habitat for breeding and developing larvae. Suitable aquatic habitat typically includes temporary vernal pools, sand or gravel washes, and small streams that are often seasonal (Stebbins and McGinnis 2012, p. 157). However, eggs and larvae of western spadefoot have been observed in a variety of permanent and temporary wetlands, both natural and altered, including rivers, creeks, artificial ponds, livestock ponds, sedimentation and flood control ponds, irrigation and roadside ditches, roadside puddles, tire ruts, and borrow pits, indicating a degree of ecological plasticity (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270). Although western spadefoot has been observed to inhabit and breed in wetlands altered or created by humans, survival and reproductive success in these pools have not been evaluated relative to that in unaltered natural pools. Temporary wetlands may be optimal aquatic breeding habitat due to reduced abundance of both native and nonnative predators, many of which require more permanent water sources (Jennings and Hayes 1994, p. 96; Stebbins and McGinnis 2012, p. 158).

Depending on temperature and annual rains, western spadefoot breeding, and oviposition occurs from October to May, most often in temporary pools and drainages from winter or spring rains (Stebbins 1985, p. 57). Age of sexual maturity is unknown but considering the relatively long period of subterranean dormancy (8 to 10 months), individuals may require at least 2 years to mature (Jennings and Hayes 1994, p. 94). Females deposit eggs in numerous, small, and irregular cylindrical clusters of 10 to 42 eggs, with an average of 24 eggs (Storer 1925, p. 157; Stebbins and McGinnis 2012, p. 156). Eggs range in size from 0.04 to 0.07 inches and are light olive-green or sooty on top and light colored on the bottom (Stebbins and McGinnis 2012, p. 156). Eggs hatch in 0.6 to 6 days depending on the temperature (Brown 1967, p. 747). Larval development can be completed in 3 to 11 weeks depending on food resources and temperature, and development must be completed before the pools dry (Burgess 1950, p. 49– 51; Feaver 1971, p. 53; Morey 1998, p. 86). Metamorphosing larvae may leave the water while their tails are still relatively long (0.4 inch) and move toward suitable terrestrial burrowing habitat (Storer 1925, p. 159). Metamorphic western spadefoots have been documented using desiccation crack microhabitat as refugia (Alvares and Kerrs 2023).

Population Status

The historical range of the northern distinct population segment of the western spadefoot is entirely in California. It includes the area of the Sacramento and San Joaquin Valleys from Shasta to Kern Counties including the lower elevation foothill areas of the Sierra Nevada Mountains and low-elevation and valley areas in the northern Coast Range from Tehama County south to Santa Clara County. In the southwest portion of the northern distinct population segment's range, the occupied area extends from southern Santa Cruz County to southern Santa Barbara County of the Coast Range and is contiguous with the Central Valley portion of the distinct population segment’s range. (88 FR 84252). They have been found at sites from sea level up to 4,500 feet in the Sierra Nevada foothills (Stebbins and McGinnis 2012, p. 157).

The northern distinct population segment of the western spadefoot is thought to be extirpated throughout many historical locations within the Central Valley (Stebbins 1985, p. 67; Jennings and Hayes 1994, p. 96). In the northern western spadefoot range, the largest declines have been observed in the Sacramento Valley and San Joaquin Valley, while declines in abundance have been more modest in the Coast Ranges (Fisher and Shaffer 1996, p. 1387). A species distribution model for the northern western spadefoot range (north of Santa Barbara) found the areas predicted to have suitable habitat are patchily distributed along the foothills surrounding the Central Valley and in the southwestern quarter of the northern western spadefoot range including the Salinas Valley (Rose et al. 2020, entire).

Recovery Plan Information

No recovery plan exists for the western spadefoot; however, western spadefoots are threatened by urbanization, road construction, off-road vehicular traffic, illegal dumping, livestock grazing, and other edge effects that degrade habitat quality. In southern California, within the southern western spadefoot clade, over 80 percent of the habitat once known to be occupied by western spadefoot has been developed or converted to uses that are incompatible with successful reproduction and recruitment (Jennings and Hayes 1994, p. 96). Development can directly destroy aquatic breeding pools and underground burrows, or it can alter the hydrology such that aquatic breeding pools may not form where a population once existed. Furthermore, overabundant vegetation reduces the quality of aquatic breeding pools by causing them to dry more quickly, which then has impacts on reproduction and abundance. Anthropogenic warming increases the overall likelihood of extreme droughts in California into the future, and drought decreases the quality and quantity of aquatic breeding pools available for western spadefoot (Williams et al. 2015, pp. 6819, 6826).

Activities that produce low frequency noise and vibration, such as grading for development and seismic exploration for natural gas, in or near habitat for western spadefoot, may be detrimental to the species. Other threats include burrow collapse or destruction from heavy equipment operation and ground disturbance activities. Disturbances that cause western spadefoot to emerge at inappropriate times could result in mortality or reduced fitness (Dimmitt and Ruibal 1980, pp. 27–28).

Nonnative predators may be predating on western spadefoot. Furthermore, nonnative predators may compete with western spadefoot for prey and habitat (Morey and Guinn 1992, p. 153).

Environmental Baseline

The Northern DPS of the western spadefoot occurs entirely within the state of California. Thus, the status description above also serves as the environmental baseline for this consultation.

Literature Cited

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Western Spadefoot (*Spea hammondii*), Southern DPS

Listing Status

The western spadefoot is currently under federal review for listing under the Act. On December 4, 2023, the Service proposed to list two distinct population segments of the western spadefoot as threatened, the northern distinct population segment and the southern distinct population segment (88 FR 84252). A species status assessment was issued in May 2023 (Service 2023a), compiling biological information and conditions on both distinct population segments.

Life History and Habitat

The western spadefoot ranges in size from 1.5 to 2.5 inches snout to vent length (Stebbins and McGinnis 2012, p. 156). They are dusky green or gray on their backs and often have four irregular light-colored stripes, with the central pair of stripes sometimes distinguished by a dark, hourglass-shaped area; furthermore, spadefoot have yellow eyes with vertical pupils. Adult western spadefoot forage on a variety of small invertebrate prey. Stomach content examinations have found food that includes grasshoppers, true bugs, moths, ground beetles, predaceous diving beetles, ladybird beetles, click beetles, flies, ants, and earthworms (Morey and Guinn 1992, p. 155). The diet of western spadefoot larvae is unknown. However, the larvae of plains spadefoot (*Scaphiopus bombifrons*) consume planktonic organisms and algae, fairy shrimp, and will scavenge dead organisms, including other spadefoot larvae (Bragg 1962, p. 144; Bragg 1964, pp. 17–23). Adult, juvenile, and presumably larval western spadefoot consume food items that are also used by other co-occurring amphibians including: pacific tree frog (*Pseudacris regilla*), California tiger salamander (*Ambystoma californiense*), and western toad (*Anaxyrus boreas*) (Morey and Guinn 1992, p. 155).

Western spadefoots are primarily terrestrial and inhabit underground burrows. Western spadefoots can burrow up to 3 feet below ground during the dry season to avoid temperature extremes and desiccation (Stebbins and McGinnis 2012, p. 157). Western spadefoots emerge from their burrows to forage and breed following seasonal rains in winter and spring (Dimmitt and Ruibal 1980, p. 21; Jennings and Hayes 1994, p. 94). Emergence is likely related to soil moisture at a depth of 10cm, air moisture, or a sound or vibration cue from the rain (Dimmitt and Ruibal 1980, p. 26; Halstead 2021). Most western spadefoot surface activity is nocturnal to reduce water loss. Little is known regarding the land surface types western spadefoot can traverse or the distances that western spadefoot may travel from aquatic resources for dispersal. A study looking at movement of western spadefoot individuals in an Orange County population found that the mean distance moved away from breeding pools was 131.36 feet (Baumberger 2013, p. 14), with the longest movement of an individual being 1,985 feet (Baumberger et al. 2020, p. 7).

Western spadefoot habitat is primarily open treeless grasslands, scrub, or mixed woodland and grassland where aquatic breeding habitat is available (Stebbins and McGinnis 2012, p. 157). Western spadefoot requires both aquatic and terrestrial habitat components in proximity to meet all life history requirements. Western spadefoots are primarily terrestrial and require upland habitats for feeding and for constructing burrows for long dry-season dormancy (Stebbins and McGinnis 2012, pp. 154–158).

Western spadefoots use aquatic habitat for breeding and developing larvae. Suitable aquatic habitat typically includes temporary vernal pools, sand or gravel washes, and small streams that are often seasonal (Stebbins and McGinnis 2012, p. 157). However, eggs and larvae of western spadefoot have been observed in a variety of permanent and temporary wetlands, both natural and altered, including rivers, creeks, artificial ponds, livestock ponds, sedimentation and flood control ponds, irrigation and roadside ditches, roadside puddles, tire ruts, and borrow pits, indicating a degree of ecological plasticity (Beever et al. 2016, p. 132; Nicotra et al. 2015, p. 1270). Although western spadefoot has been observed to inhabit and breed in wetlands altered or created by humans, survival and reproductive success in these pools have not been evaluated relative to that in unaltered natural pools. Temporary wetlands may be optimal aquatic breeding habitat due to reduced abundance of both native and nonnative predators, many of which require more permanent water sources (Jennings and Hayes 1994, p. 96; Stebbins and McGinnis 2012, p. 158).

Depending on temperature and annual rains, western spadefoot breeding, and oviposition occurs from October to May, most often in temporary pools and drainages from winter or spring rains (Stebbins 1985, p. 57). Age of sexual maturity is unknown but considering the relatively long period of subterranean dormancy (8 to 10 months), individuals may require at least 2 years to mature (Jennings and Hayes 1994, p. 94). Females deposit eggs in numerous, small, and irregular cylindrical clusters of 10 to 42 eggs, with an average of 24 eggs (Storer 1925, p. 157; Stebbins and McGinnis 2012, p. 156). Eggs range in size from 0.04 to 0.07 inches and are light olive-green or sooty on top and light colored on the bottom (Stebbins and McGinnis 2012, p. 156). Eggs hatch in 0.6 to 6 days depending on the temperature (Brown 1967, p. 747). Larval development can be completed in 3 to 11 weeks depending on food resources and temperature, and development must be completed before the pools dry (Burgess 1950, p. 49– 51; Feaver 1971, p. 53; Morey 1998, p. 86). Metamorphosing larvae may leave the water while their tails are still relatively long (0.4 inch) and move toward suitable terrestrial burrowing habitat (Storer 1925, p. 159). Metamorphic western spadefoots have been documented using desiccation crack microhabitat as refugia (Alvares and Kerrs 2023).

Population Status

The historical range of the southern distinct population segment of the western spadefoot includes portions of southern California and northwestern Baja California, Mexico (88 FR 84252). In California, the species occurred in valleys and low-lying areas of portions of the Coast Range from extreme southeastern Santa Barbara County south to Ventura, Los Angeles, San Bernardino, Orange, Riverside, and San Diego Counties. However, due to habitat loss and degradation, the species is now patchily distributed in southern California and mostly extirpated from the urbanized areas of Los Angeles and San Diego. Most remaining populations are isolated by habitat fragmentation resulting from land use conversion and urbanization (88 FR 84252).

Recovery Plan Information

No recovery plan exists for the western spadefoot; however, western spadefoots are threatened by urbanization, road construction, off-road vehicular traffic, illegal dumping, livestock grazing, and other edge effects that degrade habitat quality. In southern California, within the southern western spadefoot clade, over 80 percent of the habitat once known to be occupied by western spadefoot has been developed or converted to uses that are incompatible with successful reproduction and recruitment (Jennings and Hayes 1994, p. 96). Development can directly destroy aquatic breeding pools and underground burrows, or it can alter the hydrology such that aquatic breeding pools may not form where a population once existed. Furthermore, overabundant vegetation reduces the quality of aquatic breeding pools by causing them to dry more quickly, which then has impacts on reproduction and abundance. Anthropogenic warming increases the overall likelihood of extreme droughts in California into the future, and drought decreases the quality and quantity of aquatic breeding pools available for western spadefoot (Williams et al. 2015, pp. 6819, 6826).

Activities that produce low frequency noise and vibration, such as grading for development and seismic exploration for natural gas, in or near habitat for western spadefoot, may be detrimental to the species. Other threats include burrow collapse or destruction from heavy equipment operation and ground disturbance activities. Disturbances that cause western spadefoot to emerge at inappropriate times could result in mortality or reduced fitness (Dimmitt and Ruibal 1980, pp. 27–28).

Nonnative predators may be predating on western spadefoot. Furthermore, nonnative predators may compete with western spadefoot for prey and habitat (Morey and Guinn 1992, p. 153).

Environmental Baseline

The Southern DPS of the western spadefoot occurs throughout much of California, but also occurs in Baja California, Mexico. However, we have limited information regarding occurrences outside of California. Thus, the status description above also serves as the environmental baseline for this consultation.

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Yosemite Toad (*Anaxyrus canorus*) and its Critical Habitat

Listing Status

The Yosemite toad was listed as threatened on June 30, 2014 (79 FR 24256). Critical habitat was designated for this species on August 26, 2016 (81 FR 59045).

Life History and Habitat

Yosemite toads are found in moist environments that include meadows, edges of forest, grasslands, and shallow pools of water, and are often in sunny spots. Adults can be found in riparian habitats, shallow water, moist meadows, borders of forest, and grassland. Juveniles can be found in ponds, lakes, vernal pools, and slow-moving streams. Adults burrow in soil, leaf litter, and underground rodent burrows from October through April or May. Yosemite toads emerge from their burrows after the snow has melted. Tadpoles stay in shallow pools of water until metamorphosis is complete.

Yosemite toads are inactive during hot, dry, and cold weather due to a low tolerance of temperature ranges. Yosemite toads will burrow underground if it is too hot or too cold. If they are exposed to hot or freezing temperatures, it can cause death. Yosemite toads overwinter in underground burrows for 6 to 8 months (USDA et al. 2015).

Breeding for Yosemite toads occurs from May to July, depending on the snow melt. Males appear at the breeding pond a few days before females, and some defend a small breeding territory. Breeding occurs in shallow edges of pools, lakes, and slow-moving streams. The male climbs on the female's back and fertilizes the eggs as they are laid. Females lay 1,500 to 2,000 eggs, once every 2 to 4 years. Eggs are laid in clear, jelly-like strings. Occasionally, the water in the breeding site will evaporate before the eggs can hatch, causing death (Davidson et al. 2015, USFWS 2015). After hatching, tadpoles metamorphose within 5 to 7 weeks. There can be a high mortality rate with metamorphosis. Tadpoles are preyed upon, and pools of water can evaporate or freeze, which can cause death. Juveniles also can have high overwinter mortality rates (USDA et al. 2015).

Yosemite toads migrate to and from their breeding pond and nonbreeding habitat. Yosemite toads will locally migrate close distances to breeding ponds and further upland to nonbreeding locations where they can burrow and forage for food.

Adult Yosemite toads hunt for food in waterbodies as well as on land. Adults wait for an invertebrate to come to them, and then use their sticky tongue to capture it. Adults eat various small invertebrates such as flies, spiders, ants, and beetles (USFWS 2015). Tadpoles will graze for food at the bottom of shallow waterbodies. Tadpoles are mostly herbivorous, but will eat small organic detritus. Tadpoles also eat algae, zooplankton, and plant material (Davidson et al. 2006).

Population Status

Rangewide Status of the Species

Yosemite toads are endemic to California. Historically, Yosemite toads ranged from the Alpine County to Fresno County in areas above 1,980 to 3,414 meters (m) (6,300 to 11,380 feet [ft.]). The majority of the Yosemite toad population is found between 2,590 and 3,048 m (8,500 and 10,000 ft.). Areas where the toad was found included Grass Lake, Blue Lake, and Ebbetts Pass. Currently, the Yosemite toad is found in scattered locations throughout its historic range. Its current habitat covers only 50 percent of its historic range. Yosemite toads only occur in the Sierra Nevada (IUCN 2015).

Population Summary

Yosemite toad populations are declining; they now exist in only 50 percent of historically known sites, even in unaltered habitat. Remaining populations are small and scattered in comparison to historic conditions. Remaining populations consist of a small number of breeding adults.

Threats

Yosemite toads are declining because of habitat loss. Habitat loss and fragmentation has been caused by construction of new roads, parking lots, water diversion, and cattle grazing. In addition, many of the waterbodies have been heavily polluted by human recreation and now have degraded water quality. Riverbanks have been damaged; this has caused disruption of vegetation and erosion along the banks, in turn resulting in excess sedimentation in the lakes, streams, and ponds. These conditions are either unsuitable for the Yosemite toad to live in, or render the habitat unable to provide the type of vegetation or protection that the Yosemite toad requires. Habitat loss, damage, and fragmentation are killing Yosemite toads; they are unable to adapt to poor water quality conditions, limiting the amount of quality habitat available to them (USDA 2015).

Amphibian Chytrid fungus (*Batrachochytrium dendrobatidis*, Bd) is a known cause for amphibian declines worldwide. Although its specific effects on the Yosemite toad are still being researched, the disease has been found in dead Yosemite toads. Because many species closely related to the Yosemite toad have been negatively affected by Bd, it is thought that the fungus will have a detrimental effect on the Yosemite toad population. One species that is being exterminated by this disease is the mountain yellow-legged frog, which is found in environments overlapping those of the Yosemite toad, exposing Yosemite toads to this disease. In addition, Bd thrives in cold temperatures; the fungus spores are spread through waterbodies across the Sierra Nevada, where the Yosemite toad is found (Davidson et al. 2015, California Herps 2015, IUCN 2015, USDA 2015).

Yosemite toads have a low tolerance for both extreme cold and hot temperatures—meaning that any climate shift, even slight, could have a negative effect on Yosemite toad populations. In addition, Yosemite toads breed in shallow pools of water, and changes to the temperatures can have an effect on the hydrologic cycle. Decreases in water availability can be detrimental to the continuation of Yosemite toad populations, because such changes can result in stranding and death of eggs and tadpoles. This has already been found to cause death in an entire year's cohort when the water evaporates rapidly. Adults will be affected by climate change, because a reduction in melting snowpacks has the potential to lead to a loss of foraging, breeding, and refugia habitat. Severe winters may force extended overwintering, which can kill toads through stress, a reduction of feeding and breeding time, and a reduction in resources needed to survive, especially for an extended hibernation (USDA et al. 2015).

Livestock grazing has the potential to affect all life stages of Yosemite toads. Cattle eat and trample the meadows where adult Yosemite toads are found, eliminating vegetation, compacting the ground, decreasing site productivity, and causing habitat damage. Livestock have also created water quality degradation and nitrogen pollution; destroyed banks; or made banks unstable and susceptible to erosional forces. Both adults and eggs have been crushed by cattle. Theses alterations and damages create unsuitable living conditions for the Yosemite toad, and destroy the habitat in which they can be found (Davidson et al. 2015, California Herps 2015, IUCN 2015, USDA 2015).

The contribution of ultraviolet (UV-B) radiation to amphibian decline is currently being debated in the scientific community. The depletion of atmospheric ozone has led to an increase in UV-B radiation, which can affect and destroy egg embryos. Most scientists say that current levels of UV-B radiation do not affect Yosemite toads; but if the ozone becomes weaker, it could have a pronounced effect on the species (Davidson et al. 2015, USDA 2015).

Yosemite toads are very sensitive to water quality issues. A variety of pesticides are used in large quantities in California's central valley. These pesticides can affect suitable habitats for the frog when wind, acid rain, and storms conduct in contact with the drift line of the pesticides. Pesticides can harm eggs and larval or adults as a direct toxin or by causing developmental mutations, malformations, sterilization, and weakened immune systems (Davidson et al. 2015, California Herps 2015, IUCN 2015, USDA 2015).

Many roads have been created in the Sierra Nevada as the number of visitors has increased. Roads fragment Yosemite toad habitat, creating pollution and run-off that affect water quality. In addition, there are high amphibian mortalities caused by automobile traffic, especially during spring storms when amphibians can often be found on roadways (USDA 2015).

Five-Year Status Review

Currently, there are no five-year status reviews for this species. On February 10, 2020, the USFWS initiated a 5-year status reviews of 66 species in California and Nevada, including the Yosemite toad.

Critical Habitat

On August 26, 2016, the U.S. Fish and Wildlife Service (Service) designated critical habitat for the Sierra Nevada yellow-legged frog (*Rana sierrae*), the northern distinct population segment (DPS) of the mountain yellow-legged frog (*Rana muscosa*), and the Yosemite toad (*Anaxyrus canorus*) under the Endangered Species Act of 1973, as amended (Act). There is significant overlap in the critical habitat designations for these three species. The designated area, taking into account overlap in the critical habitat designations for these three species, is in total approximately 733,357 hectares (ha) (1,812,164 acres (ac)) in Alpine, Amador, Calaveras, El Dorado, Fresno, Inyo, Lassen, Madera, Mariposa, Mono, Nevada, Placer, Plumas, Sierra, Tulare, and Tuolumne counties, California. All critical habitat units and subunits are occupied by the respective species. There are 16 units of designated critical habitat.

There are 303,889 ha (750,926 ac) of designated critical habitat for the Yosemite toad. This area represents approximately 28 percent of the historical range of the Yosemite toad in the Sierra Nevada. All units designated as critical habitat are considered occupied (at the unit level) and include lands within Alpine, Tuolumne, Mono, Mariposa, Madera, Fresno, and Inyo counties, California.

Critical habitat units are designated for Alpine, Tuolumne, Mono, Mariposa, Madera, Fresno, and Inyo counties, California. Within these areas, the primary constituent elements of the physical or biological features essential to the conservation of the Yosemite toad consist of two components:

i. Aquatic breeding habitat. (A) This habitat consists of bodies of fresh water, including wet meadows, slow-moving streams, shallow ponds, spring systems, and shallow areas of lakes, that: (1) Are typically (or become) inundated during snowmelt; (2) Hold water for a minimum of 5 weeks, but more typically 7 to 8 weeks; and (3) Contain sufficient food for tadpole development. (B) During periods of drought or less than average rainfall, these breeding sites may not hold surface water long enough for individual Yosemite toads to complete metamorphosis, but they are still considered essential breeding habitat because they provide habitat in most years.

ii. Upland areas. (A) This habitat consists of areas adjacent to or surrounding breeding habitat up to a distance of 1.25 kilometers (0.78 miles) in most cases (that is, depending on surrounding landscape and dispersal barriers), including seeps, springheads, talus and boulders, and areas that provide: (1) Sufficient cover (including rodent burrows, logs, rocks, and other surface objects) to provide summer refugia, (2) Foraging habitat, (3) Adequate prey resources, (4) Physical structure for predator avoidance, (5) Overwintering refugia for juvenile and adult Yosemite toads, (6) Dispersal corridors between aquatic breeding habitats, (7) Dispersal corridors between breeding habitats and areas of suitable summer and winter refugia and foraging habitat, and/or (8) The natural hydrologic regime of aquatic habitats (the catchment). (B) These upland areas should also maintain sufficient water quality to provide for the various life stages of the Yosemite toad and its prey base.

Recovery Plan Information

No recovery plan has been created for the Yosemite toad.

Environmental Baseline

The Yosemite toad and its designated critical habitat occur in the Sierra Nevada, California. Please refer to information above for the environmental baseline.

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Reptiles

Alameda Whipsnake (= Striped Racer) (*Masticophis lateralis euryxanthus*) and its Critical Habitat

Listing Status

The Alameda whipsnake, also known as striped racer, was listed as threatened on December 5, 1997 (62 FR 64306). No Distinct Population Segments have been defined. Critical habitat was designated for this species on October 2, 2006 (71 FR 58176).

Life History and Habitat

Alameda whipsnakes are typically associated with small to large patches of chaparral or coastal scrub vegetation, interspersed with other native vegetation types and rock lands (areas containing large percentage of rocks, rocky features, and/or rock-bearing soil types). Alameda whipsnakes were also observed using adjacent vegetation types, including grassland, oak savanna, and oak-bay woodland, up to 150 m (500 ft.) from coastal scrub and chaparral. Alameda whipsnakes use all slope aspects and brush community canopy closures, but were found to be concentrated on slopes facing south, southwest, southeast, east, or northeast. Alameda whipsnakes usually had more than one core area, separated by more northerly aspects. Northerly aspects were used on a regular basis to move between core areas. Selection for southerly and easterly aspects is likely related not only to consistently warmer temperatures, but is also associated with the availability of morning sun, which promotes emergence earlier in the day and maximizes the activity period for foraging, mate finding, and digestion (USFWS 2011). Chaparral and coastal scrub vegetation serve as the center of home ranges, providing for foraging opportunities and concealment from predators. Core areas have been found to center around patches of coastal scrub or chaparral as small 0.2 hectare (ha) (0.5 acre [ac.]) embedded in a mosaic of other dominant vegetation types (USFWS 2011). Whipsnakes also require rock outcrops or talus. Small rodent burrows are important retreats, and brush piles and deep soil crevices can also serve as important habitat features. These habitat features are essential for normal behaviors such as breeding, reproduction, and foraging, because they provide egg-laying sites, refuge from predators, thermal cover, shelter, winter hibernacula, and increased foraging opportunities. Whipsnake habitat was directly lost to urban growth; fragmentation due to freeway construction and commercial and residential developments also created barriers to species dispersal, further isolating populations and subpopulations (USFWS 2011).

Alameda whipsnakes are ovoviviparous and have been observed in polyandrous partnerships. Courtship and mating occur from late March through mid-June. During this time, males have been found to move throughout their home range, and females have been found to remain at or near their hibernaculum until mating is complete. A female was observed copulating with more than one male during a mating season, but the extent to which females mate with multiple males (polyandry) is unknown. Suspected egg-laying sites were located in patches of grassland, within 3 to 6 m (10 to 20 ft.) of coastal scrub, and were also found in areas of low density scattered scrub intermixed with grassland. Rock outcrops or talus, small rodent burrows, brush piles, and deep soil crevices are essential for normal behaviors such as breeding, reproduction, and foraging, because they provide egg-laying sites, refuge from predators, thermal cover, shelter, winter hibernacula, and increased foraging opportunities (USFWS 2011). Sperm is stored by the male over winter, and copulation commences after emergence from winter hibernacula. Females begin yolk deposition in mid-April, and intervals of 47, 50 and 55 days have been recorded between dates of first known mating and first egg laid. The average clutch size was found to be 7.21 (with a range of 6 to 11), with a significant correlation between body size and clutch size. Incubation lasts about 3 months, and young appear in late summer and fall (USFWS 2011). Hatchlings have been observed or captured above ground from August through November. Hatchlings have been observed with prey in their stomachs prior to winter hibernation, indicating parental care. California whipsnakes (*Masticophis lateralis*) reach maturity in 2 to 3 years, with adults growing to nearly 1.5 m (5 ft.). Based on a study of captive California whipsnakes, they may live for 8 years (USFWS 2011).

Alameda whipsnakes are opportunistic and active daytime predators. They prey extensively on western fence lizards (*Sceloporus occidentalis*), and are often used as an example of a feeding specialist (USFWS 2005). When hunting, the Alameda Whipsnake commonly moves with its head held high and occasionally moves it from side to side to peer over grass or rocks for potential prey (USFWS 2005). Prey is apprehended quickly, pinioned under loops of the body, and engulfed without constriction. In addition to western fence lizards, Alameda whipsnakes feed on a variety of secondary prey; frogs (*Pseudacris* sp. and *Lithobates* sp.), skinks (*Scincidae* sp.), alligator lizards (*Elgaria* sp.), snakes, small birds, amphibians, California slender salamanders (*Batrachoseps attenuatus*), small mammals, fish, and insects are also important in the whipsnake's diet (NatureServe 2015; USFWS 2005; USFWS 2011). The Alameda whipsnake is semi-arboreal and can escape into or hunt in shrubs or trees. Adult Alameda whipsnakes have a bimodal seasonal activity pattern, with peaks during the spring mating season and smaller peak during late summer and early fall. They generally retreat to winter hibernacum in November and emerge in March; however, short periods of aboveground activity such as basking in the immediate vicinity of the hibernaculum may occur during this time. The Alameda whipsnake is an active daytime predator (USFWS 2011). Rock outcrops are an important feature of their habitat, because they provide retreat opportunities for whipsnakes and promote lizard populations (USFWS 2005).

Population Status

Rangewide Status of the Species

The Alameda whipsnake inhabits the inner Coast Ranges in western and central Contra Costa and Alameda counties, California. The historical range was continuous, but has been fragmented into five disjunct populations: Tilden–Briones, Oakland–Las Trampas, Hayward–Pleasanton Ridge, Sunol–Cedar Mountain, and Mount Diablo–Black Hills (62 FR 64306).

The range of the Alameda whipsnake and phenotypic-intergrade specimens includes mosaics of chaparral, coastal scrub, and adjacent vegetation types throughout Contra Costa County, most of Alameda County, and small portions of northern Santa Clara and western San Joaquin counties. This range can be subdivided into five populations that correspond to relatively contiguous mosaics of suitable habitat types that are fragmented by urban development, transportation corridors, and a lack of coastal scrub and chaparral vegetation in the Tri-Valley. Alameda whipsnakes have been found to be locally abundant, and are the dominant snake species when habitat quality is high (USFWS 2011).

Population Summary

The current population size, trend levels, and minimum viable population size are undescribed. There are five populations (corresponding to the species' recovery units) within a fragmented regional metapopulation: 1) Tilden–Briones; 2) Oakland–Las Trampas; 3) Hayward–Pleasanton Ridge; 4) Mount Diablo–Black Hills; and 5) Sunol–Cedar Mountain. Two additional recovery units are associated with movement corridors: Caldecott Tunnel Corridor and Niles Canyon/Sunol Corridor (USFWS 2002; USFWS 2011). Population and species-level trends are assumed to be in decline (a short-term decline of 10 to 30 percent), based on the continued habitat loss, alteration, and fragmentation of known extant habitat (NatureServe 2015; USFWS 2011). In the five populations, there are varying degrees of isolation due to natural and human-caused barriers; these result in varied gene flow within populations and little to none between populations. The boundaries of these five populations and two associated dispersal corridors represent the extent of suitable habitat that includes known Alameda whipsnake locations.

Habitat was directly lost to urban growth; fragmentation due to freeway construction and commercial and residential developments also created barriers to species dispersal, further isolating populations and subpopulations (USFWS 2011).

Remaining natural habitat in these areas may provide movement corridors for the Alameda whipsnake, but it is as yet unknown whether whipsnakes are able to use these corridors in a manner that would promote gene flow (USFWS 2002; USFWS 2011). Little population abundance data exists for the Alameda whipsnake. However, Alameda whipsnakes have been found to be locally abundant and the dominant snake species when habitat quality is high. Almost all trapping studies targeting this species have been designed to determine presence or absence for regulatory purposes and assessing impacts to potential habitat. Monitoring is therefore most often habitat based, assuming snake abundance is positively correlated with the amount of coastal scrub or chaparral vegetation and rock lands present. No studies have been performed that have quantified Alameda whipsnake densities relative to habitat quality or quantity (USFWS 2011).

Threats

Threats to this species include:

- Urbanization and habitat destruction are the greatest threats to the Alameda whipsnake throughout much of its range.
- Numerous water storage reservoirs were constructed throughout the range of the Alameda whipsnake (i.e., San Pablo, Briones, Lake Chabot, and Upper San Leandro reservoirs). These reservoirs resulted in the inundation and large-scale losses and fragmentation of Alameda whipsnake habitat.
- Fire suppression indirectly threatens the Alameda whipsnake by allowing plants to establish a closed canopy that tends to create relatively cool conditions that are less suitable to the Alameda whipsnake, which maintains a relatively high active body temperature.
- Fire suppression: It has been determined that the natural fire return interval for the San Francisco East Bay is 10 to 30 years, and that fire suppression has exacerbated the effects of wildfires by allowing a buildup of fuels, creating the conditions for hotter fires that may directly kill Alameda whipsnakes that do not find retreat in burrows or rock crevices.
- The presence of nonnative plant species is a significant concern for the Alameda whipsnake.
- Succession of core Alameda whipsnake habitat is occurring, from coastal scrub and chaparral to other native vegetation types. It is hypothesized this succession is due to the removal of disturbance regimes. This threat is greatest on more mesic sites where fire and grazing have been removed, particularly on sites in the fog belt in the East Bay Hills.
- Because Alameda whipsnakes forage in grasslands between stands of scrub, livestock grazing that significantly reduces or eliminates plant cover in these grasslands could lead to an increased loss of Alameda whipsnakes and their prey to predation.
- Loss and fragmentation of habitat as a result of road and trail construction is a stressor for the Alameda whipsnake. Roads can impede gene flow and dispersal. Networks of roads and trails fragment habitat, reduce patch size, and increase the ratio of edge to interior habitat.
- Global climate change increases the frequency of extreme weather events, such as heat waves, droughts, and storms. Extreme events, in turn, may cause mass mortality of individuals and significantly contribute to determining which species will remain or occur in natural habitats.

Five-Year Status Review

On April 27, 2012, the USFWS conducted a five-year status review of the Alameda whipsnake, which resulted in no change in listing status (77 FR 25112).

Critical Habitat

On October 2, 2006, the U.S. Fish and Wildlife Service designated critical habitat for the Alameda whipsnake (71 FR 58176). Six critical habitat units were designated in Alameda, Contra Costa, Santa Clara, and San Joaquin counties, California.

Seven critical habitat units (1, 2, 3, 4, 5A, 5B, and 6) are designated as critical habitat for the Alameda whipsnake, encompassing approximately 154,834 acres (ac) (62,659 hectares (ha)), as follows:

- Unit 1: Tilden-Briones; Alameda and Contra Costa counties (34,119 ac (13,808 ha)).
- Unit 2: Oakland-Las Trampas; Contra Costa and Alameda counties (24,436 ac (9,889 ha)).
- Unit 3: Hayward-Pleasanton Ridge; Alameda County (25,966 ac (10,508 ha)).
- Unit 4: Mount Diablo-Black Hills; Contra Costa and Alameda counties (23,225 ac (9,399 ha)).
- Unit 5A: Cedar Mountain; Alameda and San Joaquin counties (24,723 ac (10,005 ha)).
- Unit 5B: Alameda Creek Unit; Alameda and Santa Clara counties (18,214 ac (7,371 ha)).
- Unit 6: Caldecott Tunnel; Contra Costa and Alameda counties (4,151 ac (1,680 ha)).

Critical habitat units are designated for Alameda, Contra Costa, San Joaquin, and Santa Clara counties, California. The primary constituent elements (PCEs) of critical habitat for the Alameda whipsnake are the habitat components that provide:

(i) Scrub/shrub communities with a mosaic of open and closed canopy: Scrub/shrub vegetation dominated by low- to medium-stature woody shrubs with a mosaic of open and closed canopy, as characterized by the chamise, chamise-eastwood manzanita, chaparral whitethorn, and interior live oak shrub vegetation series occurring at elevations from sea level to approximately 3,850 feet (1,170 meters). Such scrub/shrub vegetation within these series form a pattern of open and closed canopy used by the Alameda whipsnake for shelter from predators; temperature regulation, because it provides sunny and shady locations; prey-viewing opportunities; and nesting habitat and substrate. These features contribute to support a prey base consisting of western fence lizards and other prey species such as skinks, frogs, snakes, and birds.

(ii) Woodland or annual grassland plant communities contiguous to lands containing PCE 1: Woodland or annual grassland vegetation series comprised of one or more of the following: Blue oak, coast live oak, California bay, California buckeye, and California annual grassland vegetation series. This mosaic of vegetation supports a prey base consisting of western fence lizards and other prey species such as skinks, frogs, snakes, and birds, and provides opportunities for: Foraging, by allowing snakes to come in contact with and visualize, track, and capture prey (especially western fence lizards, along with other prey such as skinks, frogs, birds); short and long distance dispersal within, between, or adjacent to areas containing essential features (i.e., PCE 1 or PCE 3); and contact with other Alameda whipsnakes for mating and reproduction.

(iii) Lands containing rock outcrops, talus, and small mammal burrows. These areas are used for retreats (shelter), hibernacula, foraging, and dispersal, and provide additional prey population support functions.

Recovery Plan Information

A final recovery plan has not been issued; however, a draft recovery plan was issued in November 2002 (USFWS 2002).

Reclassification Criteria

No reclassification criteria have been identified.

Delisting Criteria

Delisting criteria included below are from the draft recovery plan.

- Specified recovery areas are secured and protected from incompatible uses (USFWS 2002). a) Protection for 75 to 100 years of 90 percent of "long-term protection" habitat; and b) Permanent protection of 100 percent of focus areas ("protection in perpetuity" habitat, as refined based on spatial analysis and surveys. Areas include population centers, connectivity areas, corridors, and buffer areas).
- Management plans oriented to species conservation (and adaptively updated based on current research) are approved and implemented for recovery areas (USFWS 2002). Management plans that have the survival and recovery of the species as objectives are: a) Approved and implemented on 100 percent of all focus areas; b) Approved and implemented on 30 percent of lands outside of focus areas but within the recovery unit boundaries; c) Approved, and implementation has begun in an additional 20 percent of the recovery units outside the focus areas; and d) Assured of adequate funding for long-term management.
- Monitoring in recovery areas demonstrates stable or improving trends in species populations and successional diversity of natural habitat (USFWS 2002). a) Representative populations or subpopulations representing the genetic variation and geographic extent of the species, as identified by surveys and genetic study, are stable or increasing with evidence of natural recruitment for a period of 1.5 fire cycles (approximately 60 years) that include normal disturbances; and b) Habitat monitoring shows a mosaic of multi-age class stands, and that habitat fragmentation has not appreciably increased (less than 5 percent) in any recovery unit over current (2002) conditions.
- Threats are ameliorated or eliminated, and fire techniques for habitat management are studied and implemented (USFWS 2002).
- Achieve a mosaic of habitats, ideally through reestablishment of natural fire frequency (USFWS 2002).
- Increased public awareness in the four-county area on urban/wildland issues (USFWS 2002).

Recovery Actions

A final recovery plan has not been issued; however, a draft recovery plan was issued in November 2002 and contained draft recovery actions. The 2011 5-Year Review also contains recommended actions. Both the draft recovery actions and the recommended actions are presented below (USFWS 2002, USFWS 2011).

- Form a Recovery Implementation Team that cooperatively implements specific management actions necessary to recover the species (USFWS 2002).
- Conduct public outreach and education; and develop and implement a regional cooperative program (USFWS 2002).
- Conduct mapping, assessment, and analysis exercise (USFWS 2002).
- Protect and conserve the ecosystems upon which the species depends (USFWS 2002).
- Protect and secure existing populations and habitat (USFWS 2002).
- Survey historical locations and other potential habitat where this species may occur (USFWS 2002).
- Conduct necessary biological research and use results to guide recovery/conservation efforts (USFWS 2002).

- Prepare management plans and implement appropriate management in areas inhabited by this special-status species (USFWS 2002).
- Augment, reintroduce, and/or introduce this species (USFWS 2002).
- Develop a tracking process for the completion of recovery tasks and the achievement of delisting criteria (USFWS 2002).
- Refine delisting criteria (USFWS 2002).
- Conduct status reviews of the species to determine whether listing as endangered or threatened is necessary (USFWS 2002).
- Assess the applicability, value, and success of this recovery plan to the recovery of Alameda whipsnake every 5 years until the recovery criteria are achieved (USFWS 2002).
- Promote the eradication of blue gum (Eucalyptus globules), Monterey pine (Pinus radiata), Monterey cypress (*Cupressus macrocarpa*), and French broom (*Genista monspessulana*), and other nonnative invasive species in the San Francisco East Bay (USFWS 2011).
- Focus land protection efforts on undeveloped parcels in the Wildland Urban Interface to reduce urban sprawl into chaparral and coastal scrub vegetation, and to reduce the need for fuel reduction treatments in Alameda whipsnake habitat (USFWS 2011).
- Conduct a genetic study, using nuclear DNA, to determine the genetic basis for the phenotype and to determine whether there is a geographic boundary separating the Central and the Southern California clades, whether individuals from each of these clades coexist, and whether gene exchange between the two clades occurs (USFWS 2011).

Environmental Baseline

The Alameda whipsnake and its designated critical habitat occur in Alameda, Contra Costa, Santa Clara, and San Joaquin counties, California. Please refer to information above for the environmental baseline.

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Giant Garter Snake (*Thamnophis gigas*)

Listing Status

The giant garter snake was listed as threatened under the Endangered Species Act on October 20, 1993 (Service 1993). The loss and subsequent fragmentation of habitat is the primary threat to the species.

Life History and Habitat

Giant garter snakes inhabit marshes, sloughs, ponds, small lakes, low-gradient streams, and other waterways and agricultural wetlands such as irrigation and drainage canals, rice fields, and the adjacent uplands. The following three habitat components have been identified as the most important to the giant garter snake: 1) a fresh-water aquatic component with protective emergent vegetative cover that will allow for foraging; 2) an upland component near the aquatic habitat that can be used for thermoregulation and for summer shelter in burrows; and 3) an upland refugia component that will serve as winter hibernacula (Service 2017). Giant garter snakes appear to be most numerous in rice-growing regions. The diverse habitat elements of rice-lands contribute structure and complexity to this man-made ecosystem. Spring and summer flooding and the fall drying of rice fields coincide closely with the biological needs of the species (Service 1999). In the summer, giant garter snakes are most likely found in aquatic habitats, typically in active rice fields and most often under aquatic vegetation cover (Service 2012). Giant garter snakes are absent from larger rivers and other water bodies that support introduced populations of large, predatory fish, and from wetlands with sand, gravel, or rock substrates (Service 1993). Giant garter snakes need enough water to provide food and cover during the active season from early spring through mid-fall. They also need emergent wetland plants such as cattails (*Typha* sp.) for coverage and foraging, and grassy banks and openings in vegetation for sunning. During the winter, when they are largely inactive, giant garter snakes need small mammal burrows and other crevices above flood elevations (Service 1999; Service 2012).

Population Status

Giant garter snakes have a population of 2,500 to 100,000 snakes throughout 13 known populations; however, two are presumed extirpated and three have been combined into a single population, leaving nine extant populations identified by surveys conducted in 2011. The populations are genetically different from each other, leading to a push to have distinct population segments. The short-term population-level trend of this species is a decline of 10 to 30 percent. The long-term population-level trend is a decline of 30 to 50 percent (NatureServe 2022; Service 2012). Currently, populations of the giant garter snake are found in the Sacramento Valley and isolated portions of the San Joaquin Valley; however, the species is extirpated from most of the San Joaquin Valley. Extant populations are distributed in portions of rice production zones of Sacramento, Sutter, Butte, Colusa, and Glenn counties, along with the western border of the Yolo Bypass in Yolo County, and along the eastern fringes of the Sacramento-San Joaquin Delta from the Laguna Creek-Elk Grove region of central Sacramento County southward to the Stockton area of San Joaquin County. As of 2017, there are 9 known populations, found at: (1) Butte Basin; (2) Colusa Basin; (3) Sutter Basin; (4) American Basin; (5) Yolo Basin; (6) Cosumnes-Mokelumne Basin; (7) Delta Basin; (8) San Joaquin Basin; and (9) Tulare Basin (Service 2017).

The species is threatened by:

1) Habitat loss, fragmentation and degradation due to urbanization, infrastructure development and agricultural conversion, including changing fields from rice production to orchards;
2) Invasive aquatic plants and removal techniques for those plants, including herbicides or mowing; and
3) The impacts of climate change, including:

a) flooding, which can displace snakes and bury them under debris or cause drowning when overwintering in burrows, and
b) drought, due to the species' dependence on permanent wetlands.

Critical Habitat

Critical habitat has not been designated at this time.

Recovery Plan Information

If a recovery plan has been developed, describe that here and any important information that would influence the conclusion regarding precluding recovery of the species.

The Recovery Plan for the Giant Garter Snake was published by the Service in September 2017 (Service 2017). The strategy used to recover the giant garter snake is focused on protecting existing, occupied habitat and identifying and protecting areas for habitat restoration, enhancement, or creation including areas that are needed to provide connectivity between populations. The goal of this recovery plan is to reduce threats to and improve the population status of the giant garter snake sufficiently to warrant delisting. To achieve this goal, we have defined the following objectives:

1) Establish and protect self-sustaining populations of the giant garter snake throughout the full ecological, geographical, and genetic range of the species.
2) Restore and conserve healthy Central Valley wetland ecosystems that function to support the giant garter snake and associated species and communities of conservation concern such as Central Valley waterfowl and shorebird populations.
3) Ameliorate or eliminate, to the extent possible, the threats that caused the species to be listed or are otherwise of concern, and any foreseeable future threats.

Environmental Baseline

The species only occurs within the State of California, please refer to the information above regarding the species environmental baseline.

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Northwestern Pond Turtle

Listing Status

The northwestern pond turtle is not listed under the Act; however, it is currently proposed threatened and under federal review for listing under the Act (88 FR 68370).

In July 2012, the Service was petitioned to list 53 species of reptiles and amphibians, including the western pond turtle (*Actinemys marmorata*), as threatened or endangered under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531). The Service published a substantial 90-day petition finding on April 10, 2015 (80 FR 19262). After publication of the 90-day finding, the western pond turtle was split into two separate species: the northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*). The Service published a proposed rule to list both species as threatened, with a Section 4(d) rule, on October 3, 2023 (88 FR 68370), but did not propose designation of critical habitat due to a lack of sufficient data from which to perform an analysis. The proposed rule also served as a warranted 12-month finding for the two species. Both species will be considered by the Service as "proposed threatened" until publication of a final listing rule in the Federal Register.

Life History and Habitat

The northwestern pond turtle is a medium-sized turtle with size varying geographically, with the largest animals occurring in the northern part of the range (Holland 1994). The maximum carapace (shell) length (CL) is 241 millimeters (mm) (Lubcke and Wilson 2007). Adults typically range in size between 160 to 180 mm long and weigh between 500 to 700 grams (g; Bury et al. 2012). The northwestern pond turtle is sexually dimorphic: Females tend to have a smaller head, less angled snout, taller and rounder carapace, flat (rather than concave) plastron (underside of shell), and thinner tail as compared to males (Holland 1994, Rosenberg et al. 2009). Colors and markings vary geographically and by age with most appearing olive to dark brown, or blackish, occasionally without pattern but usually with a network of spots, lines, or dashes of brown or black that often radiate from growth centers of shields (Bury et al. 2012, Stebbins and McGinnis 2018). The plastron is yellowish, blotched with blackish or dark brown, and occasionally unmarked (Stebbins and McGinnis 2018). Coloration of the head and neck vary by sex, geography, and age (Hays et al. 1999). Males usually have a light-yellowish chin and underside of the throat whereas females tend to have dark prints or rosette rings that usually remain throughout their life. Hatchlings are generally a brown-olive color with visible mottling on the head and neck (Hays et al. 1999) that darken with age. Eggs are off-white, elliptical-oval shaped, and range from 32 to 42 mm in length and from 18 to 25 mm in diameter (Bury et al. 2012). When hatchlings emerge from the nest, they weigh 3 to 7 g with a 25 to 31 mm CL (Bury et al. 2012). The shell of hatchlings is soft and pliable, and the tail is nearly as long as the shell (Ashton et al. 1997, Stebbins and McGinnis 2018). Northwestern pond turtle shells harden by age 3 or 4 (Bury et al. 2012).

Northwestern pond turtles are semi-aquatic, having both terrestrial (hereafter "upland") and aquatic life history phases. Eggs are laid in upland habitat, and hatchings, juveniles, and adults use both upland and aquatic habitat. The amount of time spent on land varies by location and aquatic habitat type. Upland environments are required for nesting, overwintering and aestivation (i.e., warm season dormancy), basking, and movement/dispersal. Aquatic environments are required for breeding, feeding, overwintering, sheltering, basking, and movement/dispersal. The northwestern pond turtle can be found in perennial or intermittent water bodies including streams, rivers, irrigation ditches, ponds, lakes, and reservoirs.

Activity Patterns

<u>*Basking*</u>

Northwestern pond turtles engage in both emergent (i.e., out of the water) and aquatic basking. Basking is essential for thermoregulation and in turn, physiological functions such as metabolism, digestion, reproduction, and growth. Additional benefits of emergent basking include drying of the shell and skin for parasite and algal control.

The amount of time spent basking varies throughout the range depending on water and air temperature. On the Trinity River in northwestern California, northwestern pond turtles spent more time emergent basking in the main fork, which had cooler water temperatures, than in the warmer south fork (Ashton et al. 2011). Similarly, at both an intermittent stream and perennial stream in Sequoia National Park, California, northwestern pond turtles were more likely to be emergent basking as air temperature increased (Ruso et al. 2017). At the University of California, Davis campus, in northern California, northwestern pond turtles were more abundant at basking sites when water temperatures were warm, and the sites were unshaded (Lambert et al. 2013). Basking structures may be especially important below dams with cold water discharge (Ernst and Lovich 2009). During emergent basking, northwestern pond turtles will retreat into the water or may seek shade once it gets too hot or when body core temperature has reached a desirable level (Ernst and Lovich 2009). Aquatic basking is where they lay completely or almost completely submerged in [warmer] shallow water or within a floating mat of vegetation; and are therefore less visible than when emergent basking (Holland 1992).

Overwintering/Aestivation

The northwestern pond turtle overwinters–in a physical state of little to no activity–during the cooler months of the year in either upland or aquatic environment (Holland 1994, Ultsch 2006). In contrast, aestivation is a period of inactivity, usually in response to the hottest time of year or dry conditions, that only occurs in upland habitat (Hays et al. 1999).

The amount of time spent overwintering and/or aestivating varies geographically and within populations and is likely influenced by climate and hydrological conditions. At two sites in California, western pond turtles left intermittent ponds as they dried out and overwintered in upland habitat, returning to the ponds weeks or months after they refilled (Pilliod et al. 2013, Zaragoza et al. 2015). Similarly, western pond turtles inhabiting intermittent streams may respond to late summer drying and winter flooding by moving into upland habitat (Rathbun et al. 2002). However, in perennial streams and rivers, turtles may remain active until fall/winter storms increase flows and reduce water temperatures (Belli 2016). In northern California, beginning in September, northwestern pond turtles spent seven months of the year away from the Trinity River to overwinter in upland habitat, while others moved to nearby lentic bodies of water (i.e., a lake and a slough) as far as 500 m from the river (Reese and Welsh 1997). Movements, in this case, may have been to avoid winter flood events (Reese and Welsh 1997, Rathbun et al. 2002). Moving to upland habitats above the flood line is generally more common for turtles occupying lotic (i.e., actively moving water) habitats. Along the central California coast, northwestern pond turtles that occupied pond habitat overwintered on-site, whereas most turtles from an adjacent stream left with the first heavy rains and overwintered in the upland habitat or moved to the pond (Davis 1998). In response to spring storms, some turtles remained within the stream under banks or within submerged shoreline or riparian vegetation, whereas others left the stream and moved a minimum of 4 m away (Belli 2016). The range of behaviors between populations and individuals suggests that northwestern pond turtles use several overwintering and aestivation strategies (Holland 1994, Ultsch 2006, Zaragoza et al. 2015). At study sites on the Trinity River and in Santa Rosa in northern California, overwintering locations across successive years were very similar, with distances between overwintering sites as short as one meter (Reese 1996). However, radio-tagged western pond turtles that were tracked for two winters on the Carrizo Plain Ecological Reserve, San Luis Obispo County, California did not have overwintering site fidelity (i.e., they did not return to the same sites) (Pilliod et al. 2013).

Movements and Dispersal

Northwestern pond turtles move between aquatic and upland habitats to nest, overwinter, and aestivate. Males generally move farther than females or juveniles (Bury 1972). Home ranges average 1 hectare for males, 0.3 hectare for females, and 0.4 hectare for juveniles (Bury 1972). Overwintering behavior is variable, and likely more common in seasonally inundated ponds than permanent water (Pilliod et al. 2013). Holland (1994) found overwintering sites at two streams/rivers that were 15 to 260 m from aquatic habitat. In northern California along the Trinity River, some turtles moved to upland habitat to either overwinter or aestivate while others moved to lentic bodies of water (i.e., standing bodies of water) as far as 500 m from the river (Reese and Welsh 1997). The pattern and frequency of these movements vary with habitat, size of the aquatic system, suitability of upland habitat, season, climate, environmental stress (e.g., drought, high stream flow), sex, and life stage (Hallock et al. 2017). In central California, northwestern pond turtles spent over half of the year in upland habitat, moving 255 to 1,096 m within the upland habitat but never moving farther than 343 m from seasonal ponds. Western pond turtles moved in different directions, used different microhabitats, and left ponds at different times (Pilliod et al. 2013).

Dispersal (generally defined as a onetime movement of a juvenile from its natal area to establish its own home range) of western pond turtles is generally not well understood. Genetic analyses suggest that most dispersing turtles stay within their natal drainage (Spinks and Shaffer 2005), but few accounts of juvenile (or the rarer adult) dispersal exist. Within aquatic habitat, a dispersal distance of 7 kilometers (km) upstream was observed (5 km overland distance) (Holland 1994). Dispersal may also occur via aquatic habitats during flood events (Rosenberg et al. 2009). Along the central California coast, Holland (1994) recorded less than 10 dispersal events between drainages during a 10-year study with over 2,100 captures and recaptures across 21 drainages, suggesting that overland movements are uncommon. In that study, the longest overland distance recorded in an area considered to be under the best circumstances (mild climate and short distances between water features), was a single individual travelling 5 km. Holland (1994), also states that no movements between drainages were detected from three other sites with over 1,100 hundred captures and recaptures over a 7-year period. During an extreme drought, Purcell et al. (2017) documented a 2.6 km straight-line distance movement overland in a radio-tagged turtle, with a minimum total distance of 3.3 km moved before the individual found water.

Diet

The western pond turtle is omnivorous and considered a dietary generalist (Holland 1994), consuming a wide variety of food items, but animal matter appears to constitute a larger portion of the diet than plant material (Bury 1986, Holland 1994). Prey are primarily taken in water but can be captured or scavenged on land. However, food obtained on land must be returned to the water for consumption, as they appear to be unable to swallow food above water (Holland 1994). Stomach content analysis reveals a diet consisting of small aquatic invertebrates, small vertebrates (e.g., fish, tadpoles, and frogs), carrion, and plant material (Bury 1986, Holland 1994). In northern California, contents of 77 stomachs included aquatic insects such as dragonfly larvae, mayflies, stoneflies, caddisflies, midges, beetles, and other insects, including upland prey items (e.g., grasshoppers) (Bury 1986). Bury (1986) found that 44 percent of females consumed plant material compared to 10 percent for males. Juveniles consumed mostly invertebrates (Bury 1986), and hatchlings primarily feed on nekton (i.e., free-swimming aquatic animals) and larvae of small aquatic insects (Holland 1994).

Reproduction

Courtship behaviors have been observed from April through November, with mating observed in May through September (Holland 1992), and based on limited observations, appear to occur underwater

(Holland and Bury 1988, Holland 1992, Goodman 1997, Ashton 2007, Bettelheim 2009). For example, in Monterey County, central California, courtship behaviors were observed in mid-April within a 1.5 m-deep pool with copulation documented the following day in shallow water at a depth of approximately 0.1 m (Bettelheim 2009). In southern California, Holland (1988) observed possible courtship behavior in 2 m-deep water in mid-June. In northern California, mating has been observed in "spring" (Reese 1996).

The age and size at which northwestern pond turtles reach sexual maturity is poorly understood and seems to vary by geography and locale (Holland 1994, Rosenberg et al. 2009, Bury et al. 2012). In general, males exhibit external signs of sexual dimorphism around 110 to 120 mm CL (Bury et al. 2012). In coastal central California, the average male reached 120 mm CL in 3.6 years compared to 4.1 years for females and reached 150 mm CL in 8.3 years for males versus 11.1 years for females (Germano and Rathbun 2008). In Washington, males reach sexual maturity in 10 to 12 years (Hays et al. 1999).

Wide variation occurs throughout the ranges of the two species, but in general, most females carrying eggs are over 6 years old (Bury et al. 2012). In Oregon and northern California, females start carrying eggs when they are at least 120 mm CL and typically 8 to 10 years of age. In southern California, the smallest known reproductive female was approximately 111 mm CL and at least 6 to 7 years old, while the smallest reproductive female in Oregon was 131.3 mm CL (Holland 1994). In coastal central California, a female as young as 4 years old and measuring 141 mm CL was documented carrying eggs (Germano and Rathbun 2008). At two sewage treatment facilities in the San Joaquin Valley, California, females were documented carrying eggs at 4.4 years of age with a CL of 155 mm. In these areas, warmer water and high nutrient loads may have increased aquatic invertebrates, providing increased nutrition for faster growth rates (Germano 2010).

Egg laying occurs from May through July, with northern populations laying eggs later in the season than those in the south (Bury et al. 2012). Gravid females leave the water in the late afternoon or early evening and move into upland habitats to excavate a nest (Holland 1994). Females may be out of the water for a few hours to several days with actual nest excavation and egg laying taking from 2 to over 10 hours. Females may make several forays into upland areas prior to actual oviposition and may abandon nest scrapes prior to laying eggs, potentially because of hitting rocks or roots or because of disturbance, which northwestern pond turtles are extremely sensitive to (Holland 1994). Females will moisten the soil around the nest by urinating prior to digging the nest chamber (Holland 1994, Hays et al. 1999).

Females excavate nests in upland habitat 3 to 500 m from aquatic habitat in compact, dry soils (Storer 1930, Holland 1994, Holte 1998), with an average linear distance from water of 51 m (Davidson and Alvarez 2020). Soil conditions and the frequency and degree of disturbance in the upland habitat, likely influence nest distribution (Thomson et al. 2016). Soils need to be loose enough to allow nest excavation, and typically have a high clay or silt component; likely due in part to the proximity to water bodies. Disturbance needs to be infrequent or of sufficiently low intensity that nesting females are not disturbed while digging nests or laying eggs (Ernst and Lovich 2009). Nests are shallow, generally between 9 and 12 centimeters (cm) below the surface (Holland 1994). After the nest is excavated and eggs deposited, females pack the chamber using surrounding material such as mud, dry soil, and vegetation to form a plug that closes off the "neck" of the nest chamber (Holland 1994).

Clutch size varies from 1 to 13 eggs and is positively correlated with body size (Holland 1994, Holte 1998). In a meta-analysis by Bury et al. (2012), mean clutch size ranged from 4.5 to 8.5 eggs. Nesting frequency also varies across the range, based on female age, geographic location, and environmental conditions such as temperature or resource availability (Holte 1998). Most females appear to deposit eggs every other year, but some may oviposit every year (Holland 1992). Double clutches have been documented in southern California (Goodman 1997), coastal Central California (Scott et al. 2008, Germano and Rathbun 2008), Oregon (Riley 2006 in Rosenberg et al. 2009), and Washington (Hays et al. 1999, Schmidt and Tirhi 2015).

Incubation time is 80 to 126 days (Holland 1994), and hatching rates average approximately 70 percent, with complete nest failure being common (Holland 1994). Hatching success is dependent on soil moisture and temperature during incubation: low precipitation and warmer weather during the summer months improved hatching success, whereas cool, wet summers led to reduced hatching success (Holte 1998).

Northwestern pond turtles exhibit temperature-dependent sex determination (TSD) during incubation (Ewert et al. 1994). In California, female hatchlings were more likely when 30 percent of the sex-determining period occurred above 29 degrees Celsius (°C) (84 degrees Fahrenheit (°F)) (Christie and Geist 2017). Lower fluctuations in temperature resulted in development of males, whereas females developed in nests with high and low temperature fluctuations. Temperatures within nests were found to fluctuate daily, varying by more than 20 °C (36 °F) (Geist et al. 2015, Christie and Geist 2017) with higher maximum temperatures reducing egg viability (Christie and Geist 2017).

In southern and central California, some hatchlings may emerge from the nest chamber in late-summer to early-fall, whereas others overwinter in the nest chamber and emerge in spring (Holland 1994). In the northern parts of the range, hatchlings overwinter in the nest (Holland 1994, Reese and Welsh1997). In western Oregon, hatchlings delayed emergence until spring, and typically remained within 2 m of nests for as long as 59 days after initial emergence (Rosenberg and Swift 2013). During migration from their nests to aquatic habitat, hatchlings embedded themselves in soil for up to 22 days at stop-over sites. Hatchlings entered aquatic habitat on average 49 days after initial emergence and traveled an average of 89 m from their nest site. Hatchlings detected in water were always within 1 m of shore and in areas with dense submerged vegetation and woody debris (Rosenberg and Swift 2013).

Western pond turtles can nearly double in size within a year of hatching (Germano and Rathbun 2008, Germano 2010, Bury et al. 2012). Growth rates can vary greatly based on several factors such as geography and environmental conditions (Bury et al. 2012). For example, Holland (1994) found that turtles between 100 to 110 mm in length are generally 4 to 5 years old but may be as young as 3 or as old as 12. In Oregon, northwestern pond turtles were slightly larger than in California, although the Oregon turtles had a slower growth rate, possibly due to cooler temperatures (Germano et al. 2022).

Survivorship and Longevity

Germano (2016) reported that annual mortality rates for young age classes appear to vary greatly, with mortality of juveniles less than 80 mm CL estimated at 26.9 percent and juveniles up to 120 mm CL at 16.2 percent at a site in the San Joaquin desert in northwestern Kern County, California. In contrast, annual mortality rates for juveniles during their first three years was 85 to 90 percent in the Pacific Northwest (Holland 1994).

Holland (1994) suggested western pond turtle annual survivorship is lowest in the smaller size classes and increases as turtles approach their reproductive years at around 120 mm CL. Further, as reproductive turtles age and become larger, average annual survivorship reaches 95 to 97 percent (Holland 1994). The maximum lifespan of western pond turtles is unknown. However, they are known to be long-lived after reaching adulthood, with some living 55 years (Bury et al. 2012). These old individuals are rare in natural populations, but some may successfully reproduce even late in life, based on a radiograph of a 55-year-old female with eggs (Kaufman and Garwood 2022).

Habitat

Northwestern pond turtles require both aquatic and upland habitats that are within proximity and connected to one another. As habitat generalists, northwestern pond turtles occur in a broad range of permanent and ephemeral water bodies including rivers and streams, lakes, ponds, reservoirs, settling ponds, marshes, vernal pools, irrigation ditches, and other wetlands, including within tidal estuaries (Spinks et al. 2003, Ernst and Lovich 2009, Bury et al. 2012, McGinnis 2018).

Despite their ability to use a wide range of aquatic features, suitable aquatic habitats are often rare, due mainly to widespread urbanization and agricultural conversion. Consequently, northwestern pond turtle distribution is fragmented across their range, following the arrangement of suitable aquatic habitat, especially in areas with extensive open, dry terrain between waterways (Bury et al. 2012). Movements between aquatic and upland habitats are typically less than 500 m (Reese and Welsh 1997), thus aquatic and upland habitats must be adjacent. In a study in northern California, radio-tagged males used upland habitat in at least ten months of the year, emphasizing the importance of upland habitat in addition to aquatic habitat (Reese and Welsh Jr 1997).

Aquatic Habitat

Northwestern pond turtles use aquatic habitat for breeding, feeding, overwintering, and sheltering. Suitable aquatic habitat must contain abundant basking sites, underwater shelter sites (e.g., undercut banks, submerged vegetation, mud, rocks, logs), and standing or slow-moving water (Holland 1992, Reese and Welsh 1998, Hays et al. 1999, Ernst and Lovich 2009). Northwestern pond turtles inhabiting lentic aquatic habitat, such as ponds, lakes, and slack water habitats, often overwinter within the aquatic environment, burying themselves within the bottom substrate, such as mud. Various depths of water provide northwestern pond turtles with habitat necessary for overwintering and hatchling growth. Primary habitat for hatchlings and young juveniles is shallow water with dense submerged vegetation and logs, which most likely provides shelter, prey, and thermoregulatory requirements or other functions for survival (Holland 1994, Rosenberg and Swift 2013).

Basking Sites

Emergent basking (i.e., basking above water or on adjacent upland areas) usually takes place on logs, rocks, emergent vegetation, shorelines, and essentially any other substrate located within and adjacent to aquatic habitat (Holland 1994, Hays et al. 1999). The location of emergent basking sites above or adjacent to aquatic habitat allows for quick retreat into the water if there is perceived danger (Storer 1930). At a site in northern California, stream microhabitats containing emergent basking sites had more turtles present than those without available emergent basking sites (Reese and Welsh 1998). Aquatic basking occurs in shallow water, a top layer of vegetation, or in submerged vegetation, such as algal mats. Aquatic basking may be used when emergent basking sites are limited or not present and provide a warmer ambient temperature than the surrounding water (Jennings and Hayes 1994, Reese and Welsh 1998).

Nesting Habitat

Nesting occurs in upland areas that are 3 to 400 m from aquatic habitat (Holland 1994, Holte 1998). Nesting habitat varies greatly across the species range, but typically females excavate nests in compact, dry soils with sparse vegetation that contains short grasses and forbs and little or no tree canopy cover to allow for exposure to direct sunlight (Holland 1994, Holte 1998, Rathbun et al. 2002, Rosenberg et al. 2009, Riensche et al. 2019). Along the central coast of California, all successful and attempted nest sites were excavated in compact, hard soils with sparse vegetative cover that included coastal sage scrub, exotic annual grasslands, and weed patches (Rathbun et al. 2002). At a study site in Oregon, nest sites had low, dense vegetation with heights averaging 4.8 cm (range = 0 to 20 cm) (Holte 1998), while at a southwest Washington study site, nesting site vegetation heights were 24 to 45 cm (Lucas 2007 in Rosenberg et al. 2009). At this site, where forest vegetation provided canopy cover, turtles selected more open canopies (average of 14 percent) for nesting, especially southerly aspects, and soil temperatures at nest sites were found to be warmer compared to random sites (Lucas 2007 in Rosenberg et al. 2009). Nests generally occur on south or west aspects but can occur on northwest and southeast aspects (Holland 1994, Lucas 2007 in Rosenberg et al. 2009). Most nest sites are on low to moderate slopes (25 degrees or less), but nest site slope can vary from 0 to 60 degrees (Holland 1994).

Hatchling Upland Habitat

Little is known about upland habitat requirements for hatchlings after emerging from the nest. In western Oregon, use of upland habitat and movement by hatchlings varied, and hatchlings were generally found buried into soil or detritus where they were hidden from view (Rosenberg and Swift 2013). After departing these areas, individual hatchlings made stops for varying durations in a variety of habitats. Habitat features included small patches of forest floor (embedded approximately 8 cm under detritus), small patches of forest (buried approximately 5 to 8 cm in the detritus or directly under moss in dense shrub cover), and in sparsely vegetated areas (typically embedded in soil and completely covered by moss) (Rosenberg and Swift 2013).

Upland Overwintering/Aestivation Habitat

Upland habitat used for northwestern pond turtles overwintering and aestivation varies greatly across the range, but generally occurs above ordinary high-water lines or beyond the riparian zone; although understanding of specific microsite conditions is limited (Lucas 2007, Rathbun et al. 2002, Oregon Department of Fish and Wildlife 2015). In the Trinity River system in northern California the greatest distance northwestern pond turtles traveled from their aquatic habitat to upland overwintering sites was approximately 500 m (Reese and Welsh 1997) and Holland (1994) found overwintering sites at two streams/rivers that ranged from 15 to 260 m from aquatic habitat. While vegetation communities differ from site to site, open areas were avoided for overwintering, and leaf litter was present at most sites (Reese and Welsh 1997, Davis 1998, Rathbun et al. 2002). In central California, northwestern pond turtles generally overwintered in areas where they would be exposed to direct sunlight during a portion of the day (Rathbun et al. 2002). In multiple studies in California, overwintering northwestern pond turtles were found buried beneath 5 to 10 cm of leaf litter (Reese and Welsh 1997, Rathbun et al. 2002).

Population Status

The historical range of the northwestern pond turtle extends along the Pacific Coast from British Columbia, Canada south to southern California. In Washington, the northwestern pond turtle occurs mainly in the vicinity of the Puget Sound and in Oregon the northwestern pond turtle occurs throughout the state west of the Cascade Range. In California the northwestern pond turtle range includes the entire northern two-thirds of the state except in the Sierra Nevada and the central coast. A small portion of the range extends east into Nevada in the Lake Tahoe region (see range maps in Ernst and Lovich 2009 and

Stebbins and McGinnis 2018). The congeneric southwestern pond turtle (*Actinemys pallida*) occurs along the central and southern California coast south into Baja, Mexico.

Northwestern pond turtles have been found at sites from brackish estuarine waters at sea level up to 2,048 meters (m) elevation (Ernst and Lovich 2009) but mostly occur below 1,371 m (Stebbins and McGinnis 2018). Populations in the vicinity of Puget Sound, the Columbia Gorge, and the Carson and Truckee Rivers in Nevada are considered to be isolated from other populations (Holland 1994).

Historical accounts from Vancouver Island and mainland British Columbia, Canada in the lower Fraser River watershed may represent transplanted individuals; no reports of the species are known from either region since 1966 (Gregory and Cambell 1984 in Ernst and Lovich 2009), and northwestern pond turtles are considered extirpated from British Columbia, Canada (Ministry of Environment 2012). Single records from southwestern Idaho and Grant County, Oregon (Nussbaum et al. 1983 in Ernst and Lovich 2009) are likely of introduced individuals (Ernst and Lovich 2009), and other isolated populations within the northwestern pond turtles' native range may also represent introductions (Thomson et al. 2016).

Manzo et al. (2021) collated rough estimates of northwestern pond turtle population sizes from available peer-reviewed literature, reports, and unpublished data sets and found that population size averaged 20.7 individuals (range = 1 to 100+ individuals): Sites with the highest population estimates occurred along the Trinity River in Trinity County, California, and in parts of California's Central Valley (Fresno and Kern counties). While there were several populations estimated over 100 individuals in California and one site with over 100 individuals in Nevada, there was only one population estimated to be over 50 individuals in Oregon (Manzo et al. 2021). Two sites with a mean annual capture of less than 1 individual per year were both in [arid] Kern County, California (Manzo et al. 2021).

In Washington, current population estimates are derived from mark/recapture efforts, population models, and the minimum numbers of northwestern pond turtles observed during surveys at all six northwestern pond turtle sites (Hallock et al. 2017, Bergh and Wickhem 2022, Washington Department of Fish and Wildlife 2022). The total minimum estimated population size for northwestern pond turtle in Puget Sound and the Columbia Gorge was approximately 481 and 281, respectively (although this total involves summing population sizes across years).

Critical Habitat

No critical habitat has been designated for the northwestern pond turtle.

Recovery Plan Information

No recovery plan exists for the northwestern pond turtle. However, the Service identified the following threats to the northwestern pond turtle in the species status assessment (Service 2023): (1) habitat loss and fragmentation; primarily from urbanization and agricultural conversion, (2) disturbance via recreational activities such as fishing, boating, and off-highway vehicle use, (3) alteration of natural hydrology through dam building, water diversions, stream channelization, etc., (4) predation by native and nonnative species (e.g., bullfrogs, largemouth bass), (5) competition with nonnative turtle species such as the red-eared slider (*Trachemys scripta elegans*), (6) roadkill mortality, (7) diseases including respiratory disease and shell disease, (8) commercial and private collection as pets and food, (9) toxicants such as pesticides, herbicides, and heavy metals, and (10) climate change impacts including increasing temperatures, drought, extreme flood events, and high severity wildfire.

The conservation needs for northwestern pond turtle includes conserving large blocks of suitable aquatic and associated upland habitat and maintaining connectivity by providing suitable habitat linkages for dispersal. Management activities that address threats to this species include controlling nonnative plants such as *Arundo donax*, controlling non-native aquatic predators and competitors such as fish, bullfrogs, crayfish, and red-eared sliders, and limiting predation by urban predators, such as dogs, ravens, and mammalian mesopredators such as coyote and raccoon (Service 2023, pp. 48, 50-51, 61).

Environmental Baseline

The northwestern pond turtle occurs throughout much of California, but also occurs in the states of Oregon and Washington. However, we have limited information regarding occurrences outside of California. Thus, the status description above also serves as the baseline for this consultation.

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San Francisco Garter Snake (*Thamnophis sirtalis tetrataenia*)

Listing Status

The San Francisco garter snake was listed as endangered on March 11, 1967 (32 FR 4001). No critical habitat has been designated for the San Francisco garter snake.

Life History and Habitat

San Francisco garter snakes are habitat specialists with several strict habitat requirements. Necessary habitat for San Francisco garter snakes includes densely vegetated standing freshwater habitats with some open water areas, open grassy uplands and shallow marshlands for breeding, and rodent burrows for hibernacula (shelters where they spend dormant winter months) and refugia (USFWS 2006). San Francisco garter snakes occur in the vicinity of standing water—chiefly ponds, lakes, marshes, and sloughs (USFWS 1985). However, temporary ponds and other seasonal water bodies are also used. Emergent and bankside vegetation such as cattails (*Typha* sp.), bulrushes (*Scirpus* sp.), spike rushes (*Juncus* sp.), and water plantain (*Alisma* sp.) apparently are preferred and used for cover (USFWS 1985; USFWS 2006). The interface between stream and pond habitats is used for basking, while nearby dense vegetation or water often provides escape cover. If floating algal mats or rush mats are available, snakes will use these, because they are apparently more secure basking sites (USFWS 1985). Shallow water near shore is essential from May to July to ensure the successful hatching and metamorphosis of amphibian prey items, particularly Pacific tree frogs and California red-legged frogs (USFWS 2006). San Francisco garter snakes also require open grassy uplands and shallow marshlands with adequate emergent vegetation for breeding (USFWS 2006). Flora composition in the upland habitat sites includes, but is not limited to, coyote bush (*Bacharis pillularis*), wild oat (*Avena fatua*), wild barley (*Hordeum* sp.), and various brome species (*Bromus* sp.). San Francisco garter snakes may prefer an "early successional" grassland/shrub matrix with brush densities ranging from one average-sized bush per 30 m^2 (323 sq. ft.) to one large bush per 20 m^2 (215 sq. ft.). By maintaining these ratios, there is sufficient cover from predators, while allowing for exposed surfaces to facilitate thermoregulation. The San Francisco garter snake also depends on ground-burrowing rodents to create burrows for snakes to use as hibernacula and refugia during the winter (USFWS 2006). The connectivity between aquatic and upland habitat is important and is currently threatened by development and infrastructure, including roads and highways (USFWS 2006).

San Francisco garter snakes mate in the spring or fall, and mating is concentrated in the first few warm days of March. Males actively search for females, which are presumably found by scent. Many males may simultaneously court a single female. The augmented frequency in spring mating is thought to be due to the increased likelihood of encountering a mate as individuals emerge from hibernacula and concentrate near aquatic hunting grounds. Mating occurs on open grassy slopes, typically in the morning. Ovulation generally occurs in late spring, pregnancy in early summer, and live birth of young sometime in July or August. Like many members of the genus *Thamnophis*, females can store sperm throughout the winter. Mating aggregations of San Francisco garter snake have been observed in late October and early November (USFWS 1985). Females are ovoviviparous (internal fertilization and young are born live, but no placental connection) and typically bear young in secluded areas, either hidden in dense vegetation or under some type of cover (Stanford University 2013). Litter sizes range from 3 to 85 young and average between 12 to 24 young (USFWS 1985), which are 12.5 to 20 cm (5 to 8 in.) in length at birth (Stanford University 2013). The lifespan of San Francisco garter snakes is unknown, but likely does not exceed 10 years (Stanford University 2013). The sex ratio of San Francisco garter snakes is also unknown, but in other garter snakes (*T. sirtalis*) subspecies, males outnumber females (USFWS 2006). Shallow water near shore is essential from May to July to ensure the successful hatching and metamorphosis of amphibian prey items, particularly Pacific tree frogs and California red- legged frogs (USFWS 2006). San Francisco garter snakes may depend on ground-burrowing rodents to create burrows, which snakes occupy during winter months (USFWS 2006).

San Francisco garter snakes are opportunistic carnivores that primarily feed on ranid frogs, including Pacific tree frogs (*Pseudacris regilla*) and California red-legged frogs (*Rana draytonii*) (USFWS 2006). Immature California newts (*Taricha torosa*), recently metamorphosed western toads (*Anaxyrus boreas*), bullfrogs, (*Rana catesbeiana*), threespine stickleback (*Gasterosteus aculeatus*), and mosquitofish (*Gambusia affinis*) have also been recorded in the diets of San Francisco garter snakes (USFWS 1985). Individuals on the Stanford University property have been documented to feed on invertebrates and possibly small rodents and birds in addition to amphibians and fish (Stanford University 2013). During the spring and early summer, feeding occurs near or in ephemeral ponds inhabited by Pacific tree frogs, the primary food source for San Francisco garter snakes during this time. Although juvenile San Francisco garter snakes may initially capture and consume Pacific tree frog metamorphs (tadpoles that have recently gained adult frog features) in upland habitat, they have principally been observed moving back to aquatic sites to feed on the young-of-year frogs once these wetter areas begin to dry up and the tree frogs begin to disperse. Mature individuals prey on Pacific tree frogs as well, although they also eat California red-legged frogs during the late summer months. The late emergence of California red-legged frogs allows for a necessary second cycle of feeding by adult San Francisco garter snakes after the Pacific tree frogs have retreated from the drying wetlands to upland aestivation areas (USFWS 2006). Young are born ranging from 13 to 20 cm (5 to 8 in.) in length, and adults can reach a maximum of 130 cm (51 in.) (Stanford University 2013). Prey items are usually captured in wetlands, either in emergent vegetation or in areas of shallow open water (Stanford University 2013; USFWS 2006). Bullfrogs, largemouth bass, and sunfish compete with San Francisco garter snakes for California red-legged frog and Pacific tree frog tadpoles (USFWS 2006).

San Francisco garter snakes are nonmigratory, but move between pond foraging habitats and upland wintering sites seasonally. Peak activity occurs between March and July, which may correspond with dispersal patterns of their prey. Radio tracking studies indicate that most individuals remain within 100 to 200 m (328 to 656 ft.) of pond foraging habitats and wintering upland sites. San Francisco garter snakes do not appear to move distances greater than 1 km (0.6 mi.), but they may disperse to new areas in pursuit of prey. Roads and highways may adversely affect dispersal and movement of the San Francisco garter snakes (USFWS 2006).

Population Status

Rangewide Status of the Species

The San Francisco garter snake is endemic to the San Francisco Peninsula and is known only from San Mateo County, California. Historically, San Francisco garter snakes were found on the San Francisco Peninsula from approximately the San Francisco County line, south along the eastern and western bases of the Santa Cruz Mountains at least to the Upper Crystal Springs Reservoir, and along the coast south to Año Nuevo Point, San Mateo County, California (USFWS 1985; USFWS 2006).

Current range is assumed to be equivalent to historic range. Recent surveys suggest that there has likely been very little decrease in the overall range of the San Francisco garter snake compared to its historic distribution; however, they have likely been extirpated from individual localities within what is considered to be the historic range/distribution (USFWS 2006).

Population Summary

There are six known populations of San Francisco garter snake: West of Bayshore, Laguna Salada, San Francisco State Fish and Game Refuge, Pescadero Marsh, Año Nuevo State Reserve, and Cascade Ranch. Little data exist regarding population trends, demographic features, and demographic trends for San Francisco garter snake. In the absence of reliable data regarding trends in the number of individuals in any given population, trends have been inferred from changes in habitat quality and quantity (USFWS 2006). Three of the six known populations appear to be declining, one is likely stable or increasing, and two are unknown (USFWS 2006).

The West of Bayshore population, near the San Francisco International Airport, appears to have declined between 1983 and the mid-1990s, possibly due to drought (USFWS 2006). The Laguna Salada population is declining due to saltwater intrusion, and the Pescadero Marsh population is likely declining due to saltwater intrusion (USFWS 2006). The population statuses are unknown for the San Francisco Fish and Game Refuge and Cascade Ranch populations (USFWS 2006). The population at Año Nuevo State Reserve is likely stable or increasing (USFWS 2006). Overall, the species has experienced a short-term decline of 10 to 30 percent (NatureServe 2015).

In 2020, a Status of the Species report provides an analysis of the current and future condition of 12 population complexes throughout the current range of the species, and also describes a 13th population complex that was formerly considered the most abundant population but is now considered to be extirpated (USFWS 2020).

Threats

Habitat loss and degradation of remaining habitat are the primary threats to the recovery of San Francisco garter snake. The degradation of habitat is primarily due to fragmentation resulting from expansion of infrastructure to support increasing residential and commercial developments, including new roads, improved utilities matrices, and recreational facilities. Secondarily, habitat is degraded by management practices conflicting with the needs of the San Francisco garter snake, including the allowance of serial succession, the increased use of perch ponds (shallow artificial water impoundments often used in San Mateo for irrigation) with decreasing use of stock ponds, the dredging of waterways, and recreational use of off-highway vehicles. Finally, fluctuations in water levels at reservoirs, flood control and channelization, and saline inundation events can result in further habitat degradation (USFWS 2006).

The amount of illegal collection of the San Francisco garter snake and its effects on the species is not clear. The San Francisco garter snake has been illegally collected by amateur herpetologists, and some amount of illegal collection likely still occurs. It is unclear what the impact of unauthorized take is on wild San Francisco garter snake populations, or what can be done to reduce this impact (USFWS 2006).

The epidemic of chytrid fungus (*Batrachochytrium dendrobatidis*), a potentially deadly parasite, poses a threat to most of the San Francisco garter snake's natural prey base. Outbreaks of chytrid fungus are increasing in size and severity throughout the world, perhaps due to recent climate changes that have resulted from abnormal weather patterns. Because of the rapid pace at which chytrid fungus can spread, a lethal outbreak on the Peninsula could be capable of extirpating entire cohorts of amphibians. In the absence of an adequate food source, such an event could lead to catastrophic declines in all garter snake populations range-wide (USFWS 2006).

Probable San Francisco garter snake predators include bullfrog (*Rana catesbeieana*), American crow (*Corvus brachyrhynchos*), red-tailed hawk (*Buteo jamaicensis*), red-shouldered hawk (*Buteo lineatus*), great egret (*Ardea alba*), snowy egret (*Egretta thula*), black crowned night heron (*Nycticorax nycticorax*), northern harrier (*Circus cyaneus*), great blue heron (*Ardea herodias*), long tailed weasels (*Mustela frenata*), and largemouth bass. In all cases, the extent that these predators influence San Francisco garter snake populations is not known (USFWS 2006).

Introduced high densities of mosquitofish have been observed attacking California red-legged frog tadpoles. The stress produced from these attacks was shown to slow develop of the tadpoles, limiting the viability of individuals. With a reduction in the population of California red-legged frogs at a location with mosquitofish, San Francisco garter snakes could experience a similar decline in numbers (USFWS 2006).

Parasites may have been responsible for several mortalities of juvenile San Francisco garter snakes captured at the West of Bayshore location. Parasitic species encountered include a tapeworm, several flagellate protists, and eight different occurrences of nematode worms. Mosquitofish throughout the northern San Francisco Bay Area may serve as hosts for parasitic tapeworms and thorny-headed worms. These parasites could possibly be transmitted to animals that prey on mosquitofish, which include various ranid species and potentially San Francisco garter snakes (USFWS 2006).

One of the greatest threats to the San Francisco garter snake is the reduction of habitat quality resulting from the elimination of disturbance events throughout the Peninsula. Primarily, this is based on changes in management that encourage seral ecosystems. Dynamic grass-dominated uplands provide for, and are potentially maintained by, burrowing rodents that create tunnel systems used by San Francisco garter snakes for hibernacula during the winter months. The loss in recent years of ecological disturbance throughout the majority of San Mateo County has made it possibly for brush species to dominate former grasslands, potentially precluding burrowing animals. Fire suppression has allowed for the domination of these woody species across the coastal landscape, limiting the extent of grasslands that were likely important movement corridors between aquatic habitats. Augmented production levels of cattails also contribute to the loss of open water in aquatic systems. Additionally, the loss of traditional grazing practices on public lands has allowed for the accumulation of dense brush-dominated canopies across the remaining grasslands, which may decrease habitat suitability for the San Francisco garter snake. Reintroducing domestic grazing to grasslands could improve and restore habitat conditions for the San Francisco garter snakes (USFWS 2006). The perpetuation of seral conditions also has negatively impacted suitable aquatic habitat. Cattails (*Typha* sp.) and other emergent aquatic vegetation species may increase siltation rates in freshwater marshes due to the high water demands of these species, as well as their ability to trap overland runoff. The augmented production level of cattails contributes to the loss of the open-water component in aquatic systems. Open water, combined with emergent vegetation, creates a matrix of habitat elements thought to be necessary for Pacific tree frog and California red-legged frog populations—which are crucial for San Francisco garter snake aquatic habitat—already threated by salinization events and the presence of bullfrogs (USFWS 2006).

Increased presence of invasive species can compete for resources with the San Francisco garter snake or hunt individual San Francisco garter snakes directly. Bullfrogs, largemouth bass (*Micropterus salmoides*), and sunfish (Centrarchidae) consume California red-legged frog and Pacific tree frog tadpoles, and bullfrogs may prey directly on San Francisco garter snakes (USFWS 2006).

Steep banks and earthen dams associated with artificial water impoundment reduce the suitability of an area for San Francisco garter snakes. High grade slopes may reduce basking opportunities because of the absence of level areas in close proximity to dense vegetation. Reservoirs are often absent of adequate vegetation, exposing both the snake and its prey to additional predators (USFWS 2006).

Roads and highways may adversely affect dispersal and movement of San Francisco garter snakes. Reptiles often use roads for thermoregulation, which can lead to mortality due to vehicular strikes. Highways may also adversely affect dispersal and movement of amphibian prey species (USFWS 2006).

Five-Year Status Review

There have been two five-year status reviews for this species: one on October 2, 2006 and a more recent one on May 21, 2020. The latest five-year status review conducted a comparison of current condition of the San Francisco garter snake to the recovery criteria for the species. There is only one population with over 200 individuals, and populations with the smallest abundance estimates may have shifted sex ratios (USFWS 2020). Thus, the downlisting criteria for this species are not met (USFWS 2020). The review concluded that the San Francisco garter snake would remain an endangered species (USFWS 2020).

Critical Habitat

No critical habitat has been designated for the San Francisco garter snake.

Recovery Plan Information

On September 11, 1985, a Recovery Plan was issued for the San Francisco garter snake (USFWS 1985).

Reclassification Criteria

A primary objective of the 1985 Recovery Plan is to protect and maintain a minimum of six San Francisco garter snake populations, each containing 200 adult snakes (1:1 sex ratio). If this goal is obtained and maintained for 5 consecutive years for six of the ten populations, consideration for threatened status would be appropriate. The six significant populations include the West of Bayshore property (San Francisco International Airport), San Francisco State Fish and Game Refuge property (San Francisco Public Utilities Commission), Laguna Salada/Mori Point property (City of San

Francisco/National Park Service), Pescadero Marsh and Año Nuevo State Reserve properties (California State Parks), and Cascade Ranch property (private landowner) (USFWS 1985; USFWS 2006).

Delisting Criteria

Protect and maintain a minimum of ten San Francisco garter snake populations with approximately 200 adults (1:1 sex ratio) at each site within the snake's historic range for 15 consecutive years; delisting can then be considered. The recovery criteria include the six significant populations and the creation of four populations at undefined sites (USFWS 1985; USFWS 2006).

The recovery plan proposed that conservation agreements be signed with each of the landowners controlling the lands containing the six significant populations identified in the plan. However, no agreements have been completed to date and the additional four populations proposed in the recovery plan have not been identified. Additionally, although the precise population ratios of San Francisco garter snakes are unknown, studies of the eastern garter snake (*Thamnophis sirtalis sirtalis*) and the red-sided garter snake (*T.s. infernalis*) indicate that those sub-species do not exhibit 1:1 sex ratios, with males outnumbering females in the wild. If the sex ratios of San Francisco garter snakes are similar to the eastern and red-sided garter snakes, then a sex ratio of 1:1 may not be the appropriate criterion (USFWS 2006). In response to the issues described above, an updated recovery outline was prepared by the U.S. Fish and Wildlife Service (USFWS) in July 1995. In 2004, the Sacramento Fish and Wildlife Office established a San Francisco garter snake working group comprising USFWS employees familiar with current issues facing the species. The group's purpose is to design and implement specific conservation actions that could be performed prior to, and concurrent with, updating the recovery plan. The group is preparing an interim recovery implementation document consistent with the 1995 recovery outline to assist in guiding recovery actions until a revised recovery plan can be developed (USFWS 2006).

Recovery Actions

- Use legal authorities to protect San Francisco garter snake and its habitat by enforcing laws and regulations to promote the conservation of the San Francisco garter snake and its habitat, evaluating success of law enforcement, and proposing appropriate new regulations or revisions (USFWS 1985).
- Protect the six known San Francisco garter snake colonies through appropriate management. These colonies include Pescadero Marsh Natural Preserve, Año Nuevo State Reserve, San Francisco State Fish and Game Refuge, the San Francisco Airport Millbrae site, and at least four additional populations (USFWS 1985).
- Assess population trends and make modifications in management plans if necessary. This includes developing population estimation techniques and conducting population surveys as necessary at Pescadero Marsh Natural Preserve, Año Nuevo State Reserve, San Francisco State Fish and Game Refuge, the Millbrae/Airport site, the Laguna Salada site, Cascade Ranch, and any additional sites discovered (USFWS 1985).
- Identify additional recovery needs for the San Francisco garter snake and modify prime objective/management plans accordingly. This includes obtaining life history data necessary to manage and eventually delist the San Francisco garter snake, determining habitat relationships, reevaluating introgression between the red-sided garter snake and the San Francisco garter snake, and identifying essential habitat (USFWS 1985).
- Provide for public information and awareness by providing onsite interpretive programs on public lands, preparing a small brochure on the San Francisco garter snake and the recovery program, and developing a slide-tape program for public presentations (USFWS 1985).
- Develop an updated recovery plan and an expanded San Francisco garter snake working

group (USFWS 2006).

- Encourage conservation among private landowners (USFWS 2006).
- Continue ongoing habitat restoration and enhancement for wild populations (USFWS 2006).
- Complete captive holding facilities for use in head starting programs, in the restoration of worldwide zoo populations, and as temporary lodging during habitat maintenance (USFWS 2006).
- Increase research of population trends, demography, and phylogenetics (USFWS 2006).
- Increase law enforcement at vulnerable locations (USFWS 2006).

Environmental Baseline

The San Francisco garter snake occurs in the San Francisco Peninsula and is known only from San Mateo County, California. Please refer to information above for the environmental baseline.

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Southwestern Pond Turtle

Listing Status

The southwestern pond turtle is not listed under the Act; however, it is currently proposed threatened and under federal review for listing under the Act (88 FR 68370).

The Service was petitioned to list 53 species of reptiles and amphibians, including the western pond turtle (*Actinemys marmorata*), as threatened or endangered under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531–1543), in July 2012. On April 10, 2015, we published a 90-day finding that the petition presented substantial scientific or commercial information indicating that listing may be warranted for the western pond turtle (80 FR 19262–19263). Since then, the western pond turtle was split into two separate species, the northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*). The species status assessment was issued in April 2023 (Service 2023), compiling biological information and condition on both species.

Life History and Habitat

The southwestern pond turtle is a medium-sized turtle, with adults ranging from 4.3 to 7.1 inches long (maximum carapace (shell) length) (Germano and Riedle 2015, p. 104). Females tend to have a smaller head, less angled snout, taller and rounder carapace, flat (rather than concave) plastron (underside of shell), and thinner tail as compared to males (Holland 1994, pp. 2-4; Rosenberg et al. 2009, p. 10). Colors and markings vary geographically and by age with most appearing olive to dark brown, or blackish, occasionally without pattern but usually with a network of spots, lines, or dashes of brown or black that often radiate from growth centers of shields (Bury et al. 2012, p. 4; Stebbins and McGinnis 2018, pp. 204–205). Hatchlings are generally a brown-olive color with visible mottling on the head and neck (Hays et al. 1999, p. 2) that darken with age. Hatchlings are 0.98 to 1.22 inches long carapace length (CL) (approximately the size of an American quarter) (Bury et al. 2012, pp. 4, 17). The shell of hatchlings is soft and pliable, and the tail is nearly as long as the shell (Ashton et al. 1997, p. 3; Stebbins and McGinnis 2018, p. 205). The shell becomes fairly hard around three to four years of age (Bury et al. 2012, p. 4). Eggs are off-white, elliptical-oval shaped, and range from 1.26 to 1.65 inches long and from 0.71 to 0.98 inch in diameter (Bury et al. 2012, p. 15).

Seeliger (1945, entire) first proposed geographic differentiation of western pond turtles into northern and southern subspecies based on differences in coloration and the presence and shape of the inguinal scute, the plate where the carapace joins the plastron at the groin. Since then, the western pond turtle was split into two separate species, the northwestern pond turtle (*Actinemys marmorata*) and southwestern pond turtle (*Actinemys pallida*). Recent genetic results corroborate the morphologic distinctiveness (presence/absence of inguinal scutes) as one of the components differentiating northwestern and southwestern pond turtles (Shaffer and Scott 2022, p. 9).

Southwestern pond turtles are semi-aquatic, having both terrestrial and aquatic life history phases. Eggs are laid in upland terrestrial habitat, and hatchings, juveniles, and adults use both terrestrial and aquatic habitat. Terrestrial environments are required for nesting, overwintering and aestivation (warm season dormancy), basking, and movement/dispersal. Aquatic environments are required for breeding, feeding, overwintering and sheltering, basking, and movement/dispersal. Perennial (i.e., year-round) and intermittent (i.e., not year-round) bodies of water occur throughout the range. Some are flowing/lotic (e.g., streams, rivers, irrigation ditches), while others are not flowing/lentic (e.g., ponds, lakes, and reservoirs).

Preferred aquatic conditions are those with abundant basking sites, underwater shelter sites (undercut banks, submerged vegetation, mud, rocks, and logs), and standing or slow-moving water (Holland 1991, pp. 13–14; Reese and Welsh Jr. 1998a, p. 852; Hays et al. 1999, p. 10; Bury and Germano 2008, p. 001.4; Ernst and Lovich 2009, p. 175). Western pond turtles inhabiting lentic aquatic habitat, such as ponds, lakes, and slack water habitats, often overwinter within the aquatic environment, burying themselves within the bottom substrate, such as mud. Various depths of water provide western pond turtles with habitat necessary for overwintering and hatchling growth. Primary habitat for hatchlings and young juveniles is shallow water with dense submerged vegetation and logs, which most likely provides shelter, prey, and thermoregulatory requirements or other functions for survival (Holland 1994, pp. 1-14, 2-12; Rosenberg and Swift 2013, p. 119). Western pond turtles are extremely wary and will rapidly flee from basking sites into the water when disturbed by the sight or sound of people at distances of greater than 328 feet (Bury and Germano 2008, p. 001.5).

Nesting habitat is in close proximity to aquatic habitat and is typically characterized as having sparse vegetation with short grasses and forbs and little or no canopy cover to allow for exposure to direct sunlight (Holland 1994, p. 2-10; Rathbun et al. 2002, p. 232; Rosenberg et al. 2009, pp. 16–17; Riensche et al. 2019, p. 97). Females excavate nests in compact, dry soils that are 9.84 to 1,312 feet from water (Holland 1994, p. 2-10; Holte 1998, p. 54). Additional features of nesting habitat/sites that may be important include aspect, slope, and vegetation.

Overwintering is a state of little to no activity (e.g., brumation) that occurs during the cooler months of the year and can occur in either upland or aquatic environment (Holland 1994, p. 2-7; Ultsch 2006, pp. 341, 356). Southwestern pond turtles also use upland habitat for migration (intra-population (within local populations) movements occurring between aquatic and upland environments), dispersal (movement between populations/watersheds), and aestivation. Aestivation is a period of inactivity, usually in response to the hottest time of year or dry conditions (Hays et al. 1999, p. 7) that occurs in terrestrial habitat.

The western pond turtle is omnivorous and considered a dietary generalist (Holland 1994, p. 2-5), consuming a wide variety of food items. Prey resources are primarily found within water but can be captured or scavenged on land. Food captured or scavenged on land must be brought back to water for consumption, as they appear to be unable to swallow in the air (Holland 1994, p. 2-6). Animal matter appears to constitute a larger portion of the diet than plant material (Bury 1986, pp. 518–520; Holland 1994, pp. 2-5–2-6). Stomach contents reveal the diet consists of small aquatic invertebrates, with small vertebrates (fish, tadpoles, and frogs), carrion, and plant material (Bury 1986, p. 516; Holland 1994, pp. 2-5–2-6). Nonnative predators include American bullfrogs (*Lithobates catesbeianus*; hereafter bullfrogs) and invasive fish, such as large and smallmouth bass (*Micropterus* sp.; hereafter bass). Native predators of western pond turtles include raccoons, skunks, foxes, coyotes. mink, herons, river otters, burrowing small mammals, and giant water bugs.

Southwestern pond turtles mature slowly and have low fecundity but are potentially long-lived. In southern California, the smallest known reproductive female was approximately 4.37 inches carapace length and at least 6 to 7 years old (Holland 1994, p. 5-2). Courtship behaviors have been observed from April through November, with mating observed in May through September (Holland 1991, p. 23). Oviposition usually occurs from May through July (Bury et al. 2012, p. 15). Clutch size for western pond turtles varies from 1 to 13 eggs, and is positively correlated with body size (Holland 1994, p. 5-2; Holte 1998, p. 5). Incubation time is approximately 80 to 126 days (Holland 1994, pp. 2-10, 5-7). Western pond turtles exhibit temperature-dependent sex determination (TSD) during incubation (Ewert et al. 1994, p. 7). In California, female hatchlings were more likely when 30 percent of the sex-determining period occurred above 84 degrees Fahrenheit (Christie and Geist 2017:49). In addition, lower fluctuations in temperature resulted in development of males, whereas females developed in nests with high and low temperature fluctuations. In southern and central California, some hatchlings may emerge from the nest chamber in late-summer to early-fall, whereas others overwinter in the nest chamber and emerge in spring (Holland 1994, p. 2-10). The maximum lifespan of western pond turtles is unknown. However, they are long-lived species after reaching adulthood, with some northwestern pond turtles living to at least 55 years of age (Bury et al. 2012, p. 17).

Home range size and configuration varies between age class, sex, and location. Measured home ranges of western pond turtles average 2.5 acres for males, 0.7 acre for females, and 1 acre for juveniles (Bury 1972, entire). Female pond turtles in two southern California streams had home ranges that were longer and smaller (Goodman and Stewart 2000) than those observed by Bury (1972, entire), likely because the streams in southern California tend to be narrower so pond turtles have to move further distances to obtain sufficient resources. Western pond turtles are capable of dispersing substantial distances, although large overland movements are uncommon. The longest overland distance recorded in an area considered to be under the best circumstances (mild climate and short distances between water features), was a single individual travelling 3.11 miles. Holland (1994, p. 2-9), also states that no movements between drainages were detected from three other sites with over 1,100 hundred captures and recaptures over a 7-year period. During an extreme drought, Purcell et al. (2017, pp. 21, 24) documented a 1.62 miles straight-line distance movement overland in a radio-tagged turtle, with a minimum total distance of 2.05 miles moved before the individual found water.

Population Status

The historical range of western pond turtles extends along the Pacific coast from British Columbia, Canada to the northern part of Baja California, Mexico, primarily west of the Sierra Nevada and Cascade ranges (Ernst and Lovich 2009, p. 173; Stebbins and McGinnis 2018, p. 205). Western pond turtles have been found at sites from brackish estuarine waters at sea level up to 6,719 feet (Ernst and Lovich 2009, p. 176) but mostly occur below 4,980 feet (Stebbins and McGinnis 2018, p. 205). The range of the southwestern pond turtle is restricted to those populations inhabiting the central Coast Range south from the middle of Monterey Bay to the species' southern range boundary in Baja California. A new population found south of the nearest reported population represents a range extension of 59.34 miles (and the only oasis population within the Central Desert ecoregion in Baja California) (Valdez-Villavicencio et al. 2016, p. 265).

Shaffer and Scott (2022, entire) clarified areas of previous uncertainty immediately south, east, and west of the San Francisco Bay, where there were no specimens used in Spinks et al. (2014, p. 2233) when describing northwestern and southwestern pond turtles, and the range around the San Francisco Bay presented in Thomson et al. (2016, p. 297). Based on these genomic data, Shaffer and Scott recommended that the border along the coast between the two species was in the middle of Monterey Bay (Shaffer and Scott 2022, p. 5). It also clarified the contact zone between the two species at the edge of the South Coast Ranges where they meet the floor of the Central Valley; although there are individuals with genetics from both species along the area where the species come into contact in this area, it appears that the boundaries are adjacent but do not overlap (Shaffer and Scott 2022, pp. 4–5).

Critical Habitat

No critical habitat has been designated for the southwestern pond turtle.

Recovery Plan Information

Habitat destruction and alternation are primary threats to the southwestern pond turtle. Extensive land conversion due to urbanization and agriculture has resulted in substantial losses to both upland and aquatic habitats across the range (Holland 1994, p. 1-23; Hays et al. 1999, pp. ix, 31; Spinks et al. 2003, p. 258; Bury and Germano 2008, p. 001.6; Rosenberg et al. 2009, p. 40; Thomson et al. 2016, pp. 300–301). As a result, a large fraction of the remaining habitat in southern California existing only as patches with little suitable upland habitat available for nesting (Thomson et al. 2016, p. 301). Overall, the range of the southwestern pond turtle is fragmented to varying degrees by human activities, with some sites extirpated, and in many cases, only small, isolated groups or individuals remaining (Holland 1991, p. 13).

Aquatic resources used by the western pond turtle have experienced high levels of loss, alteration, and degradation throughout the range of the two species (Reese and Welsh Jr. 1998b, p. 505; Germano 2010, p. 89). A substantial portion of the losses of aquatic habitat are due to anthropogenic water use (e.g., dams and diversions for the purposes of providing water for human use). Moreover, within the historical range of the western pond turtle, an extensive system of hydrologic infrastructure, including dams, reservoirs, diversions, and aqueducts, supports extensive agricultural and municipal water uses, and provides domestic water to many densely populated areas (Lund et al. 2007, p. 43; Hanak et al. 2011, pp. 19–69). These alterations include stream channelization, altered flow regimes, groundwater pumping, water diversions, damming, and water regulation for flood risk management (flood control), which affect hydrology, thermal conditions, and structure of western pond turtle aquatic and upland habitat.

Loss of upland habitat adjacent to southwestern pond turtle aquatic habitat can isolate pond turtles from surrounding populations and eliminate nesting sites, thus limiting the ability to successfully reproduce (Holland 1994, entire; Spinks et al. 2003). Agricultural areas and grazing pastures provide suitable habitat for nesting southwestern pond turtles, but certain practices, such as plowing and irrigation, could destroy nests (Crump 2001, entire). Western pond turtle eggs have permeable shells that have been observed to rupture after absorbing excess moisture, killing the pond turtle embryo (Feldman 1982, p. 10). For example, this could be a problem in urban areas that are irrigated (Spinks et al. 2003, p. 263).Roads can affect western pond turtle viability because of vehicles killing or injuring individuals or disturbing basking behavior, and by reducing connectivity between populations, which reduces migration between upland and aquatic habitat (Rosenberg et al. 2009, p. 41; Nyhof 2013, p. 43; Thomson et al. 2016, p. 301; Nicholson et al. 2020, entire; Manzo et al. 2021, p. 494, S1 text supplement).

Development can also indirectly lead to habitat degradation and/or mortality as a result of down cutting and erosion, introduction of non-native plants and animals, water pollution, and recreational activities (Holland 1991, entire). Increased runoff from irrigation results in down cutting and erosion which can eliminate pools, basking sites, and refugia used by pond turtles and isolates the aquatic environment from the surrounding upland environment. Invasion by nonnative aquatic plant species, such as Arundo spp. can alter the stream hydrology and displace emergent aquatic vegetation that provides refuge for juvenile turtles. Introduced non-native and urban-related animals include predators (e.g., non-native fish, bullfrogs, crayfish, dogs, and corvids) and competitors (e.g., non-native turtles, such as the red-eared slider).

Recreational activities such as hiking, biking, fishing, boating, and off-highway vehicles, and the associated disturbance within or adjacent to aquatic and nest habitats, can affect western pond turtles in a variety of ways, depending on the region and type of recreation. Some forms of recreation may cause mortality of individuals through trampling, while others degrade habitat, disturb pond turtle behavior, and/or contribute to other threats. For example, recreational activities may interact with the threat of collection because humans may encounter the species while engaging in other activities. Western pond turtles are extremely wary and will rapidly flee from basking sites into the water when disturbed by the sight or sound of people at distances of greater than 328 feet (Bury and Germano 2008, p. 001.5).

Desiccation of waterways from drought has led to declines and extirpations of western pond turtle populations by negatively affecting the quality and/or quantity of its aquatic habitat, impacting survival, recruitment, and connectivity, and exacerbating the effects of other threats. Western pond turtle mortality during drought is well documented, and appears to occur as a result of drought-induced starvation (Lovich et al. 2017, p. 7) and/or drought-induced predation (Purcell et al. 2017, p. 21). Extended drought occurring during 1986–1987 through at least 1991 caused major population declines and extirpations in many areas, but most significantly in southern and central California (Holland 1991, p. 65). During this time, turtles in small to moderate sized watercourses were fairly abundant until 1988–1989, but as water continued to dry, resulting in major increases in distance to the next water source, turtles concentrated in the few remaining pools exhausted available prey, and were exposed to increased predation.

During normal drought conditions, when water levels are low, western pond turtles can aestivate in upland habitat or move to another water body if one is within migration and/or dispersal distance. Aestivating southwestern pond turtles remained in upland habitat for approximately 7 months (mean 201 days, range 154 to 231 days) during the 2011–2012 drought (Belli 2016, p. 57), suggesting that even in a severe drought, individuals could remain alive to repopulate the water body once conditions become suitable again (see Purcell et al. 2017, entire). However, extended drought conditions and/or increased frequency of droughts, could have substantive effects on populations, and other synergistic effects could also make repopulation by aestivating individuals unlikely. In addition, because females often forego nesting when conditions are unfavorable, extended drought can result in reduced reproduction and recruitment opportunities.

The conservation needs for western pond turtles include conserving large blocks of suitable aquatic and associated upland habitat and maintaining connectivity by providing suitable habitat linkages for dispersal. Management activities that address threats to this species include controlling nonnative plants such as *Arundo donax*, controlling non-native aquatic predators and competitors such as fish, bullfrogs, crayfish, and red-eared sliders, and limiting predation by urban predators, such as dogs, ravens, coyotes, and raccoons (Service 2023b, pp. 48, 50-51, 61). Bullfrogs have been introduced into western pond turtle habitat and influence viability of the species by increasing predation pressure on hatchlings and small juveniles, and thus are considered to have the largest impact on western pond turtle demography (Service 2023b, pp. 87-88, 89-90, 95). Because of the potential threat posed by road mortality, measures such as the installation of low-lying fine-mesh fence or barrier fencing in areas likely to be used by pond turtles may help minimize this source of mortality. In addition, because pond turtles may be collected as pets or non-native red-eared sliders purchased from the pet store could be released into the wild, public education regarding these effects would benefit this species.

Environmental Baseline

Since the southwestern pond turtle occurs primarily within California with limited information available regarding its status in Mexico, the status description above also serves as the environmental baseline for this consultation.

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Birds

California Least Tern (*Sterna antillarum browni*)

Listing Status

The California least tern was federally listed as endangered on March 8, 1969 (34 FR 5034).

Life History and Habitat

California least terns prefer beachfront habitat with sparse or low-lying vegetation and low disturbance from humans and mammalian predators. California least terns preferentially nest on unconsolidated fine to coarse sand that is interspersed with larger fragments of material and sparse ground vegetation (i.e., 0 to 20 percent total ground cover less than 16 inches tall) (Service 2020). Foraging habitat used by terns includes nearshore waters, estuarine channels, narrow bays, and other shallow water marine habitat. Typical foraging habitat is within two miles of colony sites in "relatively shallow nearshore ocean waters in the vicinity of major river mouths..." (Atwood and Minsky 1983). Information on the wintering habitat of California least terns is limited, and further study is required to understand it.

California least terns feed primarily on small fishes captured in estuaries, embayments, and shallow, nearshore waters, particularly at or near estuaries and river mouths and on occasion krill and other invertebrates. The depth of the water where the species forages is generally less than 25 feet (Service 2020).

The California least tern nests primarily between May and August. In recent years, birds have arrived at nesting sites in the last week of March to the first or second week of April (Service 2020). Breeding commences at 2 to 3 years of age. California least terns exhibit a high degree of nest site fidelity from year to year. Individuals often return to breed where they previously bred successfully or to their natal sites (i.e., where they hatched) significantly more than would be predicted if birds nested randomly (Service 2020).

Population Status

Within the United States, the California least tern was known from nesting sites located within or near 15 nesting bays, estuaries, or beaches at the time of listing in 1969. Nesting sites extended from Bair Island in San Mateo County to the Tijuana River Estuary in San Diego County. At the time of listing, there were a minimum of 256 pairs of least terns. Since listing, the California least tern's breeding range has extended northward, with additional nesting sites discovered or colonized in the San Francisco Bay area, and the Sacramento River Delta. California least tern also nest on the Pacific side of Baja California, although they have been in decline in this area since the early 2000s. In addition, isolated instances of nesting have been detected at more inland sites scattered in the Central Valley, and in one instance in Arizona (Service 2020).

California least tern nesting is confined to 29 areas that total approximately 1,204 acres of habitat along the California coast. The number of California least tern pairs nesting at each nesting area is highly variable. For example, in 2016, the number of pairs estimated nesting at sites in California ranged from 1 (e.g., Sacramento Bufferlands, Pittsburg Power Plant) to 804 (e.g., Santa Margarita River–North Beach South). In 2016, the majority (approximately 85 percent) of California least tern breeding pairs were concentrated in southern California within coastal Ventura, Los Angeles, Orange, and San Diego counties, and almost half of the birds in San Diego County nested within lands owned and managed by Marine Corps Base Camp Pendleton (Service 2020).

Recovery Plan Information

A revised recovery plan was completed for California least tern on September 27, 1985 (Service 1985). However, the criteria to assess recovery of the California least tern provided in the 1985 recovery plan do not reflect the most current information available. The recovery criteria are not threat-based, which is current policy for recovery plan development, but the criteria speak indirectly to the threats outlined in the five-factor analysis section of the 2020 5-year review. Overall, progress is being made toward satisfying the recovery criteria. However, as we concluded in the 2020 5-year Review and based on recent data, the recovery plan should be revised and updated to provide threats-based recovery criteria and address the other shortcomings of the recovery plan. Areas of the plan that need updating include inclusion of Mexico populations of California least terns, further analysis of the fledgling per pair ratio, and future impacts from a changing climate, such as seal level rise (Service 2020).

A total of 4,095 breeding pairs were reported in 2017, supporting that the species has met and exceeded Objective 1 of the recovery plan (requiring over 1,200 nesting pairs) in the United States. With 13 Coastal Management Areas and an additional three nesting areas that support secure California least tern nesting areas, Objective 2 from the recovery plan has been partially met. However, there are still not enough secured and viable breeding sites at the San Francisco and Mission Bay coastal management areas to meet this criterion. Objective 3 has not been met as productivity remains significantly below that recommended (average of 1.0 fledgling per pair) and reported values have declined significantly since the 2006 5-year review. The sustained poor productivity over the last decade is of concern and warrants further attention (Service 2020).

Environmental Baseline

The California least tern occurs primarily in California, but also occurs along the Pacific coast of Baja and on wintering grounds outside of California. However, we have limited information regarding occurrences outside of California. Thus, the status description above also serves as the baseline for this consultation.

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California Ridgway's Rail (*Rallus obsoletus obsoletus*)

Listing Status

The California Ridgway's rail was federally listed as endangered in 1970 (35 FR 16047, Service 1970). Critical habitat has not been proposed or designated.

Based on the work of Maley and Brumfield (2013), the American Ornithologist's Union (AOU) Committee on Classification and Nomenclature accepted in its 55th Supplement to the AOU Check-list of North American Birds (Chesser *et al.* 2014), revisions to the specific assignments under the genus *Rallus*. Among those changes, the species *R. obsoletus* (Ridgway's rail) and *R. crepitans* (Clapper rail) were split from *R. longirostris*, and *R. longirostris* was deleted. The U.S. Fish and Wildlife Service provided taxonomic corrections for the species on July 31, 2023, changing the common name and scientific name of the species from the California clapper rail (*Rallus longirostris obsoletus*) to the California Ridgway's rail (*Rallus obsoletus obsoletus*) (88 FR 49310).

Life History and Habitat

Historically, the California Ridgway's rail was abundant in all tidal salt and brackish marshes in the San Francisco Bay vicinity, as well as in all of the larger tidal estuaries from Marin to San Luis Obispo counties. Current distribution is restricted almost entirely to the marshes of the Bay Area and where the only known breeding populations occur. California Ridgway's rails occur almost exclusively in tidal salt and brackish marshes with unrestricted daily tidal flows, adequate invertebrate prey food supply, well developed tidal channel networks, and suitable nesting and escape cover for refuge during extreme tides. They exhibit strong site fidelity and territorial defense and are considered sensitive to disturbance. They tend to have relatively small average home ranges of 4.7 hectares (11.6 acres) and core use areas of 0.9 hectare (2.2 acres).

In south and central San Francisco Bay, and along the perimeter of San Pablo Bay, rails typically inhabit salt marshes dominated by *Sarcocornia pacifica* and *Spartina foliosa*. *Spartina* ssp. dominates the lower marsh zone (marsh plain) throughout the south and Central Bay (DeGroot 1927, Hinde 1954, Harvey 1988). *Sarcocornia pacifica* dominates the middle and sometimes upper marsh zone throughout the South and Central Bay, with *Distichlis spicata*, *Jaumea carnosa* (fleshy jaumea), *Frankenia salinia* (alkali-heath), and others mixing with occasional *Sarcocornia pacifica* in the high marsh zone. *Grindelia stricta* var. *angustifolia* occurs along the upper edge of tidal sloughs throughout the entire San Francisco Bay Estuary.

In the North Bay, Ridgway's rails also occur in tidal brackish marshes that vary significantly in vegetation structure and composition, ranging from salt-brackish marsh to fresh-brackish marsh transitions. *Bolboschoenus maritimus* (alkali bulrush), an indicator of salt-brackish marsh transitions, is sub-dominant to dominant in low marsh and lower middle marsh plains. *Schoenoplectus acutus* and *Schoenoplectus californicus* (tules), *Schoenoplectus americanus* (Olney's bulrush), and *Typha* spp. dominate the low marsh zone of fresh-brackish marsh transitions, while fresh-brackish marsh plain vegetation is a diverse, patchy mixture of dominant *Distichlis spicata*, *Jaumea carnosa*, salt rush (*Juncus arcticus* ssp. *balticus*, *Juncus lesueurii*), and numerous native and non-native herbs, grasses, and sedges. *Grindelia stricta* var. *angustifolia* (and its hybrid *Grindelia* x *paludosum* in Suisun Marsh) is the widespread dominant of high marsh vegetation in brackish marshes today, but it occurs with other tall, dense sub-shrubby or herbaceous native vegetation along marsh edges and creek banks, such as *Baccharis douglasii* (salt marsh *baccharis*), *Euthamia occidentalis* (goldenrod), *Achillea millefolium* (yarrow), *Scrophularia californica* (bee-plant), and asters (*Symphyotrichum lentum*, *Symphyotrichum chilensis*, and intermediates, *Symphyotrichum sublantus* var. *ligulatus*; now uncommon). The historically diverse high brackish marsh vegetation probably provided ample high tide flooding refuges for Ridgway's rails.

The breeding period of the California Ridgway'srail is prolonged. Pair bonding and nest building are generally initiated by mid-February. Nesting may begin as early as late February or early March (Evens and Page 1983 as cited in USFWS 2013a), and extend through July in the South Bay, and into August in the North Bay (DeGroot 1927). The end of the breeding season is typically defined as the end of August, which corresponds with the time when eggs laid during re-nesting attempts have hatched and young are mobile.

Additional information about the California Ridgway's rail biology and ecology is available in the Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California, available at: https://ecos.fws.gov/docs/recovery_plan/TMRP/20130923_TMRP_Books_Signed_FINAL.pdf (Service 2013a).

Population Status

There is currently no USFWS range-wide California Ridgway's rail monitoring program or protocol nor habitat suitability metrics available to evaluate recovery progress of the species and its habitat. The 2020 5-Year Review used the Invasive *Spartina* Project habitat information and survey data to use as indices of both site-level and range-wide changes in rail population abundance and habitat suitability. Call count data was used as an index/estimate of annual rail abundance and trend at surveyed sites and the habitat assessment information as an index/estimate of trend for the area of habitat described as suitable for surveyed sites. The USFWS did not attempt to estimate or model rail densities or abundance for unsurveyed areas. Accordingly, because the call count surveys and habitat assessments did not include all possible habitat, we consider our estimates of population abundance and habitat area to be minimum estimates and actual population abundance is likely higher.

Overall, the estimated range-wide California Ridgway's rail population has increased since the 2013 5-Year Review. The 2013 5-Year Review and Recovery Plan referenced the Liu et al. (2009) estimate of an average population of 1,426 rails between 2005 and 2008 (for comparison, the USFWS currently estimated average for the same time period was 890 rails). The index estimated range-wide annual population for 2011 was 899 rails and for 2018 was 1,192 rails (USFWS 2020).

At a recovery unit scale, the increase since 2011 in population estimate was observed in both the San Pablo Bay and Central/South San Francisco Bay Units (USFWS 2020). The San Pablo Bay Recovery Unit had some increase in rail numbers between 2011 and 2018, with 290 birds in 2011 and 353 birds in 2018, but ended the time period nearly slightly lower proportionately, supporting about 32 percent and 30 percent of the range-wide population in 2011 and 2018, respectively (USFWS 2020). The Central/South San Francisco Bay unit experienced a greater increase in rail numbers between 2011 and 2018, with 607 birds in 2011 and 839 birds in 2018. The proportion of the range-wide population in the Central/South San Francisco Bay Recovery Unit also increased slightly, supporting about 67 percent and 70 percent of the range-wide population in 2011 and 2018, respectively (USFWS 2020). The Suisun Bay Recovery Unit did not experience an increase, with rail counts in that unit remaining at or near zero for the entire data series (USFWS 2020). It is noted that establishment of sustainable populations in the Suisun Bay Unit at levels prescribed in the Recovery Plan may be considered indicative of the species occupying its full range under optimal habitat and population conditions (Service 2013a, 2013b).

The 2020 5-Year Review analysis suggests that while the California Ridgway's rail population appears to have increased across both the San Pablo Bay and Central/South San Francisco Recovery Units since the 2013 5-Year Review, the distribution of rails has become increasingly concentrated to fewer sites and less habitat area. No change in the species' listing status was recommended in this 5-year review.

Threats to the species include, but are not limited to, habitat destruction and modification including the implementation of the Invasive *Spartina* Project and sea-level rise, low adult survivorship (ranging from 0.49 to 0.52), and predation of adults and eggs/nestlings.

For the most recent comprehensive assessment of the species' range-wide status, please refer to the California Ridgway's rail 5-Year Review, available at https://ecos.fws.gov/docs/five_year_review/doc6592.pdf (Service 2020).

Critical Habitat

Critical habitat has not been proposed or designated.

Recovery Plan Information

The USFWS published the Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California in 2013 (USFWS 2013a). Recovery of the California Ridgway's rails requires a combination of interim and long-term actions. Interim actions are those necessary to maintain current populations, while long-term actions focus on recovering the species throughout its range. Interim actions involve monitoring current populations (number and distribution), non-native predator and invasive plant control, reducing human disturbance and protection of existing habitat. Long-term actions involve large-scale tidal marsh restoration and implementation of long-term management plans.

Environmental Baseline

The California Ridgway's rail only occurs within the State of California. Please refer to the information above.

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Coastal California Gnatcatcher (*Polioptila californica*)

Listing Status

The coastal California gnatcatcher (gnatcatcher) was federally listed as threatened on March 30, 1993. The primary threat was habitat loss associated with development (58 FR 16742). Critical habitat was designated on December 19, 2007 (72 FR 72010).

Life History and Habitat

The range and distribution of the gnatcatcher is closely aligned with coastal scrub vegetation. This vegetation is typified by low (less than 3 feet), shrub and sub-shrub species that are often drought deciduous (Service 2010).

The gnatcatcher is nonmigratory and defends breeding territories ranging in size from 2 to 14 acres. The home range size of the gnatcatcher varies seasonally and geographically, with winter season home ranges being larger than breeding season ranges and inland populations having larger home ranges than coastal. The breeding season of the gnatcatcher generally extends from late February through July (sometimes later), with the peak of nest initiations (start-ups) occurring from mid-March through mid-May (Service 2010).

Juveniles are dependent upon or remain closely associated with their parents for up to several months following departure from the nest and dispersal from their natal (place of birth) territory. Dispersal of juveniles generally requires a corridor of native vegetation that provides certain foraging and sheltering requisites and that connects to larger patches of appropriate sage scrub vegetation (Service 2010).

Population Status

The range of the gnatcatcher is coastal southern California and northwestern Baja California, Mexico, from southern Ventura and San Bernardino counties, California, south to approximately El Rosario, Mexico, at about 30 degrees north latitude, which is approximately the same as at listing (Service 2010). We don’t have reliable estimates for the numbers of coastal California gnatcatchers across its range, but Winchell and Doherty (2008) estimated there were 1,324 (95 percent confidence interval: 976–1,673) gnatcatcher pairs over a 111,006-acre area on public and quasi-public lands of Orange and San Diego counties.

Available evidence indicates modification, curtailment, and destruction of gnatcatcher habitat has been occurring over the recent past and we anticipate these actions to continue over the foreseeable future due to development and wildfire. Regardless of the potential magnitude of the threat, the effects of development resulting from population growth in the region have been tempered in by implementation of regulatory mechanisms, especially the State’s Natural Community Conservation Planning process and the Federal Habitat Conservation Plan process (Service 2010).

A genetic study published by Vandergast et al. (2019) assessed the genetic connectivity within the U.S. portion of the gnatcatcher’s range. The study finds that gnatcatchers are retaining genetic connectivity and a large effective population size throughout most of the U.S. range. This study supports the current method of preserving “core and linkages” through local Habitat Conservation Plans as a strategy for conserving the gnatcatcher in southern California. Conversely, evidence of reduced connectivity and loss of genetic diversity was found within population aggregations within the northern portion of the subspecies’ range (i.e., Ventura and Los Angeles counties) where urbanization has led to increasing habitat fragmentation and a loss of surrounding suitable habitat within 16 miles of those aggregations. This suggests further habitat loss, fragmentation, or degradation within the subspecies’ range could lead to a loss of population connectivity and genetic diversity within the subspecies, as is evident from the emerging population structure within Ventura and Los Angeles counties (Vandergast et al. 2019).

Critical Habitat

The 11 designated critical habitat units for the coastal California gnatcatcher include 197,303 acres of Federal, State, local, and private land in Ventura, Los Angeles, Orange, Riverside, San Bernardino, and San Diego counties (72 FR 72010). Designated critical habitat includes habitat throughout the subspecies' range in a variety of climatic zones and vegetation types to preserve the genetic and behavioral diversity that currently exists within the subspecies. Physical and biological features of designated critical habitat for the coastal California gnatcatcher are those habitat components that are essential for the primary biological needs of foraging, nesting, rearing of young, intra-specific communication, roosting, dispersal, genetic exchange, or sheltering (72 FR 72010). These include:

1) Dynamic and successional sage scrub habitats (i.e., Venturan coastal sage scrub, Diegan coastal sage scrub, Riversidean sage scrub, maritime succulent scrub, Riversidean alluvial fan scrub, southern coastal bluff scrub, and coastal sage-chaparral scrub) that provide space for individual and population growth, normal behavior, breeding, reproduction, nesting, dispersal, and foraging; and
2) Non-sage scrub habitats such as chaparral, grassland, and riparian areas, in proximity to sage scrub habitats that provide space for dispersal, foraging, and nesting.

Environmental Baseline

The coastal California gnatcatcher occurs primarily in California, but also occurs in northwestern Baja. However, we have limited information regarding coastal California gnatcatcher in northwestern Baja. Also, the designated critical habitat occurs entirely within California. Thus, the status description above also serves as the baseline for this consultation.

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Least Bell's Vireo (*Vireo bellii pusillus*)

Listing Status

The least Bell's vireo was federally listed as endangered on May 2, 1986 (51 FR 16474), driven by anthropogenic modification of the subspecies' riparian breeding habitat (e.g., through flood control, water impoundment and diversion, urban development, agricultural conversion, and livestock grazing) and because of reduced vireo nest productivity (i.e., through anthropogenically elevated levels of brood parasitism by brown-headed cowbirds (*Molothrus ater*)). Critical habitat was designated on February 2, 1994 (59 FR 4845).

Life History and Habitat

Least Bell's vireos are obligate riparian breeders, typically inhabiting structurally diverse woodlands along watercourses. They occur in several riparian habitat types, including cottonwood-willow woodlands/forests and mule fat scrub, plus also mesquite woodlands in the deserts; nesting and foraging may sometimes also occur in neighboring upland areas. Two features that appear to be essential: (1) the presence of dense cover within 3-6 feet of the ground, where nests are typically placed, and (2) a dense, stratified canopy for foraging (Service 1998). Although least Bell's vireos typically nest in willow-dominated areas, plant species composition does not appear to be as important a determinant of nesting site selection as habitat structure.

Least Bell's vireos are insectivorous, preying on a wide variety of insects, including bugs, beetles, grasshoppers, moths, and particularly caterpillars (Service 1998). Vireos arrive in southern California breeding areas by mid-March to early April, with males arriving before females and older birds arriving before first-year breeders (Service 1998). Vireos generally remain on the breeding grounds throughout the summer and fall, sometimes until late September, although some post-breeding migration may begin as early as late July (Service 1998). Male vireos establish and defend breeding territories through singing and physically chasing intruders, with territories typically ranging in size from 0.5 to 7.5 acres (Service 1998).

Population Status

With an estimated 2,968 least Bell's vireo territories in the United States as of 2006, the number of least Bell's vireo territories has increased 10-fold since listing in 1986, when only 291 territories were known. Existing territories occur in San Diego, Riverside, Orange, San Bernardino, Los Angeles, Ventura, Santa Barbara, Inyo, and Kern counties, with infrequent nesting in Monterey, San Benito, and Stanislaus counties (Service 2006).

The federal listing of least Bell's vireo has helped to significantly reduce further impacts due to urbanization, and agricultural practices and grazing have otherwise declined. In addition, nonnative plant removals have helped restore habitat. Cowbird brood parasitism continues to be a significant threat to the vireo. Cowbird trapping in vireo breeding areas has proven a successful tool to halt vireo population declines over the short term, but trapping may not be the best method for long-term recovery of the vireo. It remains unclear as to the best way to manage this threat and additional research is needed to resolve this issue (Service 2006).

A relatively recent threat has emerged that has the potential to significantly impact least Bell's vireo nesting throughout its range. A disease complex involving two species of ambrosia beetles – the polyphagous shot hole borer (*Euwallacea* sp. 1) and Kuroshio shot hole borer (*Euwallacea* sp. 5), a mix of associated fungi (Lynch et al. 2016), and other pathogens are causing widespread damage to trees in riparian ecosystems throughout southern California (Eskalen et al. 2013). For example, vireo-occupied habitat in the Tijuana River (Recovery Unit 1) was infested and an estimated 140,000 trees or 35 percent of the trees showed extensive damage from the disease complex (Boland 2016). However, it is not clear whether the effects of shot hole borer infestations will result in long-term impacts to least Bell's vireo habitat. For example, there has been riparian vegetation regrowth in the effected portions of the Tijuana River, and while the regrown trees have not been reinfested by shot hole borers, there is concern that they may in the future (Boland and Uyeda 2020).

Critical Habitat

Designated critical habitat for least Bell's vireo encompasses a total of about 38,000 acres at 10 localities in portions of 6 counties in southern California. The physical and biological features of designated critical habitat include riverine and floodplain habitats (particularly willow-dominated riparian woodland with dense understory vegetation maintained, in part, in a non-climax stage by periodic floods or other agents) and adjacent coastal sage scrub, chaparral, or other upland plant communities (59 FR 4845).

Recovery Plan Information

A draft recovery plan for least Bell's vireo was released on May 6, 1998 (Service 1998); however, this plan has not been finalized. Although the least Bell's vireo has not met the downlisting goals of the draft recovery plan for several hundred or more breeding pairs of least Bell's vireo at all 11 identified sites, the overall population trend since the time of the listing for 10 of the 11 sites has been positive. In addition, despite the ongoing threat of brood parasitism by cowbirds, the least Bell's vireo population has increased by 10-fold since the time of its listing. Cowbird trapping is well established at Camp Pendleton and within the Prado Basin of the Santa Ana River, which support the two largest concentrations of least Bell's vireo. Wholesale loss and degradation of riparian habitats has halted, and riparian habitat restoration efforts are ongoing in many areas.

However, the following concerns persist: 1) further research is needed to address the primary threat of brood parasitism by cowbirds on the long-term recovery of the least Bell's vireo; 2) without intensive habitat management and cowbird control at the main population sites, which is currently linked to section 7 consultations under the Act, or new evidence to suggest that vireo can persist without management intervention, vireo populations are likely to return to the low levels that necessitated its listing should intensive management cease; 3) a Population Viability Analysis determined that there was no imminent threat of extinction to the least Bell's vireo, but that was based on maintaining reproductive rates correlated with extensive cowbird control; and 4) draft recovery goals established for delisting need further assessment based on current knowledge of population trends and species distribution throughout the State. Although least Bell's vireo populations have increased in coastal southern California, in the desert regions in the eastern part of the state, and in northwestern Baja California, Mexico, the subspecies remains almost entirely absent from portions of its historical range in the Central Valley and coastal central California. The Service is currently evaluating the least Bell's vireo's listing status and will be publishing a 5-year status review in the future.

Environmental Baseline

Since the least Bell's vireo and its designated critical habitat occur entirely within California, except when on wintering grounds, the status description above also serves as the baseline for this consultation.

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Light-footed Ridgway's Rail (*Rallus obsoletus levipes*)

Listing Status

The light-footed Ridgway's rail was federally listed as endangered on March 8, 1969 (34 FR 5034), primarily due to habitat loss and modification.

Life History and Habitat

The light-footed Ridgway's rail is a reclusive bird that resides in marsh habitats of coastal southern California and northern Baja California, Mexico. Rails are predominantly crepuscular, resting throughout the middle of the day, with activity peaking during the mornings and evenings. The rail is an omnivorous and opportunistic forager with a broad diet, living hidden among dense vegetation (Service 2020). The birds forage throughout the estuary and surrounding habitats, with considerable foraging occurring among the higher marsh dominated by *Salicornia* species, *Limonium californicum*, and *Triglochin* species (Service 2020). The diet comprises upland and marsh fauna such as tadpoles (*Hyla* species), California killifish (*Fundulus parvipinnis*), California voles (*Microtus californicus*), beetles (Coleoptera), various snails (including *Helix* species, *Cerithidea californica*, and *Melampus olivaceus*), fiddler and hermit crabs (including *Pachygrapsus crassipes*, *Hemigrapsus oregonensis*, and *Uca crenulata*), crayfish, isopods, other decapods, and some plant material (Service 2020).

The light-footed Ridgway's rail generally resides in coastal marshes (estuaries) (Service 2020). Coastal marshes occur at the interface between two hydrologic systems, where inland freshwater meets and mixes with marine saltwater. These estuaries are dynamic habitats that change daily with the tides, seasonally with the weather, and interannually with the climate. Under natural conditions, many west coast estuaries are typically subject to seasonal mouth closure (Service 2020). Anthropogenic changes to the hydrology, such as ditching and tidal restriction, of many southern California estuaries has resulted in an alteration of this pattern (Service 2020).

Population Status

Currently, the U.S. range of light-footed Ridgway's rails in California extends from southern Ventura County in the north to the Mexican border in the south. This represents a contraction in the range from its historical maximum and since the subspecies was listed in 1969. Even in 1985, when the recovery plan was written, light-footed Ridgway's rails were found as far north as Carpinteria Marsh in southern Santa Barbara County (Service 2020). In the most recent decades, rails have been reliably detected in only four marsh habitats across the range, all of which are located in the two southernmost coastal counties (Orange and San Diego). At most of the remaining marshes, rails are found intermittently, with populations "blinking" on and off over time. Though smaller, these marsh habitats serve not only as stopover habitat for dispersal, but also as life-long territories for a smaller number of pairs, improving the species' representation and redundancy. In total, rails are extant or presumed extant in various numbers at 20 surveyed marshes along the California coast. Light-footed Ridgway's rail also occurs in Mexico, but there is limited information regarding their status in this portion of their range.

The locations where the majority of rails are found are areas with unrestricted tidal flows, natural channelization, and freshwater inputs that help support tall cordgrass growth, resulting in abundant nesting and refugia habitat. Areas with these characteristics are decreasing in many places due to tidal inundation, competition from invasive plants, and drought (Service 2020).

Surveys in 1980 estimated 203 pairs across 11 marsh sites. Since, the population has fluctuated between a low of 142 pairs in 1985 to a high of 656 pairs in 2016 (Service 2020). Since 2016, the numbers of light-footed Ridgway's rail pairs have been in decline, dropping from 656 pairs to 308 in 2019 (Service 2020).

Recovery Plan Information

A recovery plan was issued on June 24, 1985 (Service 1985) and revised on October 4, 2019 (Service 2019). The light-footed Ridgway's rail has not met the criteria for downlisting or delisting, indicating that the threats facing the subspecies have not been sufficiently reduced. Current estimates of suitable habitat, number of pairs, and marshes occupied are insufficient to ensure appropriate resiliency of the subspecies. The rail continues to remain absent from parts of its historical range (Santa Barbara and Los Angeles counties) and occupies fewer marshes than is needed to provide sufficient protection from catastrophic events (redundancy) and the adaptive capacity (representation) to ensure viability of the subspecies long term. Lastly, the status and distribution of the subspecies in Baja California, Mexico remains largely unknown. Recovery efforts are needed to increase the species viability (resiliency, redundancy, and representation) until such time that we can demonstrate that the recovery criteria are met (Service 2020).

Environmental Baseline

Since the light-footed Ridgway's rail occurs primarily within California with limited information available regarding its status in Mexico, the status description above also serves as the baseline for this consultation.

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Marbled Murrelet (*Brachyramphus marmoratus*) and its Critical Habitat

Listing Status

The murrelet was listed as threatened in Washington, Oregon, and northern California on September 28, 1992 (57 FR 45328). On May 24, 1996, the Service designated critical habitat for the murrelet in Washington, Oregon, and California (61 FR 26256). On October 5, 2011, the Service published a final rule revising critical habitat for the murrelet (76 FR 61599). On August 4, 2016, the Service determined that critical habitat for the murrelet as designated in 1996 and revised in 2011 met the statutory definition of critical habitat under the Act (81 FR 51348). The current designation includes 3,698,100 ac. of critical habitat in Washington, Oregon, and California. The Service published a recovery plan for the murrelet in September 1997 (Service 1997).

Life History and Habitat

Murrelets are long-lived seabirds that spend most of their life in the marine environment, with breeding adult birds annually nesting in the forest canopy of mature and old-growth forests. Because of their small body size, cryptic plumage, crepuscular activity, fast flight speed, solitary nesting behavior, and secretive behavior near nests, murrelet nests have been extremely difficult to locate (Hamer and Nelson 1995). In California, breeding occurs from about March 24 through September 15, is asynchronous, and spread over a more prolonged season than for most temperate seabirds. Data from murrelet populations throughout North America show that approximately 84 percent of murrelet young fledge from their nests by August 18 (Hamer and Nelson 1995). The latest published fledging date was a record of a fledgling found on September 21 in Oregon (Hamer and Nelson 1995). However, a live murrelet fledgling was found on a road in Prairie Creek Redwoods State Park, Humboldt County, California on September 24, 2017, only a few miles south of the action area (U.S. Fish and Wildlife Service, Arcata Field Office, unpublished data).

Murrelets have a naturally low reproductive rate; they lay just one egg per year and supposedly first breed at age 3. Re-nesting in the event of nest failure appears to be uncommon but does occur (Hébert et al. 2003, Piatt et al. 2007). Incubation is shared by both sexes with incubation shifts lasting 24 hours and exchanges occurring at dawn (Nelson 1997). Chicks fledge 27 to 40 days after hatching (Nelson 1997). Flights by adults are made from ocean feeding areas to inland nest sites at all times of the day, but most often at dusk and dawn (Hamer and Cummins 1991, Nelson and Hamer 1995).

Murrelets are known to be opportunistic feeders, diving after small schooling fish and large pelagic crustaceans (e.g., euphausiids, mysids, amphipods). They will carry a single energy-dense fish to their chick: typically, larger sand lance, immature herring, anchovy, smelt, and occasionally salmon smolts (Burkett et al. 1995, Carter and Sealy 1987, Nelson 1997).

Habitat Use

Throughout most of their breeding range, including the listed range from Washington to California, murrelets use old-growth coniferous forest habitat for nesting, and forage in the nearshore marine environments. Nests are not built, but rather the egg is placed in a small depression or cup made in moss or other debris on the limb (Service 1997). At the northern end of the range, ground-nesting occurs in the Aleutian Islands and parts of southern Alaska. The distance inland that murrelets breed is variable and influenced by a number of factors; however, the Service considers 50 mi. as the maximum inland distance for determining habitat suitability and amount of habitat within the listed range (Service 2009).

In California, radio-marked murrelets confirmed that breeders forage more closely to nesting habitat once nesting is initiated than non-breeders (Hébert and Golightly 2008, Peery et al. 2009). In northern California, mean home range size was 253 square mi. (mi^2) for non-nesters and 93 mi^2 for nesters (Hébert and Golightly 2008). Mean along-shore movement was 43 mi. for nesting females and 49 mi. for nesting males (Hébert and Golightly 2008). Mean offshore movement was within 0.9 mi. regardless of sex or nesting status (Hébert and Golightly 2008).

Population Status

Rangewide Status of the Species

The murrelet is a small seabird that inhabits the coastal forests and nearshore marine environment along the Pacific Coast of North America from southern California to southern Alaska and the Aleutian Islands (Carter and Morrison 1992, Nelson 1997, Ralph et al. 1995). The breeding range of the murrelet extends along the Pacific Coast from Alaska to Monterey Bay in central California. Some wintering birds occur as far south as northern Baja California, Mexico. However, only the Washington, Oregon, and California population segment is federally listed as threatened (57 FR 45328).

Limited information is available on murrelet historical distribution and abundance; however, most summaries give indications that the distribution of murrelet populations was significantly reduced as habitat was removed throughout its range. Populations declined as a result. In some areas, murrelets have been locally extirpated, or only small numbers persist, risking maintenance of the species' distribution. These areas were identified as "areas of concern" (Service 1997). The areas included distribution gaps in central California, northwestern Oregon, and southwestern Washington, where very little suitable habitat remains, and what habitat does remain occurs in small patches.

Population Summary

Murrelet abundance during the early 1990s in Washington, Oregon, and California was estimated at 18,550 to 32,000 birds (Ralph et al. 1995). Based primarily on results from the NWFP's marbled murrelet monitoring program (NWFP EM Program), the 2019 murrelet population for all Conservation Zones (Service 1997) was estimated at 21,200 birds (95 percent CI: 16,400–26,000; Table 1).

Throughout the listed range of the murrelet, habitat affected by actions consulted on through Section 7 of the Act has been documented by the Service since October 2003. Most of the affected habitat is within the Oregon Coast Range and Siskiyou Coast Ranges with most of the acreage coming from patches of older forest with sufficient nest structure (Table 2).

The overall population trend from the combined 2001-2010 population estimates (Conservation Zones 1-5 [see Recovery Plan] combined) indicate a significant, rangewide annual rate of decline of about 3.7 percent (95 percent CI: -4.8 to -2.7 percent; Falxa et al. 2011).

Table 1. Summary of 2001-2019 murrelet density and population size estimates (rounded to nearest 100 birds) for all Conservation Zones combined. Source: McIver et al. 2021.

Year	Density (birds/km^2)	Bootstrap standard error (birds/km^2)	Coefficient of variation of density (%)	No. birds	No. birds lower 95% CL	No. birds upper 95% CL
2001	2.47	0.25	10.1	21,800	17,500	26,100
2002	2.56	0.31	11.9	22,500	17,300	27,800
2003	2.60	0.25	9.6	22,800	18,500	27,100

Year	Density (birds/km^2)	Bootstrap standard error (birds/km^2)	Coefficient of variation of density (%)	No. birds	No. birds lower 95% CL	No. birds upper 95% CL
2004	2.46	0.26	10.5	21,600	17,100	26,000
2005	2.30	0.25	10.7	20,200	16,000	24,400
2006	2.09	0.17	8.2	18,300	15,400	21,300
2007	1.97	0.27	13.7	17,300	12,700	22,000
2008	2.06	0.18	8.9	18,100	15,000	21,300
2009	1.96	0.21	10.6	17,200	13,600	20,800
2010	1.89	0.21	11.1	16,600	13,000	20,200
2011	2.50	0.31	12.6	22,000	16,600	27,400
2012	2.40	0.27	11.3	21,100	16,400	25,800
2013	2.24	0.25	11.1	19,700	15,400	23,900
2014	2.43	0.22	9.1	21,300	17,500	25,100
2015	2.75	0.26	9.5	24,100	19,700	28,600
2016	2.58	0.26	10.0	22,600	18,200	27,100
2017	2.62	0.26	10.1	23,000	18,500	27,600
2018	2.56	0.29	11.4	22,500	17,500	27,600
2019	2.42	0.28	11.5	21,200	16,400	26,000

Table 2. Aggregate results of all suitable habitat (ac.) affected by section 7 consultation for the murrelet: summary of effects by conservation zone and habitat type for 1 October 2003 to 19 August 2021.

Conservation zone[1]	Authorized habitat effects[2]		Reported habitat effects[2]	
	Stands[3]	Remnants[4]	Stands[3]	Remnants[4]
Puget Sound	-105	0	-1	0
Western Washington	-13	0	-12	0
Outsize CZ Area in WA	0	0	0	0
Oregon Coast Range	-5,119	-2,551	-2,717	-1,608

Conservation zone[1]	Authorized habitat effects[2]		Reported habitat effects[2]	
	Stands[3]	Remnants[4]	Stands[3]	Remnants[4]
Siskiyou Coast Range	-15,003	-187	-4,957	-187
Outside CZ Area in OR	-35	-3	0	0
Mendocino	0	0	0	0
Santa Cruz Mountains	0	0	0	0
Outside CZ Area in CA	0	0	0	0
Total	**-20,275**	**-2,741**	**-7,687**	**-1,795**

[1]Conservation Zones (CZ): Six zones were established by the Recovery Plan (Service 1997) to guide terrestrial and marine management planning and monitoring for the murrelet.

[2]Habitat includes all known occupied sites, as well as other suitable habitat, though it is not necessarily occupied. Importantly, there is no single definition of suitable habitat, though the Marbled Murrelet Effectiveness Monitoring Module is in the process. Some useable working definitions include the primary constituent elements as defined in the critical habitat final rule, or the criteria used for Washington State by Raphael et al. (2002).

[3]Stand: A patch of older forest in an area with potential platform trees.

[4]Remnants: A residual/remnant stand is an area with scattered potential platform trees within a younger forest that lacks, overall, the structures for murrelet nesting.

Threats

Several threats to murrelets, present in both the marine and terrestrial environments, have been identified. These threats collectively comprise a suite of environmental stressors that, individually or through interaction, have significantly disrupted or impaired behaviors which are essential to the reproduction or survival of individuals. When combined with the species naturally low reproductive rate, these stressors have led to declines in murrelet abundance, distribution, and reproduction at the population scale.

When the murrelet was listed under the Act and threats were summarized in the recovery plan the following anthropogenic threats were identified as having caused the dramatic decline in the species:

- Habitat destruction and modification in the terrestrial environment from timber harvest and human development caused a severe reduction in the amount of nesting habitat.
- Unnaturally high levels of predation resulting from forest “edge effects,” as well as elevated predator densities in the vicinity of areas of high human use (e.g., campgrounds, picnic areas).
- Inadequate existing regulatory mechanisms, such as land management plans (in 1992), that were considered inadequate to ensure protection of the remaining nesting habitat and reestablishment of future nesting habitat.

- Anthropogenic factors such as mortality from oil spills and entanglement in fishing nets used in gill-net fisheries.

There have been changes in the levels of these threats since the 1992 listing (Service 2004, 2009). The regulatory mechanisms implemented since 1992 that affect land management in Washington, Oregon, and California (for example, the Northwest Forest Plan [NWFP]) and new gill-netting regulations in northern California and Washington have reduced the threats to murrelets (Service 2004). The threat levels for the other threats identified in 1992 listing (57 FR 45333) including the loss of nesting habitat, predation rates, and mortality risks from oil spills and gill net fisheries (despite the regulatory changes) remained unchanged following the Service's 2004 5-year [status] review for the murrelet (Service 2004).

However, new threats were identified in the Service's 2009 5-year review for the murrelet (Service 2009). These new stressors were due to several environmental factors affecting murrelets in the marine environment. These new stressors include:

- Habitat destruction, modification, or curtailment of the marine environmental conditions necessary to support murrelets due to:
    - Elevated levels of polychlorinated biphenyls in murrelet prey species.
    - Reduced prey abundance, availability, and quality.
    - Harmful algal blooms that produce biotoxins leading to domoic acid and paralytic shellfish poisoning that have caused murrelet mortality.
    - Climate change in the Pacific Northwest.
- Anthropogenic factors that affect the continued existence of the species include:
    - Derelict fishing gear leading to mortality from entanglement.
    - Energy development projects (wave, tidal, and terrestrial wind energy projects) leading to mortality.
    - Disturbance in the marine environment (from exposures to lethal and sub-lethal levels of high underwater sound pressures caused by pile-driving, underwater detonations, and potential disturbance from high vessel traffic; particularly a factor in Washington).

Five-Year Status Review

In the 2009 5-year review, the following new threats were identified for the murrelet (Service 2009, pp. 27-67):

- Habitat destruction, modification, or curtailment of the marine environmental conditions necessary to support murrelets due to:
    - Elevated levels of polychlorinated biphenyls in murrelet prey species;
    - Changes in prey abundance and availability;
    - Changes in prey quality;
    - Harmful algal blooms that produce biotoxins leading to domoic acid and paralytic shellfish poisoning that have caused murrelet mortality; and
    - Climate change in the Pacific Northwest.
- Manmade factors that affect the continued existence of the species include:
    - Derelict fishing gear leading to mortality from entanglement;
    - Energy development projects (wave, tidal, and on-shore wind energy projects) leading to mortality; and
    - Disturbance in the marine environment (from exposures to lethal and sub-lethal levels of high underwater sound pressures caused by pile-driving, underwater detonations, and potential disturbance from high vessel traffic; particularly a factor in Washington state).

The 2019 5-year review did not describe new threats from this list, but did reference new information on increasing at risk of mortality in trawling gear, but that the scope and severity of the threat to murrelets of entanglement in derelict fishing gear has not changed (Service 2019, p. 64).

Climate change, combined with effects from past management practices, is exacerbating changes in forest ecosystem processes and dynamics to a greater degree than originally anticipated under the NWFP. Environmental variation affects all wildlife populations; however, climate change presents new challenges as systems may change beyond historical ranges of variability. In some areas, changes in weather and climate may result in major shifts in vegetation communities that can persist in particular regions.

The 2019 5-year review concluded that climate change could exacerbate the impacts of continued nesting habitat loss and fragmentation (Service 2019, p. 64) and will affect the environmental baseline for murrelets and other listed species. Although it appears likely that the murrelet will be adversely affected by long-term consequences of climate change, we are not able to specifically quantify the magnitude of effects to the species (Service 2009, p. 34). The threats present in both the marine and terrestrial environments collectively comprise a suite of environmental stressors that, individually or through interaction, have likely disrupted or impaired behaviors which are essential to the reproduction or survival of individuals. When combined with the species naturally low reproductive rate, these stressors have led to declines in murrelet abundance, distribution, and reproduction at the population scale within the listed range.

On August 5, 2024, the U.S. Fish and Wildlife Service completed a five-year status review of the marbled murrelet and concluded that the species' threatened status would remain unchanged (USFWS 2024).

Critical Habitat

On May 24, 1996, the Service designated critical habitat for the murrelet within 104 critical habitat units encompassing approximately 3.9 million acres across Washington (1.6 million), Oregon (1.5 million), and California (0.7 million). The final rule became effective June 24, 1996. The final rule indicated that the scope of the section 7(a)(2) analysis should evaluate impacts of an action on critical habitat at the conservation zone(s) or even a major part of a conservation zone (Service 1996, p. 26271).

The physical and biological features (PBFs) are features the Service determines are essential to a species' conservation (i.e., recovery) and require special management considerations. For murrelets, the Service determined the PBFs (also referred to as the primary constituent elements (PCEs)) associated with the terrestrial environment that support nesting, roosting, and other normal behaviors are essential to the conservation of the murrelet and require special management considerations. The PBFs for the murrelet are:

- PCE-1: individual trees with potential nesting platforms; and
- PCE-2: forested lands of at least one half site potential tree height regardless of contiguity within 0.8 kilometers (0.5 miles) of individual trees with potential nesting platforms, and that are used or potentially used by murrelets for nesting or roosting (Service 1996, p. 26264). The site-potential tree height is the average maximum height for trees given the local growing conditions, and is based on species-specific site index tables.

These PBFs are intended to support terrestrial habitat for successful reproduction, roosting and other normal behaviors.

Recovery Plan Information

The murrelet recovery plan identified actions necessary to stabilize the population including protecting occupied habitat and minimizing the loss of unoccupied suitable habitat. Specific actions include maintaining large blocks of suitable habitat, maintaining and enhancing buffer habitat, decreasing risks of nesting habitat loss due to fire and windthrow, reducing predation, and minimizing disturbance. Long-term conservation needs identified in the plan include:

- Increasing productivity (abundance, the ratio of juveniles to adults, and nest success) and population size.
- Increasing the amount (stand size and number of stands), quality, and distribution of suitable nesting habitat.
- Protecting and improving the quality of the near-shore marine environment.
- Reducing or eliminating threats to survivorship by reducing predation in the terrestrial environment and anthropogenic sources of mortality at sea.

Conservation Zones

Conservation zones are the functional equivalent of recovery units as defined by Service policy (Service 1997). The murrelet recovery plan (Service 1997) identified six "conservation zones" throughout the listed range of the species: Conservation Zone 1: Puget Sound; Conservation Zone 2: Western Washington Coast Range; Conservation Zone 3: Oregon Coast Range; Conservation Zone 4: Siskiyou Coast Range; Conservation Zone 5: Mendocino; and, Conservation Zone 6: Santa Cruz Mountains.

Environmental Baseline

In California, there are three marbled murrelet conservation zones: Conservation Zone 4-Siskiyou Coast Range; Conservation Zone 5-Mendocino; and Conservation Zone 6-Santa Cruz Mountains.

Conservation Zone 4 extends from North Bend, Oregon to the southern boundary of Humboldt County, California. In general, it extends inland 35 mi. from the Pacific Ocean shoreline and includes waters within 1.2 mi. of the shoreline. Conservation Zone 5 extends south from the southern boundary of Humboldt County to the mouth of San Francisco Bay and also includes marine waters within 1.2 mi. of the Pacific Ocean shoreline but extends inland a distance of up to 25 mi. Conservation Zone 6 extends south from the mouth of San Francisco Bay to Point Sur, Monterey County, California and includes marine waters within 1.2 mi. of the Pacific Ocean shoreline, and extends inland a distance of up to 15 mi. (Service 1997).

Lands considered necessary for the recovery of the murrelet within Conservation Zones 4, 5, and 6 are: (1) any suitable habitat managed by the federal government in late-successional reserves (LSRs) located in the Forest Ecosystem Management Assessment Team Zone 1, (2) other large areas of suitable habitat on federal lands outside of LSRs, (3) large areas of suitable habitat on state lands within 25 mi. of the coast in California and Oregon, (4) suitable habitat on county park lands within 25 mi. of the coast in San Mateo and Santa Cruz counties, California, and (5) suitable nesting habitat on Humboldt Redwood Company (formerly Pacific Lumber Company) lands in Humboldt County, California (Service 1997).

Marine areas in California considered necessary for recovery of the murrelet include: (1) nearshore waters (within 1.2 mi. of the shore) along the Pacific Coast from the Oregon-California border south to Cape Mendocino in northern California, including Humboldt and Arcata bays, and river mouths, and (2) nearshore waters (within 1.2 mi. of shore) along the Pacific Coast in central California from San Pedro Point south to the mouth of the Pajaro River (Service 1997).

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Northern Spotted Owl (*Strix occidentalis caurina*) and its Critical Habitat

Listing Status

The northern spotted owl (NSO) was listed as threatened on June 26, 1990, due to widespread loss and adverse modification of suitable habitat across the species' entire range and the inadequacy of existing regulatory mechanisms to conserve the owl (55 FR 26114). In 2019, the species' 5-year review documented its declining status (Service, 2019). After this review, the Service concluded that uplisting the NSO to 'endangered' was warranted, but precluded, by higher priority actions to amend the List of Endangered and Threatened Wildlife and Plants (85 FR 81144).

Life History and Habitat

Northern spotted owls are primarily nocturnal (Forsman et al. 1984, pp. 51-52) and spend virtually their entire lives beneath the forest canopy (Courtney et al. 2004, p. 2-5). They are adapted to maneuverability beneath the forest canopy rather than strong, sustained flight (Gutiérrez et al. 1995, p. 9). They forage between dusk and dawn and sleep during the day with peak activity occurring during the two hours after sunset and the two hours prior to sunrise.

Northern spotted owls seek sheltered roosts to avoid inclement weather, summer heat, and predation (Forsman 1975, pp. 105-106; Barrows and Barrows 1978; Barrows 1981; Forsman et al. 1984, pp. 29-30). Northern spotted owls become stressed at temperatures above 28°C, but there is no evidence to indicate that they have been directly killed by temperature because of their ability to thermoregulate by seeking out shady roosts in the forest understory on hot days (Barrows and Barrows 1978; Forsman et al. 1984, pp. 29-30, 54; Weathers et al. 2001, pp. 678, 684). During warm weather, northern spotted owls seek roosts in shady recesses of understory trees and occasionally will even roost on the ground (Barrows and Barrows 1978, pp. 3, 7-8; Barrows 1981, pp. 302-306, 308; Forsman et al. 1984, pp. 29-30, 54; Gutiérrez et al. 1995, p. 7).

Northern spotted owls are territorial; however, home ranges of adjacent pairs overlap (Forsman et al. 1984, p. 22; Solis and Gutiérrez 1990, p. 746) suggesting that the area defended is smaller than the area used for foraging. They will actively defend their nests and young from predators (Forsman 1975, p. 15; Gutiérrez et al. 1995, p. 11). Territorial defense is primarily carried out by hooting, barking and whistle type calls. Some northern spotted owls are not territorial but either remain as residents within the territory of a pair or move among territories (Gutiérrez 1996, p. 4). These birds are referred to as "floaters." Floaters have special significance in northern spotted owl populations because they may buffer the territorial population from decline (Franklin 1992, p. 822). Little is known about floaters other than that they exist and typically do not respond to calls as vigorously as territorial birds (Gutiérrez 1996, p. 4).

The northern spotted owl is relatively long-lived, has a long reproductive life span, invests significantly in parental care, and exhibits high adult survivorship relative to other North American owls (Forsman et al. 1984; Gutiérrez et al. 1995, p. 5). Northern spotted owls are sexually mature at 1 year of age, but rarely breed until they are 2 to 5 years of age (Miller et al. 1985, p. 93; Franklin 1992, p. 821; Forsman et al. 2002, p. 17). Courtship behavior usually begins in February or March, and females typically lay eggs in late March or April. The timing of nesting and fledging varies with latitude and elevation (Forsman et al. 1984, p. 32). After they leave the nest in late May or June, juvenile northern spotted owls depend on their parents until they are able to fly and hunt on their own. Parental care continues after fledging into September (Service 1990; Forsman et al. 1984, p. 38). During the first few weeks after the young leave the nest, the adults often roost with them during the day. By late summer, the adults are rarely found roosting with their young and usually only visit the juveniles to feed them at night (Forsman et al. 1984, p. 38). Hybridization of northern spotted owls with California spotted owls and barred owls has been confirmed through genetic research (Hamer et al. 1994, pp. 487-492; Gutiérrez et al. 1995, pp. 2-3; Dark et al. 1998, p. 52; Kelly 2001, pp. 33-35; Funk et al. 2008, pp. 161-171).

Northern spotted owls are monogamous and usually form long-term pair bonds. “Divorces” occur but are relatively uncommon. There are no known examples of polygyny in this owl, although associations of three or more birds have been reported (Gutiérrez et al. 1995, p. 10).

Population Status

Rangewide Status of the Species

There is little information regarding the total number of NSOs existing throughout their range. Existing field surveys are not extensive enough, nor consistent enough to produce reliable estimates of the range-wide NSO population size. Since the mid-1990s, range-wide demographic data from 11 long-term monitoring areas has been used as a surrogate to evaluate trends in NSO populations. Based on the demographic data, the most recent population meta-analysis found:

1) Populations experienced significant annual declines of 6-9 percent on six study areas and annual declines of 2-5 percent on five other study areas, and
2) Annual declines translated to ≤ 35 percent of the NSO populations remaining on seven study areas since 1995.
3) Barred owl presence in NSO territories is the primary factor negatively affecting apparent NSO survival, recruitment, and ultimately, rates of population change.

This analysis indicates NSO populations potentially face extirpation if the negative effects of barred owls are not ameliorated while maintaining NSO habitat across their range (Franklin et al. 2021). Weather and climate were additional factors associated with population decline.

In summary, the rangewide NSO population is in decline as a result of decades of habitat loss and degradation and the recent expansion of barred owl populations throughout its range. Given these documented declines, NSO populations range-wide have a reduced ability to withstand additional impacts.

Because range-wide population estimates are lacking, other methods have been used to understand the rangewide status of NSO. “Minimum known alive” estimates have been reported (Birdlife International 2016) but are out of date and vastly underestimate the true number of NSOs due to limited survey coverage. Without an empirical study on total population size, the best available information we use for the purpose of this PBO is Dunk et al. 2012. These authors used model simulations over time in response to various habitat scenarios to estimate the total number of NSOs. This modeling effort was started for the Recovery Plan and finalized during development of the final critical habitat rule (Service 2012). The modeling scenario for the critical habitat rule (composite 11) was selected for because it: 1) had a pessimistic habitat change scenario, and 2) reflected the final critical habitat network as reserve areas. All composites and simulations were based on estimates of a reasonable middle ground on implementation of barred owl control (midpoint between no barred owl control and complete barred owl eradication). The model simulations, assuming all female NSOs are part of a pair, using composite 11 found there were an estimated 6,662 NSOs (95 percent confidence intervals of 5,954-6,944 individuals).

While the purpose of the modeling was not intended to predict actual population size or trend in the future, it does provide general insights into population size through the lens of NSO habitat carrying capacity and other factors. What is not accounted for here is the loss of habitat from recent large wildfires since 2012 and the effects those natural events have had on the rangewide NSO population. Population modeling based on carrying capacity of suitable habitat to support territorial NSO pairs is currently in progress (Davis et al. unpublished data).

Threats

The NSO has declined across large portions of its range since 1990. The immediate threats include habitat loss from timber harvest or severe wildfire and competition with barred owls (*Strix varia*), which invaded from eastern North America. The most severe declines are occurring in the northern portion of the NSO's range, where barred owls have been established for the longest period of time. The current rate of decline raises concerns about the long-term persistence of the NSO throughout the Pacific Northwest.

Wildfire is currently the primary cause of habitat loss on Federal lands, and the rate and severity of wildfire in portions of the range are expected to increase in the future under projected climate change scenarios. Habitat for NSO on private lands has continued to decline since 1990 and has declined at a higher rate than on Federal lands; thus, Federal and State lands are expected to provide the majority of the NSO habitat for the foreseeable future. With the exception of some areas in northern California, it is unlikely NSOs will persist in areas without Federal lands.

Five-Year Status Review

In 2004 and 2011, the USFWS conducted five-year status reviews of NSO. Refer to the 2011 Recovery Plan for NSO for a complete review of the species status.

Population Summary

In the most recent meta-analysis, 26 years of survey and capture-recapture data from long-term demographic study areas (DSAs) across the range were used to analyze demographic traits, rates of population change, and occupancy parameters for NSO territories. The most recent annual rate of decline (5.3 percent) indicates the NSO's extinction risk has significantly increased since the time of listing (Franklin et al. 2021 p. 13). The populations in the DSAs have declined from 32 to over 80 percent since the early- to mid-1990s.

If this rate continues into the future, the NSO will likely decline to extirpation in the northern portion of its range in the near future where population declines have been greatest – over 60 percent. Additionally, NSO population simulations indicate that without a reduction in barred owls in NSO territories and habitat, the NSO populations in Washington and the Oregon Coast Ranges have a greater than 50 percent probability of extirpation.

Barred owl presence in NSO territories was the primary factor negatively affecting apparent survival, recruitment, and ultimately, rates of NSO population change. The analysis of NSO and barred owl detections in an occupancy framework corroborated the capture-recapture analyses with barred owl presence 1) increasing NSO territorial extinction (where NSOs leave their territories) and 2) decreasing NSO territorial colonization (where NSOs establish new territories). While landscape habitat components of higher value habitats reduced the effect of barred owls on the NSO's rates of decline, they did not reverse the negative trend. The NSO populations potentially face extirpation if the negative effects of barred owls are not ameliorated while maintaining NSO habitat across their range (Franklin et al. 2021).

Critical Habitat

A revised designation of spotted owl critical habitat was published on December 4, 2012 (77 FR 71875) and became effective January 3, 2013. In response to a stipulated settlement agreement, the Service proposed a new revised critical habitat rule in 2020 (85 FR 48487), that included exclusions to the 2012 rule. The final rule (86 FR 4820), published in January 2021, included the withdrawal of almost 3.5 million acres of critical habitat with the only modifications occurring in Oregon. A final revised rule (86 FR 62606) became effective on December 10, 2021. Critical habitat for the northern spotted owl now includes approximately 9,577,969 acres in 11 units and 60 subunits in California, Oregon, and Washington. The table below lists the units and subunits of critical habitat for NSO in California.

Critical Habitat Units for Northern Spotted Owl in California	Critical Habitat Subunits for Northern Spotted Owl in California
Unit 3	RDC 1
	RDC 2
	RDC 5
Unit 8	ECS 2
	ECS 3
Unit 9	KLW 4
	KLW 5
	KLW 6
	KLW 7
	KLW 8
	KLW 9
Unit 10	KLE 6
	KLE 7
Unit 11	ICC 1
	ICC 2
	ICC 3
	ICC 4
	ICC 5
	ICC 7
	ICC 8

The final rule for critical habitat defines the primary constituent elements (PCEs) as the specific elements of the physical and biological features (PBFs) that are considered essential to the conservation of the northern spotted owl and are those elements that make areas suitable as nesting, roosting, foraging, and dispersal habitat (Service 2012, p. 71904). In 2016, the Service returned to the use of statutory reference of PBFs rather than PCEs when evaluating and discussing the availability and function of, as well as the effects to the attributes of critical habitat in the adverse modification analysis (Service and NOAA 2016, p. 2716). References to PCE here are to be consistent with cited critical habitat rule. The PCEs should be arranged spatially such that it is favorable to the persistence of populations, survival and reproductive success of resident pairs, and survival of dispersing individuals until they are able to recruit into a breeding population (Service 2012, p. 71904). Within areas essential for the conservation and recovery of the northern spotted owl, the Service has determined that the PCEs are:

1. Forest types that may be in early-, mid-, or late-seral stages and that support the northern spotted owl across its geographic range;
2. Habitat that provides for nesting and roosting;
3. Habitat that provides for foraging;
4. Habitat to support the transience and colonization phases of dispersal, which in all cases would optimally be composed of nesting, roosting, or foraging habitat (PCEs 2 or 3), but which may also be composed of other forest types that occur between larger blocks of nesting, roosting, or foraging habitat (Service 2012, pp. 72051-72052).

Some critical habitat subunits may contain all of the PBFs and support multiple life history requirements of the northern spotted owl, while some subunits may contain only those PBFs necessary to support the species' particular use of that habitat. All of the areas designated as critical habitat, however, do contain PCE 1, forest type. As described in the final rule, PCE 1 always occurs in concert with at least one other PCE (PCE 2, 3, or 4; Service 2012, p. 72051). Northern spotted owl critical habitat does not include meadows, grasslands, oak woodlands, aspen woodlands, or manmade structures and the land upon which they are located (Service 2012, p. 71918).

Recovery Plan Information

The Revised Recovery Plan was published in June 2011 (Recovery Plan). It identifies competition with barred owls, ongoing loss of habitat from timber harvest, loss or modification of habitat from uncharacteristic wildfire, and loss of amount and distribution of habitat from past activities and disturbances as the primary threats (Service 2011, p. II-2 and Appendix A). To address these threats, the recovery strategy includes: 1) developing a rangewide habitat modeling framework, 2) barred owl management, 3) monitoring and research, 4) adaptive management, and 5) habitat conservation and active forest restoration (Service 2011, p. II-2). The Service also completed a rangewide, multi-step habitat modeling process to help evaluate and inform management decisions and designate critical habitat (Service 2011, Appendix C).

There are 14 recovery actions that specifically address habitat loss and degradation. Two actions of primary importance for Federal land managers are recovery actions 10 and 32:

- Recovery Action 10: "Conserve NSO sites and high value NSO habitat to provide additional demographic support to the population." This recovery action addresses both nesting/roosting and foraging habitat. Interim guidance consists of a framework to help determine and prioritize high value habitat and NSO sites for conservation (Service 2011, pp. III-44 to III-45).
- Recovery Action 32: "Because recovery requires well distributed, older and more structurally complex multi-layered conifer forests on Federal and non-Federal lands across its range, land managers should work with the Service…to maintain and restore such habitat while allowing for other threats, such as fire and insects, to be addressed by restoration management actions. These high-quality NSO habitat stands are characterized as having large diameter trees, high amounts of canopy cover, and decadence components such as broken-topped live trees, mistletoe, cavities, large snags, and fallen trees." This recovery action primarily addresses nesting/roosting habitat, but forest stands or patches meeting the described conditions are a subset of nesting, roosting and foraging habitat (Service 2011, p. III-67).

Because maintaining or restoring forests with high-quality habitat will provide additional support for reducing key threats faced by NSOs, protecting these forests should provide them with high-quality refugia habitat from negative competitive interactions with barred owls that are likely occurring where the two species' home ranges overlap.

The Recovery Plan strongly encourages land managers to be aggressive in the implementation of the recovery actions, including strategies that include active forest management. In other words, land managers should not be so conservative that, to avoid risk, they forego actions necessary to conserve forest ecosystems which are necessary to the long-term conservation of the NSO. But they should also not be so aggressive that they subject NSOs and their habitat to treatments where long-term benefits do not clearly outweigh the short-term risks. Finding the appropriate balance to this dichotomy remains an ongoing challenge for those engaged in NSO conservation (Service 2011, p. II-12).

Both the Recovery Plan and the 2012 (and 2021) critical habitat designations build on the Northwest Forest Plan and recommend continued implementation of the Plan and its standards and guidelines (Service 2011, p. I-1). This includes being consistent with the direction for Late-Successional Reserves.

In addition to recovery actions regarding habitat, there are 10 recovery actions specific to addressing barred owl threats. We have undertaken Recovery Action 30; designing and implementing large-scale control experiments to assess the effects of barred owl removal on NSO site occupancy, reproduction, and survival. We are currently planning Recovery Action 31; manage to reduce the negative effects of barred owls on NSOs, to help meet Recovery Criteria (Service 2011, p. III-65).

Environmental Baseline

In redwood forests and mixed conifer-hardwood forests along the coast of northwestern California, spotted owls occur in both old growth forests and younger forest stands, particularly in areas where hardwoods provide a multi-layered structure at an early age (Thomas et al. 1990, p. 158; Diller and Thome 1999, p. 275). In the southern portion of their range, where woodrats are a major component of their diet, northern spotted owls are more likely to use a variety of stands, including younger stands, brushy openings in older stands, and edges between forest types in response to higher prey density in some of these areas (Forsman et al. 1984, pp. 24-29).

Barred Owls

Recovery objectives in the Recovery Plan for dry forests include maintaining sufficient NSO habitat in the short-term to allow them to persist in the face of threats from barred owl expansion and habitat loss from wildfires. While large wildfires continue to be a leading cause of NSO habitat loss on federal lands, competition from barred owls is considered the primary cause of population decline (Franklin et al. 2021, Dugger et al. 2016, Service 2011). Barred owls have expanded their distribution across the range of the NSO and are now distributed throughout all of the provinces across the range. All National Forests adjacent to the KNF (Shasta-Trinity, Six Rivers, and Rogue River-Siskiyou), and private industrial timberland managers with large-scale survey efforts in the Klamath Province, have confirmed occupancy and nesting by barred owls (USDI FWS 2000-2021 consultation records for various projects).

At this time, barred owls do not appear to be as densely distributed in the California Klamath Province as in the California Coastal Province or physiographic provinces to the north. They are increasingly detected during NSO surveys throughout this province, however. The available data suggests strong demographic effects to NSOs and negative inter-specific interactions between the two species (Franklin et al. 2021, Courtney et al. 2004, Dugger et al. 2016, 2011, Gutiérrez et al. 2007, Hamer et al. 2007, Livezy and Fleming 2007, Monahan and Hijamans 2007, Van Lanen et al. 2011, Wiens et al. 2014, 2010). There is current evidence that barred owls occur in higher densities than NSOs in many parts of the range (Hamer et al. 2007, Singleton et al. 2010, Wiens et al. 2014, 2011). In a recent study, the highest densities were in the Oregon Coast Range, with up to 20 barred owls per NSO territory reported (Wiens et al. 2017).

Barred owls and NSOs share similar habitats and likely compete for food resources (Hamer et al. 2001, Gutiérrez et al. 2007, Livezey and Fleming 2007, Wiens et al. 2014). Barred owl diets are more diverse than NSO diets and include species associated with riparian and other moist habitats (e.g., fish, invertebrates, frogs, and crayfish), along with more terrestrial and diurnal species (Smith et al. 1983, Hamer et al. 2001, Gronau 2005, Wiens et al., 2014). Where the two species overlap, barred owls may be taking primary prey of NSO, reducing availability and density of NSO prey. This can lead to a depletion of prey such that NSO cannot find an adequate amount of food to support reproduction or individual survival (Gutiérrez et al. 2007, Livezey and Fleming 2007). These impacts are likely having additional effects on ecosystem processes and food webs of other species (Holm et al. 2017). In addition to competition for prey, barred owls are competing for habitat (Hamer et al. 1989, Dunbar et al. 1991, Herter and Hicks 2000, Pearson and Livezey 2003, Wiens et al. 2014).

Barred owls were initially thought to be more closely associated with early-successional forests than NSOs, based on studies conducted on the west slope of the Cascades in Washington (Hamer et al. 1989, Iverson 1993). More recent studies show they frequently use mature and old-growth forests (Pearson and Livezey 2003, Gremel 2005, Schmidt 2006, Singleton et al. 2010).

In the fire-prone forests of eastern Washington, a telemetry study conducted on barred owls and NSOs showed barred owl home ranges were located on lower slopes or valley bottoms, in closed canopy, mature, Douglas-fir forest, while NSO sites were located on mid-elevation areas with southern or western exposure, characterized by closed canopy, mature, ponderosa pine or Douglas-fir forest (Singleton et al. 2005). Several other studies in western Washington have also shown that when barred owls are present, NSO habitat use shifts upslope and into areas with steeper slopes and more marginal habitat conditions (Pearson and Livezey 2003, Gremel et al. 2005, Mangan et al. 2019, Irwin et al. 2020). The most recent rangewide meta-analysis indicates barred owl colonization of NSO territories is more likely in lower-elevation territories in most of the DSAs (Franklin et al. 2021).

Dugger and others have described synergistic effects associated with NSO territory composition and presence of barred owls. Some NSO pairs retained their territories and continued to survive and successfully reproduce, even when barred owls were present. The effects of reduced old growth forest in core areas were also compounded when barred owls were present and extinction rates of NSO territories nearly tripled when barred owls were detected under these conditions (Dugger et al. 2011).

Most recently, apparent survival, recruitment, and territory colonization and extinction rates were the key vital rates associated with barred owl presence in NSO populations (Franklin et al. 2021). The authors suggest that without barred owl management, near-term extirpation of NSOs is likely in portions of the range, and the small populations that may remain in other parts of the range will be highly vulnerable to extirpation from wildfire or other stressors, resulting in eventual extinction. Dugger et al. (2016) found the removal of barred owls in the Green Diamond study area in northern California had rapid, positive effects on NSO survival and rates of population change. Removal of barred owls here resulted in increases in NSO occupancy with an estimated survival rate of 0.859 compared with 0.822 in areas where barred owls were not removed (Diller et al. 2016). The study area had an overall lower density of barred owls compared with other portions of the NSOs range, but the results suggest NSOs are likely to recolonize their former territories following barred owl removal.

The meta-analysis of the larger, multi-year barred owl removal experiment (Wiens et al. 2021) in five DSAs across the range also demonstrates the removal of invasive barred owls has a strong, positive effect on survival of native NSOs, and subsequently reduced long-term NSO population declines. Removal of barred owls also influenced the dispersal dynamics of resident NSOs in at least two study areas where NSO from territories that did not have barred owl removal showed an increased estimated probability of movement to territories where barred owls had been removed. The results of the barred owl control experiments across the NSOs range indicate that persistence and recovery of NSO populations are possible with active control, at least over the short term, in managed areas (Wiens et al. 2021).

The research and literature clearly demonstrate the negative influence barred owls are having on NSO site occupancy, fecundity, reproduction, apparent survival, and detectability. The data indicates that over the last 26 years, they are significantly contributing to NSO population declines (Olson et al. 2005, Forsman et al. 2011, Dugger et al. 2011, 2016, Franklin et al. 2021).

As barred owls have expanded, the occupancy of historical and new NSO territories is declining and NSO territory extinction is increasing. Where barred owls and NSOs overlap in spatial distribution, habitat use, and prey use, there is a high potential for interference competition (Wiens et al. 2014, Dugger et al. 2011). Spatial avoidance may be one way for NSOs to reduce these competitive interactions; however, this may put them at greater risk for predation and limit the resources available to them. Habitat loss will likely further constrain the two species to the same set of limited resources, thereby increasing competitive pressure and leading to additional negative impacts to NSO (Wiens et al. 2014). However, NSO recovery will also require short and long-term availability of older forests and suitable habitat on the landscape (Wiens et al. 2021, Franklin et al. 2021).

The current condition for barred owls and NSOs further supports previous recommendations to conserve and preserve high-quality habitat (Forsman et al. 2012, 2011, Dugger et al. 2011, Service 2011, 2012). NSOs can be displaced because of fire or habitat reductions from forest management. They may have increased difficulty in finding new territories to colonize, or in expanding their home ranges to compensate for habitat reductions when barred owls are present on the landscape. In areas where NSO and barred owl compete directly for resources, maintaining larger amounts of older forest (nesting/roosting habitat) may help NSOs persist in the short term (Dugger et al. 2011, 2016).

There are current information gaps regarding 1) the ecological interactions between NSOs and barred owls (Service 2011, p. III-62), and 2) the effects of forest management on their interactions (Courtney et al. 2004, Service 2011). These factors are not fully understood or described, and ongoing and future monitoring may provide further understanding.

While the scientific literature has explored the link between climate change and the invasion by barred owls, changing climate alone is unlikely to have caused the invasion (Livezey 2009). In general, climate change can increase the success of introduced or invasive species in colonizing new areas. Invasive animal species are more likely to be generalists, like the barred owl, than specialists, such as the NSO. Generalists can typically adapt more successfully to a changing climate. Recent forecasts indicate climate change will have long-term and variable impacts on forest habitat at local and regional scales. Locally, this could involve shifts in tree species composition that influence habitat suitability. Frey et al. (2016) concluded that old-growth habitat will provide some buffer from the impacts of regional warming or slow the rate at which some species relying on old-growth habitat must adapt. This finding is based on modeling of the fine-scale spatial distribution, below-canopy air temperatures, in central Oregon's mountainous terrain. Similarly, Lesmeister et al. (2019) concluded that older forest can serve as a buffer to climate change and associated increases in wildfire, as these areas have the highest probability of persisting through fire events even in weather conditions associated with high fire activity.

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California Spotted Owl (*Strix occidentalis occidentalis*), Coastal-Southern California DPS

Listing Status

The California spotted owl is currently under federal review for listing under the Act. On February 23, 2023, the Service proposed to list two distinct population segments of the California spotted owl. The Service proposed to list the Coastal-Southern California distinct population segment as endangered and the Sierra Nevada distinct population segment as threatened with a 4(d) rule (88 FR 11600). A species status assessment was issued in January 2023 (Service 2023), compiling biological information and conditions on both distinct population segments. More information on the California spotted owl can be found in *The California Spotted Owl: Current State of Knowledge* (Gutiérrez et al. 2017). The Coastal-Southern California DPS will be considered by the Service as "proposed endangered" until publication of a final listing rule in the Federal Register.

Life History and Habitat

The Coastal-Southern California DPS of the California spotted owl is found in several disjoint subpopulations along the central California coast in the Coast and Transverse Ranges from Monterey County south into Ventura County, and in southern California in the Transverse and Peninsular mountain ranges from Ventura County to San Diego County in the south (Gutiérrez et al. 1995). However, there is a large gap in the species range around San Luis Obispo County where the species is not known to occur. California spotted owls are absent from the Santa Cruz Mountains (part of the Coast Range) in California, where suitable habitat appears to be present (Gutiérrez et al. 2017). The Coastal-Southern California DPS of the California spotted owl can live up to 8,400 feet in elevation (Verner et al. 1992). The Coastal-Southern California DPS of California spotted owl is assumed to function as a metapopulation, though little movement has been recorded between populations (LaHaye *et al.* 1994, Barrowclough *et al.* 2005, LaHaye and Gutiérrez 2005).

California spotted owls are long-lived (~16–23 years) with high adult survival and low reproductive output (Seamans and Gutiérrez 2007, Gutiérrez 2020). California spotted owls form monogamous pair bonds, although occasionally bonds may break, and new bonds may form due to circumstances such as the death of a mate or low reproductive output with a previous mate (Gutiérrez et al. 1995). Pairs will defend a territory from neighboring pairs and vagrant owls, and they exhibit high territory fidelity (Gutiérrez et al. 1995).

Pairs do not necessarily breed every year, but they can breed in consecutive years. In a study conducted in 1990 to 2005, the number of CSO young fledged annually per territorial female ranges from 0.478–0.988 (Blakesley et al. 2010). California spotted owls have a highly variable rate of reproduction, which may be a bet hedging reproductive strategy in which owls may postpone reproduction until temporarily poor environmental conditions improve (Franklin et al. 2000; Gutiérrez et al. 2017). The CSO also has been found to exhibit a pattern of alternate years of high and low reproduction, which population studies have labeled as the even-odd effect of reproductive output although it is unknown why this pattern occurs (Gutiérrez et al. 2017). However, using survey data from 2013-2017, Hobert et al. (2019b) did not find evidence of the even-odd effect and instead found that the probability of successful reproduction in a given year was higher if reproduction occurred in the previous year. Their study found that the probability of occupancy and successful reproduction by CSOs depends on whether owls successfully reproduced at the site the previous year, is higher at lower elevations, and is likely the result of differences in topographic and vegetation conditions (Hobart et al. 2019b).

In general, CSO live in mature, multistoried forests with complex structure, large trees, multi-layered high canopy cover, coarse woody debris, and species richness (Gutiérrez et al. 2017). National Forests within the range of CSO have established Protected Activity Centers (PAC) of about 300 acres that contain high quality habitat surrounding known CSO nesting sites. This habitat provides structures and characteristics required for nesting, roosting, and foraging. Multi-layered canopy cover and presence of large trees provide young CSOs with protection from predators and from high temperatures. California spotted owls have a low heat tolerance in comparison to other bird species, beginning to show heat stress at 84 to 93 degrees Fahrenheit. The cooler microclimates that multi-layered high canopy cover provides are important for both juveniles and adults during warm summers (Weathers 1981; Barrows 1981; Weathers et al. 2001). Additionally, multistoried forests with multilayer canopy cover also provides protection from predators (Franklin et al. 2000).

In coastal and southern California, CSO are found in riparian/hardwood forests and woodlands, live oak/big cone fir forests, and redwood/California laurel forests (Gutiérrez et al. 2017). CSO in southern California still select for territories containing larger trees (LaHaye et al. 1997) and predominantly closed canopy cover (Smith et al. 2002).

The CSO is a central-place forager and activities are concentrated around nests and roosts with foraging activities reduced the farther away from the nest or roost. Home ranges are large areas used by an individual to meet its life history requirements. Home range sizes are highly variable (1,500 to 5,400 acres), and estimates vary by study, latitude, elevation, diet, habitat structure, and individual (Atuo et al. 2019; Forest Service 2019a). Within a home range, a territory is more vigorously defended by the resident single or pair of owls. A territory is consistently used for nesting, roosting, and foraging and contains essential habitat for survival and reproduction (Gutiérrez et al. 1995; Bingham and Noon 1997; Rosenberg and McKelvey 1999; Swindle et al. 1999; Blakesley et al. 2005; Williams et al. 2011). Within a territory, the Forest Service designates a Protected Activity Center (PAC) to protect the best available 300 acres of nesting/roosting habitat surrounding and including a known activity center in as compact an area as possible (Forest Service 2019a). Activity centers are the area where CSO spend most of their time in which they nest, roost, and forage and average around 299 acres (Verner et al. 1992; Gutiérrez et al. 2017). Within the PAC, the forest stand containing the location of a nest and used for egg development through juvenile post-fledge rearing is considered the nest stand. Nest stands have fine-scale habitat features important for breeding, including high canopy cover (in general, at least 70 percent), abundant large trees (typically more than 24 inches dbh), multiple canopy layers dominated by medium-sized trees (12 to 24 inches dbh) and higher-than-average basal area (185 to 350 square feet per acre) (Verner et al. 1992; Bias and Gutierrez 1992; Moen and Gutierrez 1997; North et al. 2000; Chatfield 2005; Blakesley et al. 2005; Roberts et al. 2011). Like other bird species, some CSOs do not occupy a home range and may move within home ranges of other birds to wait for opportunities to join other breeding pairs that may die or desert their territory. These birds are often called "floaters" and their role in CSO populations is considered to be critical (Verner et al. 1992).

California spotted owls nest and roost in areas that are generally characterized as mature forest with multistoried or complex structure and that have larger/taller trees, higher canopy cover, and larger amounts of coarse woody debris (i.e., fallen dead trees and the remains of large branches on the ground) than other sites available for use within the territory. Nesting habitat contains high canopy cover, typically 70% or higher, and trees with potential nest structures such as cavities, platforms, and deformities. California spotted owl foraging habitat is composed of a diversity of vegetation types and seral stages (Roberts et al. 2017). California spotted owls do not build their own nests but rely on old, large trees or snags with many defects like cracks, decay, open cavities, broken tops, and platforms. Most nest trees are a minimum of 30 inches dbh and average 45 inches dbh (Verner et al. 1992; North et al. 2000; Gutiérrez et al. 2017).

California spotted owl foraging habitat is composed of a diversity of vegetation types and seral stages (Roberts et al. 2017). Foraging habitat may include the habitat characteristics described above for nesting/roosting habitat, but may also include younger forests, areas with medium-sized trees (11 to 24 inches QMD), and small open areas. A mosaic of mature closed-canopy forest intermixed with open-canopy patches may promote higher prey diversity and abundance (Zabel et al. 1995; Ward and Noon 1998; Franklin et al. 2000; Tempel et al. 2014; Atuo et al. 2019; Zulla et al. 2023). A study by Kuntze et al. (2023) found higher woodrat prey abundance in CSO home ranges with heterogenous vegetation types compared to home ranges dominated by mature forest. The study also found that individual owls consumed more woodrats and overall 30 percent more prey biomass in those heterogeneous home ranges compared to homogeneous home ranges, and that the increased consumption could be approximately equivalent to the energetic need required to produce one additional offspring. Habitat selection may vary by elevation as one study documented CSOs at lower elevation using sites with shorter trees, lower canopy cover, and less diversity in forest seral types than CSOs at higher elevations (Kramer et al. 2021b). Differences in habitat selection along elevation gradients may be explained by distribution of prey species as owls consume more woodrats at lower elevations and more flying squirrels at higher elevations (Kramer et al. 2021b).

California spotted owl dispersal habitat is essential to maintaining stable populations by filling territorial vacancies when resident CSOs die or leave their territories, and to providing adequate gene flow across the range of the species. At a minimum, dispersal habitat for CSO contains stands with adequate tree size and canopy closure to provide roosting opportunities, protection from avian predators, and at least minimal foraging opportunities (Forest Service 2019a).

Occupancy, colonization, adult survival, and reproductive success in CSO are all positively associated with the proportion of structurally complex forests at multiple scales (Franklin et al. 2000, Blakesley et al. 2005, Tempel et al. 2014a, Tempel et al. 2016, Zulla et al. 2022). Areas of high canopy cover provided by large (and tall) trees is important, especially near the nest site and within the core use area (Blakesley et al. 2005, North et al. 2017, Jones et al. 2018). Areas with canopy cover greater than 70 percent are considered optimal for CSO nest sites and occupancy sharply declines when canopy cover is less than 40 percent (Seamans 2005; Blakesley et al. 2005; Seamans and Gutiérrez 2007a; Tempel et al. 2014, 2016). Even in southern California where the habitat is naturally more fragmented with less canopy cover available, CSOs still select for core areas with higher canopy cover relative to what is available (Smith et al. 2002).

Low severity or patchy mixed severity wildfire is a historical natural occurrence throughout the range of the CSO, creating variable forest stands to which CSO are adapted. Owls residing in a mixed-ownership landscape preferentially forage in older forests near territory centers, but select for diverse forest cover types (seral stages) at the periphery of territories (Atuo et al. 2019). Gaps and small openings of canopy cover are tolerated within a CSO territory, but there are generally no gaps around the nest stand (North et al. 2017). A heterogenous or variable mix of old forest and open areas leads to higher survival and reproduction than uniform old forest conditions because it supports sufficient prey (Franklin et al. 2000, Hobart et al. 2019a). California spotted owls primarily prey upon a variety of small to medium sized mammals. CSO consume more northern flying squirrels (*Glaucomys sabrinus*) in higher elevation coniferous forests and more woodrats (*Neotoma* spp.) in lower elevation oak woodlands and riparian-deciduous forests (Verner et al. 1992, Hobart et al. 2019b).

Population Status

CSO are currently found throughout their known historical range, although there is evidence of a decrease in abundance in parts of the range, including in both the Sierra Nevada and southern California (Franklin *et al.* 2004; Tempel *et al.* 2014, 2022; Conner *et al.* 2016; Hanna *et al.* 2018).

There has not been a range-wide population study of CSO and occupancy data for CSO is limited, so most of our understanding of population trends comes from a few long-term studies. Although there is no information regarding either historical population sizes or estimates for minimum viable population sizes, CSO populations have declined in the study in national forests in central and southern California, such as the San Bernardino National Forest since the early 1990s (Tempel et al. 2014b, Conner et al. 2013, 2016, LaHaye et al. 2004).

Critical Habitat

No critical habitat has been designated for the Coastal-Southern California DPS of the California spotted owl.

Recovery Plan Information

No recovery plan exists for the California spotted owl. However, our analysis of the past, current, and future influences on the California spotted owl revealed that there are several factors that contribute to the current condition and pose a risk to future viability of the species. These risks include large-scale and high-severity fire, tree mortality, effects of drought and climate change, fuels reductions and forest management practices, human development, competition and hybridization with invasive barred owls, secondary ingestion of rodenticides, and regulatory mechanisms and other management actions (Service 2023, Service 2019, Gutiérrez et al. 2017). Parasites, disease, and recreation are not currently known to have population level impacts on the California spotted owl. In addition, habitat within the range of the Coastal-Southern California DPS of the CSO is naturally fragmented, with little dispersal occurring between subpopulations due to discontinuous mountain ranges (Gutiérrez et al. 2017). This natural fragmentation has been further fragmented by development/habitat loss in the greater southern California area.

Environmental Baseline

California spotted owls have been documented in the literature as occurring as far south as the Sierra Juarez and Sierra San Pedro Martir mountain ranges in Baja California Norte, Mexico. However, we have limited information regarding occurrences outside of California. Thus, the status description above also serves as the environmental baseline for this consultation.

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California Spotted Owl (*Strix occidentalis occidentalis*), Sierra Nevada DPS

Listing Status

The California spotted owl is currently under federal review for listing under the Act. On February 23, 2023, the Service proposed to list two distinct population segments of the California spotted owl. The Service proposed to list the Sierra Nevada distinct population segment as threatened with a 4(d) rule and the Coastal-Southern California distinct population segment as endangered (88 FR 11600). A species status assessment was issued in January 2023 (Service 2023), compiling biological information and conditions on both distinct population segments. More details on the California spotted owl can be found in *The California Spotted Owl: Current State of Knowledge* (Gutiérrez et al. 2017). The Sierra Nevada DPS will be considered by the Service as "proposed threatened" until publication of a final listing rule in the Federal Register.

Life History and Habitat

The Sierra Nevada DPS of the California spotted owl is continuously distributed throughout the forests of the western Sierra Nevada Mountains from Shasta County south to Tehachapi Pass (Gutiérrez et al. 2017). The Sierra Nevada DPS is also known to occur rarely in southern Modoc County, and is sparsely distributed on the eastern side of the Sierra Nevada Mountains into western Nevada (GBBO 2017). The distribution of the NSO transitions to the range of the CSO south of the Pit River to just north of Lassen Peak, and interbreeding between the two subspecies occasionally occurs (Barrowclough et al. 2011, Haig et al. 2004, Hanna et al. 2018). The Sierra Nevada DPS of CSO live from about 1,000 – 7,700 feet in elevation (Verner et al. 1992).

California spotted owls are long-lived (~16–23 years) with high adult survival and low reproductive output (Seamans and Gutiérrez 2007, Gutiérrez 2020). California spotted owls form monogamous pair bonds, although occasionally bonds may break, and new bonds may form due to circumstances such as the death of a mate or low reproductive output with a previous mate (Gutiérrez et al. 1995). Pairs will defend a territory from neighboring pairs and vagrant owls, and they exhibit high territory fidelity (Gutiérrez et al. 1995).

Pairs do not necessarily breed every year, but they can breed in consecutive years. In a study conducted in 1990 to 2005, the number of CSO young fledged annually per territorial female ranges from 0.478–0.988 (Blakesley et al. 2010). California spotted owls have a highly variable rate of reproduction, which may be a bet hedging reproductive strategy in which owls may postpone reproduction until temporarily poor environmental conditions improve (Franklin et al. 2000; Gutiérrez et al. 2017). The CSO also has been found to exhibit a pattern of alternate years of high and low reproduction, which population studies have labeled as the even-odd effect of reproductive output although it is unknown why this pattern occurs (Gutiérrez et al. 2017). However, using survey data from 2013-2017, Hobert et al. (2019b) did not find evidence of the even-odd effect and instead found that the probability of successful reproduction in a given year was higher if reproduction occurred in the previous year. Their study found that the probability of occupancy and successful reproduction by CSOs depends on whether owls successfully reproduced at the site the previous year, is higher at lower elevations, and is likely the result of differences in topographic and vegetation conditions (Hobart et al. 2019b).

In general, CSO live in mature, multistoried forests with complex structure, large trees, multi-layered high canopy cover, coarse woody debris, and species richness (Gutiérrez et al. 2017). National Forests within the range of CSO have established Protected Activity Centers (PAC) of about 300 acres that contain high quality habitat surrounding known CSO nesting sites. This habitat provides structures and characteristics required for nesting, roosting, and foraging. It is believed that multi-layered canopy cover and presence of large trees provide young CSOs with protection from predators and from high temperatures. California spotted owls have a low heat tolerance in comparison to other bird species, beginning to show heat stress at 84 to 93 degrees Fahrenheit. The cooler microclimates that multi-layered high canopy cover provides are important for both juveniles and adults during warm summers (Weathers 1981; Barrows 1981; Weathers et al. 2001). Additionally, multistoried forests with multilayer canopy cover also provides protection from predators (Franklin et al. 2000).

In the Sierras, a majority of CSO occur within mid-elevation ponderosa pine, mixed-conifer, white fir, and mixed-evergreen forest types, with few owls occurring in the lower elevation oak woodlands of the western foothills (Gutiérrez et al. 2017).

The CSO is a central-place forager and activities are concentrated around nests and roosts with foraging activities reduced the farther away from the nest or roost. Home ranges are large areas used by an individual to meet its life history requirements. Home range sizes are highly variable (1,500 to 5,400 acres), and estimates vary by study, latitude, elevation, diet, habitat structure, and individual (Atuo et al. 2019; Forest Service 2019). Within a home range, a territory is more vigorously defended by the resident single or pair of owls. A territory is consistently used for nesting, roosting, and foraging and contains essential habitat for survival and reproduction (Gutiérrez et al. 1995; Bingham and Noon 1997; Rosenberg and McKelvey 1999; Swindle et al. 1999; Blakesley et al. 2005; Williams et al. 2011). Territory size varies along a latitudinal gradient because CSO home ranges are smaller in the southern Sierra Nevada than in the northern and central Sierra Nevada (Gutiérrez et al. 2017). For this document, we consider territory size to be 800 acres in the southern Sierra Nevada National Forests (Inyo, Sierra, and Sequoia) and 1,000 acres in the north and central Sierra Nevada National Forests (Modoc, Lassen, Plumas, Tahoe, Eldorado, Stanislaus and Lake Tahoe Basin Management Unit) to align with research that has been conducted on CSO core use areas (Tempel et al. 2016). Within a territory, the Forest Service designates a Protected Activity Center (PAC) to protect the best available 300 acres of nesting/roosting habitat surrounding and including a known activity center in as compact an area as possible (Forest Service 2019). Activity centers are the area where CSO spend most of their time in which they nest, roost, and forage and average around 299 acres (Verner et al. 1992; Gutiérrez et al. 2017). Within the PAC, the forest stand containing the location of a nest and used for egg development through juvenile post-fledge rearing is considered the nest stand. Nest stands have fine-scale habitat features important for breeding, including high canopy cover (in general, at least 70 percent), abundant large trees (typically more than 24 inches dbh), multiple canopy layers dominated by medium-sized trees (12 to 24 inches dbh) and higher-than-average basal area (185 to 350 square feet per acre) (Verner et al. 1992; Bias and Gutierrez 1992; Moen and Gutierrez 1997; North et al. 2000; Chatfield 2005; Blakesley et al. 2005; Roberts et al. 2011). Like other bird species, some CSOs do not occupy a home range and may move within home ranges of other birds to wait for opportunities to join other breeding pairs that may die or desert their territory. These birds are often called “floaters” and their role in CSO populations is considered to be critical (Verner et al. 1992).

California spotted owls nest and roost in areas that are generally characterized as mature forest with multistoried or complex structure and that have larger/taller trees, higher canopy cover, and larger amounts of coarse woody debris (i.e., fallen dead trees and the remains of large branches on the ground) than other sites available for use within the territory. Nesting habitat contains high canopy cover, typically 70% or higher, and trees with potential nest structures such as cavities, platforms, and deformities. California spotted owl foraging habitat is composed of a diversity of vegetation types and seral stages (Roberts et al. 2017). California spotted owls do not build their own nests but rely on old, large trees or snags with many defects like cracks, decay, open cavities, broken tops, and platforms. Most nest trees are a minimum of 30 inches dbh and average 45 inches dbh (Verner et al. 1992; North et al. 2000; Gutiérrez et al. 2017).

California spotted owl foraging habitat is composed of a diversity of vegetation types and seral stages (Roberts et al. 2017). Foraging habitat may include the habitat characteristics described above for nesting/roosting habitat, but may also include younger forests, areas with medium-sized trees (11 to 24 inches QMD), and small open areas. A mosaic of mature closed-canopy forest intermixed with open-canopy patches may promote higher prey diversity and abundance (Zabel et al. 1995; Ward and Noon 1998; Franklin et al. 2000; Tempel et al. 2014; Atuo et al. 2019; Zulla et al. 2023). A study by Kuntze et al. (2023) found higher woodrat prey abundance in CSO home ranges with heterogenous vegetation types compared to home ranges dominated by mature forest. The study also found that individual owls consumed more woodrats and overall 30 percent more prey biomass in those heterogeneous home ranges compared to homogeneous home ranges, and that the increased consumption could be approximately equivalent to the energetic need required to produce one additional offspring. Habitat selection may vary by elevation as one study documented CSOs at lower elevation using sites with shorter trees, lower canopy cover, and less diversity in forest seral types than CSOs at higher elevations (Kramer et al. 2021b). Differences in habitat selection along elevation gradients may be explained by distribution of prey species as owls consume more woodrats at lower elevations and more flying squirrels at higher elevations (Kramer et al. 2021).

California spotted owl dispersal habitat is essential to maintaining stable populations by filling territorial vacancies when resident CSOs die or leave their territories, and to providing adequate gene flow across the range of the species. At a minimum, dispersal habitat for CSO contains stands with adequate tree size and canopy closure to provide roosting opportunities, protection from avian predators, and at least minimal foraging opportunities (Forest Service 2019).

Occupancy, colonization, adult survival, and reproductive success in CSO are all positively associated with the proportion of structurally complex forests at multiple scales (Franklin et al. 2000, Blakesley et al. 2005, Tempel et al. 2014a, Tempel et al. 2016, Zulla et al. 2022). Areas of high canopy cover provided by large (and tall) trees is important, especially near the nest site and within the core use area (Blakesley et al. 2005, North et al. 2017, Jones et al. 2018). Areas with canopy cover greater than 70 percent are considered optimal for CSO nest sites and occupancy sharply declines when canopy cover is less than 40 percent (Seamans 2005; Blakesley et al. 2005; Seamans and Gutiérrez 2007a; Tempel et al. 2014, 2016).

Low severity or patchy mixed severity wildfire is a historical natural occurrence throughout the range of the CSO, creating variable forest stands to which CSO are adapted. Owls residing in a mixed-ownership landscape preferentially forage in older forests near territory centers, but select for diverse forest cover types (seral stages) at the periphery of territories (Atuo et al. 2019). Gaps and small openings of canopy cover are tolerated within a CSO territory, but there are generally no gaps around the nest stand (North et al. 2017). A heterogenous or variable mix of old forest and open areas leads to higher survival and reproduction than uniform old forest conditions because it supports sufficient prey (Franklin et al. 2000, Hobart et al. 2019a). California spotted owls primarily prey upon a variety of small to medium sized mammals. CSO consume more northern flying squirrels (*Glaucomys sabrinus*) in higher elevation coniferous forests and more woodrats (*Neotoma* spp.) in lower elevation oak woodlands and riparian-deciduous forests (Verner et al. 1992, Hobart et al. 2019b).

Population Status

CSO are currently found throughout their known historical range, although there is evidence of a decrease in abundance in parts of the range, including in both the Sierra Nevada and southern California (Franklin *et al.* 2004; Tempel *et al.* 2014, 2022; Conner *et al.* 2016; Hanna *et al.* 2018). Information on CSO population trends in the Sierra Nevada largely comes from four long-term studies that span the latitudinal range, three of which occur primarily on National Forest System lands (Lassen, Eldorado, and Sierra) and one which occurs on the Sequoia-Kings Canyon National Park. Although there is no information regarding either historical population sizes or estimates for minimum viable population sizes, CSO populations declined in the study areas from the 1990s to 2013 in three national forests in the Sierra Nevada but not in the Sequoia-Kings Canyon National Parks study area (LaHaye et al. 2004; Conner et al. 2013, 2016; Tempel et al. 2014). While these demography studies have been the main source of empirical data about population trends to date, they may not be entirely representative of forest, ecological province, or range-wide trends for the CSO. Passive acoustic surveys throughout the western slope of the Sierra Nevada range began in 2021 and show that CSO, while rare, were well distributed across the study area (Kelly et al. 2023). Population size for the western Sierra Nevada in 2021 was estimated to be between 2,218 and 2,328 territorial individuals, depending on modeled occupancy criteria (Kelly et al. 2023). The SSA found that most CSO populations in the Sierras were in low or moderate current condition (Service 2019).

Critical Habitat

No critical habitat has been designated for the Sierra Nevada DPS of the California spotted owl.

Recovery Plan Information

No recovery plan exists for the California spotted owl. However, our analysis of the past, current, and future influences on the California spotted owl revealed that there are several factors that contribute to the current condition and pose a risk to future viability of the species. These risks include large-scale and high-severity fire, tree mortality, effects of drought and climate change, fuels reductions and forest management practices, human development, competition and hybridization with invasive barred owls, secondary ingestion of rodenticides, and regulatory mechanisms and other management actions (Service 2023, Service 2019, Gutiérrez et al. 2017). Parasites, disease, and recreation are not currently known to have population level impacts on the California spotted owl.

Environmental Baseline

The Sierra Nevada DPS of the California spotted owl only occurs within the State of California. Please refer to the information above for the environmental baseline.

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Western Snowy Plover (*Anarhynchus nivosus nivosus*), Pacific Coast DPS and its Critical Habitat

Listing Status

The Service listed the Pacific coast population of the western snowy plover (*Anarynchus nivosus nivosus*; formerly *Charadrius nivosus nivosus* andC. *alexandrinus nivosus*) as threatened on March 5, 1993 (58 FR 12864), and designated critical habitat in 1999 (64 FR 68508). Critical habitat was redesignated in 2005 (70 FR 56970) and revised in 2012 (77 FR 36727).

Life History and Habitat

Food Habits

Plovers are primarily visual foragers, using the run-stop-peck method of feeding typical of most plover species. They forage on invertebrates in the wet sand and amongst surf-cast kelp and driftwood within the intertidal zone, in dry sand areas above the high tide, on saltpans, on spoil sites, and along the edges of salt marshes, salt ponds, and lagoons. Plovers may also probe for prey in the sand and pick insects from low-growing plants (Service 2007).

Breeding

Plovers nest from early March through late September: The nesting season may be 2 to 4 weeks earlier in southern California than in Oregon and Washington. Fledging of late-season broods may extend into the third week of September throughout the breeding range (Service 2007). Plover nests consist of a shallow scrape or depression, sometimes lined with beach debris (e.g., small pebbles, shell fragments, plant debris, mud chips). As incubation progresses, plovers may add to and increase the nest lining. Driftwood, kelp, and dune plants provide protective cover for chicks to avoid predators.

Plover nesting chronology includes: (1) 3 days to more than a month for scrape construction (in conjunction with courtship and mating), (2) 4 to 5 days for egg laying, (3) incubation for 28.4 days in the early season (before May 8) to 26.9 days in the late season (Warriner et al. 1986), and (4) fledging about 1 month after hatching. Average clutch size is 3 eggs with a range from 2 to 6 eggs (Page et al. 2009). Both sexes incubate the eggs, with the female tending to incubate during the day and the male at night (Warriner et al. 1986). Plover chicks are precocial, leaving the nest with their parents within hours of hatching (Service 2007). Chicks are nonvolant (i.e., incapable of flight) for approximately 1-month post hatching. Broods rarely remain in the nesting area until fledging (Lauten et al. 2010, Warriner et al.

1986). Casler et al. (1993) reported broods would generally remain within a 1-mile radius of their nesting area; however, in some cases would travel as far as 4 mi. (6.4 km). Adult plovers frequently will attempt to lure people and predators from hatching eggs and chicks with alarm calls and distraction displays.

Habitat Use

Coastal habitats used for nesting include sand spits, dune-backed beaches, beaches at creek and river mouths, and saltpans at lagoons and estuaries (Page and Stenzel 1981, Wilson 1980). Plovers nest less commonly on bluff-backed beaches, dredged material disposal sites, salt pond levees, dry salt ponds, and gravel river bars (Page and Stenzel 1981, Powell et al. 2002, Tuttle et al. 1997, Wilson 1980).

In winter, plovers are found on many of the beaches used for nesting, as well as beaches where they do not nest. They also occur around man-made salt ponds and on estuarine sand and mud flats. In California, most wintering plovers concentrate on sand spits and dune-backed beaches. Some also occur on urban and bluff-backed beaches, which they rarely use for nesting (Page et al. 1986, Page and Stenzel 1981). South of San Mateo County, California, wintering plovers also use pocket beaches at the mouths of creeks and rivers on otherwise rocky substrates (Page et al. 1986). Roosting plovers will sit in depressions in the sand made by footprints and vehicle tracks, or in the lee of kelp, driftwood, or low dunes in wide areas of beaches (Page et al. 2009). Sitting behind debris or in depressions provides some shelter from the wind and may make the birds more difficult for predators to detect.

Population Status

Rangewide Status of the Species

The western snowy plover breeds primarily on coastal beaches from southern Washington to southern Baja California, Mexico. Historical records indicate that nesting plovers were once more widely distributed and abundant in coastal Washington, Oregon, and California (Service 2007). In Washington, plovers formerly nested at five coastal locations (Washington Department of Fish and Wildlife 1995) and at over 20 sites on the coast of Oregon (Service 2007). In California, by the late 1970s, nesting plovers were absent from 33 of 53 locations with breeding records prior to 1970 (Page and Stenzel 1981).

Population size estimates are based on breeding window surveys. In 2019, the Service detected 2,223 adult plovers rangewide during breeding season surveys conducted in all six recovery units (Service, unpublished data). Most breeding adults were from California (1,744), followed by Oregon (381) and Washington (98). During winter window surveys in 2019-2020, the Service detected 4,613 plovers rangewide. As with breeding season surveys, most wintering plovers were from California (4,154), followed by Oregon (384) and Washington (75). Winter window surveys, especially in California, detect many plovers that winter on the coast but breed inland.

Threats

Historical records indicate that nesting plovers were once more widely distributed and abundant in coastal Washington, Oregon, and California. The reasons for decline and degree of threats vary by geographic location; however, the primary threat was, and remains, habitat destruction and degradation. Habitat loss and degradation can be primarily attributed to human disturbance, urban development, introduced European beachgrass (*Ammophila arenaria*), and expanding predator populations (Service 2007). Natural factors, such as inclement weather, have also affected the quality and quantity of plover habitat (58 FR 12865). Sea level rise from climate change will likely reduce the amount of available beach nesting habitat. The 2012 revised critical habitat designations were an attempt to adjust critical habitat boundaries to reflect changes in beach morphology due to sea level rise.

Five-Year Status Review

The Service issued a 5-year review in 2006 (Service 2006),2019 (Service 2019), and 2024 (Service 2024). The 2019 5-year review noted that the taxonomic classification had changed from *Charadrius alexandrinus nivosus* to *Charadrius nivosus nivosus,* since the 2006 published 5-year review. This taxonomic and nomenclatural change did not alter the description, distribution, or listing status of the distinct population segment (DPS). The 2019 and 2024 5-year reviews concluded that the Pacific coast population of western snowy plover status would remain as threatened. Threats had not changed significantly since the 2006 5-year review.

Critical Habitat

The current critical habitat designation (77 FR 36727) includes 60 units totaling 24,526 ac. in Washington, Oregon, and California. The primary constituent elements (PCEs) of critical habitat for the plover include sandy beaches, dune systems immediately inland of an active beach face, salt flats, mud flats, seasonally exposed gravel bars, artificial salt ponds and adjoining levees, and dredge spoil sites, with:

PCE-1: Areas that are below heavily vegetated areas or developed areas and above the daily high tides.

PCE-2: Shoreline habitat areas for feeding, with little or no vegetation, that are between the annual low tide or low water flow and annual high tide or high water flow, subject to inundation but not constantly under water, that support small invertebrates, such as crabs, worms, flies, beetles, spiders, sand hoppers, clams, and ostracods, that are essential food sources.

PCE-3: Surf- or water-deposited organic debris, such as seaweed (including kelp and eelgrass) or driftwood located on open substrates that supports and attracts small invertebrates, and provides cover or shelter from predators and weather, and assists in avoidance of detection (crypsis) for nests, chicks, and incubating adults.

PCE-4: Minimal disturbance from the presence of humans, pets, vehicles, or human-attracted predators, which provide relatively undisturbed areas for individual and population growth and normal behavior.

Designated plover critical habitat by state (77 FR 36728).		
State	**No. CH units**	**CH Area (acres)**
Washington	4	6,077
Oregon	9	2,112
California	47	16,337
Total	60	24,526

Recovery Plan Information

The Service issued a recovery plan in 2007 (Service 2007). The primary objectives of the recovery plan (Service 2007) include:

- Increasing population numbers distributed across the range of the Pacific coast population of the plover;
- Conducting intensive ongoing management for the species and its habitat and developing mechanisms to ensure management in perpetuity; and
- Monitoring plover populations and threats to determine success of recovery actions and refine management actions.

The Recovery Plan includes recommendations for western snowy plover management measures for all known breeding and wintering locations. These locations have been divided into six recovery units, as follows: (1) Oregon and Washington; (2) northern California (Del Norte, Humboldt, and Mendocino counties); (3) San Francisco Bay (locations within Napa, Alameda, Santa Clara, and San Mateo counties); (4) Monterey Bay (including coastal areas along Monterey, Santa Cruz, San Mateo, San Francisco, Marin, and Sonoma counties); (5) San Luis Obispo, Santa Barbara, and Ventura counties; and (6) Los Angeles, Orange, and San Diego counties. Designation of these locations and recovery units assists in identifying priority areas for conservation planning across the western snowy plover's breeding and wintering range.

The Pacific coast population of the plover will be considered for delisting when the following criteria have been met (Service 2007):

- An average of 3,000 breeding adults has been maintained for 10 years, distributed among 6 recovery units as follows: Washington and Oregon, 250 breeding adults; Del Norte, Humboldt, and Mendocino counties, California, 150 breeding adults; San Francisco Bay, California, 500 breeding adults; Sonoma to Monterey counties, California, 400 breeding adults; San Luis Obispo to Ventura counties, California, 1,200 breeding adults; and Los Angeles to San Diego counties, California, 500 breeding adults. This criterion also includes implementing monitoring of site-specific threats, incorporation of management activities into management plans to ameliorate or eliminate those threats, completion of research necessary to modify management and monitoring actions, and development of a post-delisting monitoring plan.
- A yearly average productivity of at least one (1.0) fledged chick per male has been maintained in each recovery unit in the last 5 years prior to delisting.
- Mechanisms have been developed and implemented to assure long-term protection and management of breeding, wintering, and migration areas to maintain the subpopulation sizes and average productivity described above. These mechanisms include establishment of recovery unit working groups, development and implementation of participation plans, development and implementation of management plans for federal and state lands, protection and management of private lands, and public outreach and education.

Environmental Baseline

The vast majority of breeding western snowy plovers continue to nest in California (Page et al. 2008, 2016; California Department of Parks and Recreation [CDPR] 2016; Campbell 2017; Robinette 2016), although an increasing number are now nesting in coastal Oregon and Washington (Lauten et al. 2017; Pearson et al. 2017).

Trends: Notable Population Size Decreases in 2007, 2008, 2012, 2016, 2017, and 2018.

<u>*Analysis of Adult Population Trends (2007-2018) by Recovery Unit in California, RU2-RU6*</u>

Del Norte, Humboldt, and Mendocino (CA); RU2 – the circa-1997 baseline estimate was 50 adults. The recovery target is 150 breeding adults, total population size (Service 2007). In the 2007 downturn this RU saw a 42% loss of adults (-19 adults). The number of breeding adult plovers (30; 16 males and 14 females) was the lowest recorded since monitoring began in 2001 (Colwell et al. 2007). The RU experienced repeated decreases in 2007, 2008, 2009, 2012, and 2017. From 2012 to 2018, however, the breeding window survey estimate increased from 21 adults to 52. The shape of the population trajectory since 2012 is linear, positive, and relatively steep (least-squares best fit; AFWO, unpublished records). However, this is unit has been described by some researchers as a "sink" (Pulliam 1988; Mullin et al. 2010; Eberhart-Phillips and Colwell 2014; Hudgens et al. 2014) in which the population can only be sustained through immigration. RU2 has not approached or exceeded the population recovery target in any breeding window survey year. Nearly all plovers breeding in RU2 occur in Humboldt County, although a new location (Salmon Creek, Sonoma County) was discovered in 2018. Observed fecundity exceeded the target of 1.0 annual fledglings per male in 2016, 2017, 2018, and 2019 (Feucht et al. 2018; Feucht, pers. Comm., 2019).

San Francisco Bay (CA); RU3 – the circa-1997 baseline estimate was 264 adults. The recovery target is 500 breeding adults, total population size (Service 2007). This RU was unaffected by the 2007 downturn, but experienced repeated declines in 2006, 2008, 2011, 2012, 2014 and 2015. From 2005 to 2018, however, the breeding window survey increased from 124 adults to 235. The shape of the population trajectory (2005-2017) is linear (least squares best fit) and positive, with gradual slope and very high year-to-year fluctuation (r-squared = 0.29) (AFWO, unpublished records). The population has not attained or exceeded the recovery target in any survey year since 2005. Fecundity is not estimated in the annual intensive breeding season surveys. This RU is subject to high nest depredation rates and intraspecies aggression given its position within a highly-modified urban environment (former salt ponds and berms), competing habitat restoration needs of other listed species, and the large observed fluctuations in available habitat, especially during the first half of the nesting season, on some years (Robinson-Nilson et al. 2011; Pearl et al. 2018).

Sonoma, Marin, San Francisco, San Mateo, Santa Cruz and Monterey (CA); RU4 – the circa-1997 baseline estimate was 300 adults. The recovery target is 400 breeding adults, total population size (Service 2007). In the 2007 downturn event, this RU experienced a loss of 87 adults (24% less than the 2006 population). Since 2007, the breeding window survey estimate has increased from 257 adults (2008) to 361 (2018). The shape of the population trajectory since 2007 is linear, positive, and gradual, with minimal annual fluctuation (least-squares best fit; AFWO, unpublished records). The population has not attained or exceeded the recovery target in any survey year since 2005. In Monterey Bay, fecundity peaked at 2.0 fledglings per male in 2003 and has been unstable and declining since then, falling below 1.0 in each year since 2012 (Page et al. 2016). Since consecutive-year data have been reported (1995-2014), the fecundity estimates in the Point Reyes subpopulation have exceeded 1.0 annual fledglings per male in 12 of the last 20 years: 1996-1999; 2003-2007; and 2011-2013, including 3 of the last 5 years reported (Campbell 2017).

San Luis Obispo, Santa Barbara, and Ventura, including the northern Channel Islands (CA); RU5 – the circa-1997 baseline estimate was 886 adults. The recovery target is 1,200 breeding adults, total population size (Service 2007). In the 2007 downturn event, this RU experienced a loss of 241 adults (26% less than the 2006 population). Since 2007, the breeding window survey estimate population has increased from 676 adults (2007) to 874 (2018). The shape of the population trajectory since 2007 is linear, positive, and gradual, with minimal annual fluctuation (least squares best fit; AFWO, unpublished records). The population has not attained or exceeded the recovery target in any survey year since 2005. Fecundity data are not compiled for the entire RU due to the number of reporting jurisdictions (Federal, State, local, and private); some underfunded jurisdictions do not collect or report the supporting data on an annual basis. However, annual monitoring reports from several of the larger jurisdictions (e.g., Vandenberg Air Force Base [Robinette et al. 2016], Oceana Dunes State Vehicular Recreation Area [CDPR 2017], and Coal Oil Point Reserve [Sandoval and Nielsen 2016]) report fecundity results that exceed the recovery criterion in most years.

Los Angeles, Orange, and San Diego (CA); RU6 – the circa-1997 population baseline was 316 adults. The recovery target is 500 breeding adults, total population size (Service 2007). In the 2007 downturn event, this RU experienced a loss of 115 adults (39% less than the 2006 population). Since 2007, the breeding window survey estimate has increased from 183 adults (2007) to 451 (2018). The shape of the population trajectory since 2007 is linear, positive, and gradual, with minimal annual fluctuation (least-squares best fit) (AFWO, unpublished records). The population has not attained or exceeded the recovery target in any survey year since 2005. Fecundity data are not reported for the entire RU due to lack of supporting data in some jurisdictions to enable the compiled estimates. Annual monitoring reports from two of the larger jurisdictions (e.g., Marine Corps Base Camp Pendleton [Camp Pendleton] and Naval Base Coronado) report fecundity results that exceed the recovery criterion in most years.

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Mammals

Riparian Woodrat (= San Joaquin Valley Woodrat) (*Neotoma fuscipes riparia*)

Listing Status

The riparian woodrat was listed as endangered on February 3, 2000 (65 FR 8881). No critical habitat has been designated for the riparian woodrat.

Life History and Habitat

Riparian woodrats prefer habitat with a large amount of overall structure, with both understory vegetation and overstory cover. Although no studies have been performed to determine the specific habitat needs of the species, at Caswell Memorial State Park, riparian woodrats are most often observed in areas with a valley oak overstory and a wild grape (*Vitus californica*), willow (*Salix* sp.), blackberry (*Rubus discolor* or *Rubus ursinus*), wild rose (*Rosa californica*), or coyote bush (*Baccharis pilularis*) understory (Kelly et al. 2011). In addition, the best quality habitat appears to contain a significant midstory component of vines or small trees, which the riparian woodrat is thought to utilize in order to access the canopy, where they do a substantial amount of their foraging (Kelly et al. 2011). Other important components of riparian woodrat habitat include wooded or shrub-covered upland refugia to facilitate escape from flood events while preventing predation, and downed trees and dead snags that are used in place of stick lodges (Kelly et al. 2011). At Caswell Memorial State Park, riparian woodrats also make houses of sticks and other litter (Williams 1993). Houses typically are placed on the ground against or straddling a log or exposed roots of a standing tree and are often located in dense brush. Nests also are placed in the crotches and cavities of trees and in hollow logs (USFWS 1998, USFWS 2012).

Woodrats are, for the most part, generalist herbivores. They consume a wide variety of nuts and fruits, fungi, foliage, and some forbs (USFWS 1998).

Population Status

<u>*Rangewide Status of the Species*</u>

Known historical distribution included areas along the San Joaquin, Stanislaus, and Tuolumne rivers, and Corral Hollow, in San Joaquin, Stanislaus, and Merced counties, California (NatureServe 2015).

The current species distribution is in the lower San Joaquin Valley, California (Williams and Kilburn 1984); presently known to be extant only at Caswell Memorial State Park (Williams 1993, NatureServe 2015).

<u>*Population Summary*</u>

Williams (1993) estimated the population of the single known occurrence at 437 individuals. There are two known populations in the same general area of California: one within Caswell Memorial State Park and the other approximately five miles away within the San Joaquin River National Wildlife Refuge (Kelly et al. 2009, Kelly et al. 2011). The population, along the Stanislaus River at Caswell Memorial State Park (CMSP), had been known since before the subspecies was listed in 2000 (65 FR 8881, p. 8881). The other, about 8 kilometers (km) (5 miles (mi.)) south at the San Joaquin River National Wildlife Refuge (SJRNWR), was discovered subsequently (USFWS 2012, pp. 3, 6). The SJRNWR population is considered smaller, and possibly vulnerable to extirpation, based on low trapping success and a complete lack of observations of stick lodges (dens that riparian woodrats make out of sticks) in the area (USFWS 2012, pp. 6, 8).

Since that time, six riparian woodrats were caught during a December 2012 trapping survey at CMSP (Kelly et al. 2014, p.13). One of the captured riparian woodrats had also been caught in a previous survey at CMSP 4 years earlier. No additional trapping efforts have been conducted at CMSP since that time (Reith in litt. 2019, p.1). A single riparian woodrat was also captured at SJRNWR in May 2012 incidental to reintroduction and monitoring efforts for riparian brush rabbit (*Sylvilagus bachmani riparius*) (Kelly et al. 2014, pp. 6–8). A 2017 biological assessment of potential impacts from restoration on lands adjacent to the SJRNWR notes riparian woodrats had been captured at the refuge in 2005, 2009, 2011, and 2012, but mentions no subsequent captures (River Partners 2017, p. 19). However, automatic cameras set up on the refuge for a master's thesis study on riparian brush rabbits obtained over 300 pictures of riparian woodrats at 6 locations during the spring and summer of 2017 (Tarcha 2020, pp. 54, 71).

Threats

Threats to this species include:

- At the time of listing, the threats were a large-scale destruction of riparian habitat due to urban, commercial, and agricultural development, combined with flood control and reclamation activities such as river channelization, levee construction, dam construction, water diversion, and groundwater pumping (65 FR 8881). Areas surrounding levees have been entirely cleared of riparian vegetation and the topography has been leveled and planted with row crops, vineyards, and orchards, leaving no avenues for the riparian woodrat to disperse from its current occupied habitat. Levee construction and stream channelization has degraded the quality of the remaining habitat by increasing the size and duration of flood events within the levees (65 FR 8881). However, since the only known riparian woodrat population locations are on protected lands in the CMSP and SJRNWR, current development of occupied habitat does not pose a serious concern (USFWS 2020). In addition, there are ongoing habitat restoration measures (USFWS 2020). However, the impacts to the riparian woodrat populations due to a major flooding in 2017 have not been determined, but are potentially significant (USFWS 2020).
- Predation from coyotes (*Canis latrans*), gray foxes (*Urocyon cinereoargenteus*), long tailed weasels (*Mustela frenata*), raccoons (*Procyon lotor*), feral domestic cats (*Felis domesticus*) and dogs (*Canis lupus familiaris*), owls (Strigidae), and other raptors was known to occur in the 2000 listing rule (Kelly et al. 2009, 65 FR 8881).
- Reproductive success could also be indirectly affected by black rat presence through reduced nourishment caused by competition for food resources, increased energy expenditure in defending stick lodges or other shelter, and reduced access to high quality habitat from competition with black rats (Kelly et al. 2009, USFWS 2012).
- Both populations of riparian woodrat stand at heightened risk of extinction due to random events. Both populations reside in locations prone to flooding. Riparian woodrats, due to their arboreal nature, are somewhat cushioned from experiencing direct mortality from flood events. Instead, flood events can destroy the stick lodges that are constructed by this species, and can impact the understory that is an important component of riparian woodrat habitat (65 FR 8881).
- Wildfire, while less common than flooding, has occurred at the SJRNWR. No additional fuel management activities have been carried out at CMSP (USFWS 2020), so the level of threat from wildfires may have increased further.
- The effects of climate change include changes in types of precipitation (i.e., rain vs. snow), earlier spring run-off flow regimes, increased stream temperatures, and more generally, changes in the components of the stream hydrograph.
- The only known extant population of riparian woodrat is small, with its size limited by the available habitat. It is thus at an increased risk of extinction because of genetic, demographic, and random catastrophic events (e.g., drought, flooding, fire) that threatens small, isolated populations. Because of its breeding behavior, the effective size of woodrat populations is generally much smaller than the actual population size. This increases the risk of inbreeding depression (USFWS 1998).

- The woodrat population at Caswell Memorial State Park is vulnerable to flooding of the Stanislaus River. Because of its well-developed arboreality (ability to climb in trees), the woodrat itself is not as sensitive to flooding as some other brush-dwelling species (e.g., the riparian brush rabbit). However, woodrat houses are essential for survival and these can be severely impacted by flooding, thus affecting population viability (USFWS 1998).

Five-Year Status Review

There have been two five-year status reviews for this species: one on June 20, 2012, and a more recent one on July 8, 2020. The 2020 five-year status review concluded that the riparian woodrat would remain an endangered species, as defined in the Act (USFWS 2020). The evaluation of several threats affecting the species and analysis of the status of the species in the 2012 status review remained an accurate reflection of the species status in 2020.

Critical Habitat

No critical habitat has been designated for the riparian woodrat.

Recovery Plan Information

The riparian woodrat is covered in the Recovery Plan for Upland Species of the San Joaquin Valley, California (USFWS 1998).

Recovery Actions

- A survey and mapping of all riparian areas along the San Joaquin River (USFWS 2012).
- Develop, in collaboration with owners of riparian land and local levee-maintenance districts, an incentive program for preserving riparian vegetation (USFWS 2012).
- Develop a plan for the restoration of riparian habitat, the establishment of riparian corridors, and the reintroduction, if necessary, of riparian woodrats to suitable habitat (USFWS 2012).
- Initiate a genetic study of the CMSP woodrats, and any other riparian woodrat populations that can be sampled, to determine inbreeding levels; and devise a procedure for ensuring that translocations neither reduce genetic diversity in the parent population nor unduly restrict it in the translocated population (USFWS 2012).
- Establish conservation agreements with willing landowners that do not already have conservation easements, as appropriate and necessary, to accomplish habitat restoration, linkage, and reintroduction goals (USFWS 2012).
- Begin efforts to restore and link riparian habitat, and reintroduce woodrats as appropriate (USFWS 2012).

Environmental Baseline

The riparian woodrat only occurs in the lower San Joaquin Valley, California. Please refer to information above for the environmental baseline.

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Riparian Brush Rabbit (*Sylvilagus bachmani riparius*)

Listing Status

The riparian brush rabbit was listed as endangered on February 23, 2000 (65 FR 8881). No critical habitat has been designated for the riparian brush rabbit.

Life History and Habitat

Riparian brush rabbits occupy riparian forest with a dense shrub layer and dense thickets—including wild rose (*Rosa* sp.), willows (*Salix* sp.), and blackberries (*Rubus* sp.)—close to the Stanislaus River. Where mats of low-growing wild roses, wild grape (*Vitis californica*), and blackberries are found in savanna-like settings, brush rabbits live in tunnels through the vines and shrubs. The presence of more surface litter and lack of willows in the understory signifies areas of higher ground that are not flooded regularly or heavily (USFWS 1998). Brush rabbits frequent small clearings, where they bask in the sun and feed on a variety of herbaceous vegetation (65 FR 8881).

Individuals are intolerant of each other when they come too close, but there is no well-defined territoriality. Young are more tolerant of approach by another rabbit than are adults (USFWS 1998). Much of the remaining riparian habitat within the range of the riparian brush rabbit is confined between the Stanislaus River and a levee (NatureServe 2015). The riparian brush rabbit can climb into bushes and trees, though its climbing is awkward and its abilities limited. This trait probably has significant survival value, given that the riparian forests that are its preferred habitat are subject to inundation by periodic flooding (USFWS 1998). Riparian brush rabbits require nearly continuous shrub cover and seldom move more than 1 m (3 ft.) from cover. They will not cross large, open areas, and therefore are unable to disperse beyond the dense brush of the riparian forest at Caswell Memorial State Park. Due to these circumstances, natural dispersal is not possible (USFWS 1998).

Riparian brush rabbits reach sexual maturity the winter following their birth. The species requires riparian forests with a dense understory shrub layer for breeding. Brush rabbits live in tunnels that run through the vines and shrubs of California wild rose (*Rosa californica*) and Pacific blackberry (*Rubus vitifolius*), and require areas of higher ground that are not flooded regularly or heavily (65 FR 8881). The percentage of females active during the breeding season is unknown, but in one study, 9 of 25 female adults examined showed no signs of reproductive activity (65 FR 8881). Breeding of riparian brush rabbits is restricted to approximately January to May, putting this species at a competitive disadvantage to the desert cottontails outside the park, which breed all year. The period of gestation is about 26 to 30 days (average 27 days), the usual litter size is three or four. Females typically produce three to four (up to five) litters during the season and give birth to between two and six young per litter. On average, a female may produce 9 to 16 young each year. Following birth, the young rabbits remain in the nest about 2 weeks before venturing out, and the female will continue to suckle her young 2 to 3 weeks after their birth (65 FR 8881). Although this is a relatively high reproductive rate, it is lower than many other cottontail species, and five out of six rabbits do not survive to the next breeding season (USFWS 1998).

The riparian brush rabbit is an herbivore, feeding on grasses, sedges, clover, forbs, shoots, and leaves in small clearings adjacent to their riparian habitat. Grasses and other herbs are the most important food for brush rabbits, but shrubs such as California wild rose (*Rosa californica*), marsh baccharis (*Baccharis douglasii*), and California blackberry (*Rubus ursinus*) also are eaten. When available, green clover (*Trifolium wormskioldii*) is preferred over all other foods. Food resource distribution is limited due to the need for brush rabbits to remain within 1 meter (m) (3 feet [ft.]) of their riparian habitat to escape to the cover of a dense understory. Competition exists from the more fecund and mobile desert cottontail (*Sylvilagus audubonii*). Riparian brush rabbits are crepuscular (most active during the twilight hours around dawn and dusk). Depending on season, the main activity periods last 2 to 4 hours. The least activity is from about 10:30 a.m. to 4:00 p.m. Growth rates are fast; young rabbits reach adult size in 4 to 5 months (USFWS 1998; 65 FR 8881).

Population Status

Rangewide Status of the Species

The riparian brush rabbit is believed, based on the presence of suitable habitat, to have been historically found in riparian forests along portions of the Stanislaus River and its tributaries on the Valley floor, from at least Stanislaus County to the Delta (USFWS 1998).

By the mid-1980s, the riparian forest within the former range of the riparian brush rabbit had been reduced to a few small and widely scattered fragments, totaling about 2,100 hectares (5,189 acres). Caswell Memorial State Park, on the Stanislaus River in southern San Joaquin County, is the largest remaining fragment of suitable riparian forest and home to the only extant population of riparian brush rabbit (USFWS 1998).

Population Summary

The short-term population trend is relatively stable (NatureServe 2015). The species population trend is unknown; few captures or sightings have occurred since flooding inundated 80 percent of Caswell Memorial State Park in 1997 (NatureServe 2015). The population at Caswell Memorial State Park may have reached its lowest numbers after a flood in 1976, when survivors were removed from trees and shrubs and transported in boats by Park personnel. After flooding in 1986, the population was estimated at between 10 and 20 individuals. In 1993, the population was estimated at 213 to 312 individuals, and considered to be at carrying capacity under prevailing environmental conditions. Population estimates from 1988 to 1997 have varied from 88 to more than 600 individuals. Flooding in 1997 and 1998 reduced numbers severely. In 1997, no riparian brush rabbits were live-trapped, one was sighted, and pellets from two others were seen; in 1998, one rabbit was live-trapped (65 FR 8881).

However, over the course of several years beginning in late 1998, a series of fragmented riparian brush rabbit occurrences was discovered in the delta region of San Joaquin County (Kelly 2018, p. 211). Rabbits from the newly discovered occurrences were captured for a captive propagation program that began reintroducing riparian brush rabbits to restored habitat at the San Joaquin National Wildlife Refuge and neighboring properties in 2002 (Kelly 2018, pp. 211-212). According to the Species Status Assessment analysis, the reintroduced population is the only riparian brush rabbit population that demonstrates resiliency to withstand or bounce back from environmental or demographic stochastic events (USFWS 2020a, p. 74).

The 2020 Status of the Species Assessment described the current distribution of the riparian brush rabbit is limited to southern San Joaquin County and northern Stanislaus County (USFWS 2020a). The subspecies resides in brushy vegetation associated riparian areas along the Old, Stanislaus, Tuolumne, and San Joaquin rivers. The current distribution also includes brushy vegetation along Paradise Cut, Tom Paine Slough, and a small section of the Union Pacific Railroad right-of-way.

Threats

The destruction and fragmentation of the San Joaquin Valley riparian forest by conversion to various urban and agricultural uses, as well as its degradation through a variety of other human activities, has diminished available habitat to about 5.8 percent of its original extent. Riparian brush rabbits are confined to a narrow habitat range with no ability for natural dispersal. With behavioral restrictions on the species' freedom of movement and extensive habitat fragmentation, there is little chance that those individuals who escape drowning or predation will meet mates or reproduce (USFWS 1998).

To escape periodic flooding, riparian brush rabbits take refuge on cleared levees. The cleared levees do not provide the same protection as their typical riparian habitat, and they are more exposed to predators. This contributes directly to population decline and an elevated risk of extinction (USFWS 1998).

Long-term suppression of fire in Caswell Memorial State Park has caused a buildup of high fuel loads in the dense, brushy habitat to which the rabbits are restricted. Riparian brush rabbit habitat is highly susceptible to catastrophic wildfire that would cause high mortality and severe destruction of habitat (USFWS 1998).

Like most rabbits, the riparian brush rabbit is subject to a variety of common diseases. Contagious diseases could be easily transmitted from neighboring populations of desert cottontails. In the small, remnant brush rabbit population, this kind of epidemic could quickly destroy the entire population (USFWS 1998).

Five-Year Status Review

On July 31, 2020, a five-year status review was conducted for the riparian brush rabbit, in which the USFWS concluded that the riparian brush rabbit would remain an endangered species, as defined in the Act (USFWS 2020b). Research efforts since the species was listed have greatly improved the understanding of the species' ecology and status. Conservation efforts since listing have also improved the species' viability by increasing the amount of available habitat and establishing a new, resilient population. However, the conditions of all but the reintroduced population are poor. Therefore, the riparian brush rabbit is in danger of extinction throughout all or a significant portion of its range because of its low viability (i.e., low resiliency, low redundancy, and low representation) and the seriousness of threats (e.g., flooding, climate change, and disease) to its populations (USFWS 2020b).

Critical Habitat

No critical habitat has been designated for the riparian brush rabbit.

Recovery Plan Information

There are currently no recovery criteria for the riparian brush rabbit. Riparian brush rabbit recovery criteria were not included in the Recovery Plan for Upland Species of the San Joaquin Valley, California (USFWS 1998) because the recovery plan was written and finalized before the species was listed under the Act. However, the recovery plan considered the riparian brush rabbit a species of concern, and identified a number of generalized criteria for long-term conservation. Range-wide population monitoring should be provided for in all management plans.

Recovery Actions

Specifically, the plan identifies the following recovery actions:

- Secure and protect specified recovery areas from incompatible uses. Three or more sites, each with no fewer than 300 adults during average years (USFWS 1998).
- Management Plan approved and implemented for recovery areas that include survival of the species as an objective for all protected sites (USFWS 1998).

Population monitoring in specified recovery areas shows populations sizes of 300 or more adults during average years during a precipitation cycle at each of three or more sites (USFWS 1998).

Environmental Baseline

The riparian brush rabbit only occurs along portions of the Stanislaus River and its tributaries on the Valley floor, from at least Stanislaus County to the Delta, in California. Please refer to information above for the environmental baseline.

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Salt Marsh Harvest Mouse (*Reithrodontomys raviventris*)

Listing Status

The salt marsh harvest mouse was federally listed as endangered in 1970 (35 FR 16047, Service 1970). Critical habitat has not been proposed or designated.

There are two subspecies: the northern salt marsh harvest mouse (*Reithrodontomys raviventris halicoetes*) lives in the marshes of the San Pablo and Suisun bays, and the southern salt marsh harvest mouse (*Reithrodontomys raviventris raviventris*) is found in the marshes of Corte Madera, Richmond, and South San Francisco Bay (USFWS 2013).

As described in the Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California (USFWS 2013), the divide between the northern and southern subspecies occurs in San Pablo Bay near China Camp State Park. The southern subspecies, *Reithrodontomys raviventris raviventris*, occurs south of the break in habitat near San Pedro Point and the northern subspecies, *Reithrodontomys raviventris halicoetes* occurs to the north. The *raviventris* subspecies has a disjunct distribution. It is found from south of Point Pinole at the southeastern edge of San Pablo Bay, south around the eastern side of Central and South San Francisco Bay and the western side of the San Francisco Penninsula north to about San Mateo. It is also found in the Larkspur-Corte Madera area on the Marin Penninsula. The *halicoetes* subspecies form is found on the east side of the Bay northward essentially from San Pedro Point, around San Pablo Bay and throughout the Suisun Bay. It too, has a disjunct distribution, in that it is also found on the Contra Costa County coast from the Pittsburg area to the Carquinez Straits.

Life History and Habitat

The basic habitat of the salt marsh harvest mouse has been described as *Sarcocornia* (pickleweed)-dominated vegetation (Dixon 1908; Fisler 1965 cited in USFWS 2010, 2013). Other highly important habitat considerations include high tide/flood refugia of emergent *Grindelia* (gumplant); both at the upper edge of the marsh and within mature marshes, even at the highest high tides), seasonal use of terrestrial grassland, exploitation of suboptimal habitats, and habitat selection in brackish marsh vegetation where *Sarcocornia* is a relatively minor component, as often is the case in Suisun Bay marshes.

The Smith *et al.* (2014) publication suggests that behavioral flexibility of the salt marsh harvest mouse may allow it to adapt to using diked wetlands. The Smith *et al.* (2019) publication also suggests that salt marsh harvest mice make use of diked wetlands and that as climate change and sea level rise are predicted to threaten coastal marshes, a recovery strategy for salt marsh harvest mice could incorporate managed wetlands.

Telemetry studies of the northern salt marsh harvest mouse at Mare Island Marshes found a mean home range size of 0.21 hectare (0.52 acre), and a mean linear distance moved of 11.9 meters (39 feet) in 2 hours (Bias and Morrison 1999). Most movements occurred in June, and fewest movements occurred in November. Mare Island mean home ranges were much larger than those estimated by Geissel et al. (1988) for the southern subspecies, which were no greater than 0.15 hectare (0.37 acre). Due to different measuring techniques, no comparison between the subspecies regarding mean linear distance traveled can be made. Bias and Morrison (1993 cited in USFWS 2010, 2013; 1999) found that movements through open habitats were not restricted to rare or extraordinary events, however, Shellhammer (in litt. 2009 cited in USFWS 2010) identified that generally mice do not cross large areas of open habitats, assuming that "open habitats" mean "open space" or unvegetated habitat.

Male salt marsh harvest mice are generally sexually active from April through September, while the female breeding season extends from March through November for the northern subspecies, and May through November for the southern subspecies (Fisler 1965 cited in USFWS 2010, 2013). Bias and

Morrison (1993 cited in USFWS 2010, 2013) suggest that the breeding season of the Mare Island population (northern subspecies) extends from August through November; more than 30 percent of the females trapped were pregnant during September and October.

Additional information about the salt marsh harvest mouse biology and ecology is available in the Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California, available at: https://ecos.fws.gov/docs/recovery_plan/TMRP/20130923_TMRP_Books_Signed_FINAL.pdf (USFWS 2013a).

Population Status

There is currently no USFWS range-wide salt marsh harvest mouse monitoring program or protocol nor habitat suitability metrics available to evaluate recovery progress of the species and its habitat. For the 2021 5-year review, the USFWS reviewed new information about the spatial distribution and abundance of mice based on various reported mouse survey results from 2010 through 2019.

The 2021 5-year review noted that while capture efficiency values in fluctuate annually for almost every surveyed site, some possible trends appear. Excluding sites with two or fewer years of data, there appear to be positive population trends from 2010 to 2019 for several sites, including: Eden Landing in the Central/Southern San Francisco Bay Recovery Unit; Napa Sonoma Marsh in the San Pablo Bay Recovery Unit; and Grizzly Island East, Ponds 1-5, and Goodyear Slough in the Suisun Bay Area Recovery Unit. There also appear to be negative population trends at several sites, including: Sonoma Creek 1/Strip Marsh West (formerly Sonoma Baylands)/Tubbs Island Setback/Lower Tubbs Island in San Pablo Bay Recovery Unit; and Hill Slough Wildlife Area/Ponds 1 and 2 (and Ponds 4/4a and Areas 8 and 9), Bradmoor Island/California Water Association, Denverton, Lower Joice Island/Joice Island Unit, and East Border of Grizzly Island plus Crescent Unit in the Suisun Bay Area Recovery Unit. It is noted, however, that for several of the Suisun Bay Area Recovery Unit, sites listed as having apparent negative population trends from 2010 to 2019, the lower value in 2019 followed what appears to have otherwise constituted a positive trend through 2018.

Habitat loss that threatens the salt marsh harvest mouse is due to filling, diking, subsidence, changes in water salinity, non-native species invasions, sea-level rise associated with global climate change and pollution. In addition, habitat suitability of many marshes is further limited by small size, fragmentation, and lack of other vital features such as sufficient escape habitat.

Several marsh restoration projects in the north and south San Francisco Bay and in Suisun Marsh that may increase habitat for the salt marsh harvest mouse are in various stages of implementation (USFWS 2021).

For the most recent comprehensive assessment of the species' range-wide status, please refer to the salt marsh harvest mouse 5-Year Review, available at https://ecos.fws.gov/docs/tess/species_nonpublish/3643.pdf (USFWS 2021).

Critical Habitat

Critical habitat has not been proposed or designated for this species.

Recovery Plan Information

The USFWS published the Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California in 2013 (USFWS 2013a). The basic strategy for recovery of the salt marsh harvest mouse is the protection, enhancement, and restoration of extensive, well-distributed habitat suitable for the species. There are short- and long-term components of the general recovery strategy, as well as specific geographic elements. Both interim and long-term components are necessary; neither alone is sufficient to recover the salt marsh harvest mouse. We have identified 5 recovery units: Suisun Bay Area, San Pablo Bay, Central/South San Francisco Bay, Central Coast, and Morro Bay. Recovery criteria comprise a combination of numerical demographic targets and measures that must be taken to directly ameliorate or eliminate threats to the species in the appropriate subset of the above recovery units.

Environmental Baseline

The salt marsh harvest mouse only occurs within the State of California. Please refer to the information above.

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San Bernardino Kangaroo Rat (*Dipodomys merriami parvus*) and its Critical Habitat

Listing Status

The San Bernardino kangaroo rat was federally listed as endangered on September 24, 1998, primarily due to habitat loss associated with agricultural, urban, and industrial development and small population size (63 FR 51005). Critical habitat was designated on October 17, 2008 (73 FR 61936).

Life History and Habitat

In the final listing rule, we considered that the current range likely encompassed 9,797 acres of habitat with the appropriate soils and vegetative cover to be occupied to some degree by the subspecies as follows: 3,861 acres in the Santa Ana River; 5,161 acres in Lytle and Cajon Creeks; and 775 acres in the San Jacinto River (Service 2009). In the revised critical habitat for the San Bernardino kangaroo rat, we determined that the current range of the species encompasses at least 10,696 acres. While these acres do not encompass all habitat occupied by or suitable for the San Bernardino kangaroo rat, we believe that they do represent much of the remaining occupied habitat (Service 2009).

As identified in the final listing rule, habitat for the San Bernardino kangaroo rat has been severely reduced and fragmented by development, aggregate mining, and related activities in the San Bernardino and San Jacinto valleys (Service 2009). As a result of listing, the Service is working cooperatively with other Federal agencies and local aggregate mining operators to conserve and manage habitat for the San Bernardino kangaroo rat. Thus, the direct threats posed to San Bernardino kangaroo rat from aggregate mining are being addressed. Development within floodplain habitat will continue to increase as a result of population growth within western San Bernardino County and the demand for a larger water supply in southern California. An overall reduction in the amount of habitat available to the San Bernardino kangaroo rat and greater habitat fragmentation will continue to occur. Because of the high level of habitat loss (habitat already reduced by 96% by the time the San Bernardino kangaroo rat was emergency listed), the Service's conservation and recovery strategy is to conserve as much remaining habitat as possible. Management and coordination with Federal, State, and local government agencies and mining operations will be needed to protect San Bernardino kangaroo rat from habitat fragmentation and loss due to urban development, off-highway vehicle use, trash dumping, aggregate mining, and an increase in predators such as domestic and feral cats associated with urban development (Service 2009).

Critical Habitat

Four units of designated critical habitat occur over 32,295 acres in Riverside and San Bernardino counties including the Santa Ana River, Lytle and Cajon Creek, San Jacinto River-Bautista Creek, and the Etiwanda Alluvial Fan and Wash units (73 FR 61936). The physical and biological features of designated critical habitat include:

1. Soil series consisting predominantly of sand, loamy sand, sandy loam, or loam.
2. Alluvial sage scrub and associated vegetation, such as coastal sage scrub and chamise chaparral, with a moderately open canopy.
3. River, creek, stream, and wash channels; alluvial fans; floodplains; floodplain benches and terraces; and historic braided channels that are subject to dynamic geomorphological and hydrological processes typical of fluvial systems within the historic range of the kangaroo rat; these areas may include a mosaic of suitable and unsuitable soils and vegetation that either (a) occur at a scale smaller than the home range of the animal, or (b) form a series of core areas and linkages between them.
4. Upland areas proximal to floodplains with suitable habitat (e.g., floodplains that support the soils, vegetation, or geomorphological, hydrological and wind-driven processes essential to this species).

Environmental Baseline

Since the San Bernardino kangaroo rat and its designated critical habitat occur entirely within California, the status description above also serves as the baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 2009. San Bernardino kangaroo rat (*Dipodomys merriami parvus*) 5-year Status Review: Summary and evaluation. Carlsbad Fish and Wildlife Office, Department of the Interior. 31 pp.

Invertebrates

California Freshwater Shrimp (*Syncaris pacifica*)

Listing Status

The California freshwater shrimp was listed as endangered on October 31, 1998 (53 FR 43884). No critical habitat has been designated for the California freshwater shrimp.

Life History and Habitat

The California freshwater shrimp is found in low-elevation (less than 116 m [380 ft.]), low-gradient (generally less than 1 percent) perennial freshwater streams or intermittent streams with perennial pools, where banks are structurally diverse with undercut banks, exposed roots, overhanging woody debris, or overhanging vegetation (USFWS 1998). Excellent habitat conditions for the shrimp include streams 30 to 90 cm (12 to 36 in.) in depth, with exposed live roots along completely submerged undercut banks (horizontal depth greater than 15 cm [6 in.]), with overhanging stream vegetation and vines (USFWS 2007). California freshwater shrimp are most likely found in areas with bottom substrates dominated by sand (USFWS 1998). They require high water quality, low pollution, and good oxygen levels, and have a low tolerance for other conditions; but no data are available for defining the optimum temperature and stream-flow regime for the shrimp, or the minimum and maximum limits it can tolerate (USFWS 2007).

The California freshwater shrimp has R-selective spawning. Adults reach sexual maturity by their second summer of growth, and breeding begins in fall. To breed, the male transfers and fixes the sperm sac to the female shrimp immediately after her last molt, and the female lays 50 to 120 eggs (USFWS 2007). Females then carry the eggs with them for 8 months throughout the winter to allow for slow, overwintering development. Eggs hatch in June (NatureServe 2015). During the incubation period in which the mother carries the eggs with, her many larvae die due to either adult female death or genetic/embryonic developmental problems. As a result, the number of embryos emerging from the eggs during May and June are reduced typically by 50 percent (53 FR 43884). California freshwater shrimp live up to 3 years (USFWS 1998).

California freshwater shrimp eat mostly small decaying particles found widely distributed throughout their habitat, but will also eat algae. California freshwater shrimp may use visual, tactile, or chemical cues in foraging activities. To eat, they brush up the food with tufts at the ends of their claws and lift it to their mouths (USFWS 1998). Activities, including foraging activities, are reduced in the winter. Growth is also reduced in the winter (USFWS 1998).

Population Status

Rangewide Status of the Species

Prior to human disturbances, the California freshwater shrimp is assumed to have been common in low elevation, perennial freshwater streams in Marin, Sonoma, and Napa counties in California (NatureServe 2015; USFWS 1998).

The California freshwater shrimp is currently restricted to 23 stream segments in a few coastal streams in Marin, Sonoma, and Napa counties in California. The distribution can be separated into four general geographic regions: tributary streams in the lower Russian River drainage, which flows westward into the Pacific Ocean; coastal streams flowing westward directly into the Pacific Ocean; streams draining into Tomales Bay; and streams flowing southward into northern San Pablo Bay (NatureServe 2015; USFWS 2007).

Population Summary

It is known that the range and (most likely) population of the California freshwater shrimp has grown since the shrimp was first listed. When first listed, the California freshwater shrimp was found in 13 locations; it is now known from 23 locations. Population data for the California freshwater shrimp are limited, because few long-term studies of populations have been recorded. The number of individual California freshwater shrimp collected at six sites in Lagunitas creek increased from approximately 1,878 in 1991 to approximately 4,407 in 2000 (USFWS 2011).

Threats

Threats to this species include:

- Reduced precipitation and increased temperatures could have two compounding effects on the California freshwater shrimp. First, reduced rainfall and increased temperatures would result in lower stream flows through reduced runoff and increased evaporation, thereby increasing the likelihood that stream segments dry out during the summer months; this could result in local extirpations and further isolate populations of the shrimp. Drought could also devastate populations of the California freshwater shrimp because the loss of habitat makes it difficult for this species to repopulate affected areas. A second, compounding factor would be an increase in water demand for household and agricultural purposes, which could further reduce stream flows and increase the likelihood that stream segments harboring the species dry out (USFWS 2011).
- Various introduced fish and minnows, such as green sunfish (*Lepomis cyanellus*), bluegill (*Lepomis macrochirus*), smallmouth bass (*Micropterus dolomieui*), largemouth bass (*Micropterus salmoides*), mosquitofish (*Gambusia affinis*), prey on the California freshwater shrimp, thereby limiting the species' distribution. Additionally, several native fish species may also prey on the shrimp (USFWS 2011).
- Urban development creates impervious surfaces that increase the amount of runoff from non-point-source pollutants, as well as increased sedimentation (USFWS 2011).
- Grazing activities may destroy California freshwater shrimp habitat through the removal of riparian vegetation, adverse bank and channel changes, decreased water quality due to runoff from manure lots, increased sediment loads, change in runoff characteristics, and increased water temperatures due to a reduced riparian canopy (USFWS 2011).
- The construction of dams adversely affects California freshwater shrimp in several ways, including: (1) crushing individuals due to construction; (2) inundating habitat; (3) serving as a barrier to movement; (4) altering flow patterns; and (5) increasing sedimentation and siltation downstream when dams are washed out during high winter flows. Impoundments raise the elevation of the inundation zone, drowning the roots of riparian vegetation not adapted to periods of prolonged inundation, and likely reduce riparian vegetation in the area. Lack of riparian vegetation harms shrimp by reducing habitat complexity, increasing the potential for bank scour, reducing detritus production, and eliminating high flow refugia. During drought years, natural reductions in flow combined with water exports could result in losses to shrimp populations (USFWS 2011).

Five-Year Status Review

There have been two five-year status reviews for this species: one on January 10, 2008 and a more recent one on September 8, 2011. The latest five-year status review concluded that the California freshwater shrimp continues to meet the definition of endangered (USFWS 2011).

Critical Habitat

No critical habitat has been designated for the California freshwater shrimp.

Recovery Plan Information

On July 31, 1998, a Recovery Plan was issued for the California freshwater shrimp (USFWS 1998).

Reclassification Criteria

Downlisting from endangered to threatened will be considered when:

- A watershed plan has been prepared and implemented for Lagunitas Creek (including Olema Creek), Walker Creek (including Keys Creek), Stemple Creek, Salmon Creek, Austin Creek (including East Austin Creek), Green Valley Creek (including Atascadero, Jonive, and Redwood creeks), Laguna de Santa Rosa (including Santa Rosa and Blucher creeks), Sonoma Creek (including Yulupa Creek), Napa River (including Gamett Creek), and Huichica Creek.
- Long-term protection is assured for at least one shrimp stream in each of the four drainage units.
- The abundance of California freshwater shrimp approaches carrying capacity in each of 17 streams.

Delisting Criteria

Delisting of the California freshwater shrimp will be considered when:

- A watershed plan has been prepared and implemented for Lagunitas Creek (including Olema Creek), Walker Creek (including Keys Creek), Stemple Creek, Salmon Creek, Austin Creek (including East Austin Creek), Green Valley Creek (including Atascadero, Jonive, and Redwood creeks), Laguna de Santa Rosa (including Santa Rosa and Blucher creeks), Sonoma Creek (including Yulupa Creek), Napa River (including Gamett Creek), and Huichica Creek.
- Long-term protection is assured for at least eight shrimp streams, with at least one in each of the four drainage units.
- Shrimp-bearing streams having fewer than 8 kilometers (km) (5 miles) of potential shrimp habitat have shrimp distributed in all potential habitat; those with more than 8 km (5 mi.) of potential shrimp habitat have shrimp distributed over 8 km (5 mi.) or more.
- Populations of shrimp maintain stable populations approaching carrying capacity for at least 10 years in each of 17 streams.

Recovery Actions

- Remove existing threats to known populations of shrimp (USFWS 1998).
- Restore habitat conditions favorable to shrimp and other native aquatic species at extant localities (USFWS 1998).
- Protect and manage shrimp populations and habitat once the threats have been removed and restoration has been completed (USFWS 1998).
- Monitor and evaluate shrimp habitat conditions and populations (USFWS 1998).
- Assess effectiveness of various conservation efforts on shrimp (USFWS 1998).
- Conduct research on the biology of the species (USFWS 1998).
- Restore and maintain viable shrimp populations at extirpated localities (USFWS 1998).
- Increase public awareness and involvement in the protection of shrimp and native, cohabiting species through various outreach programs (USFWS 1998).
- Assess effects of various conservation efforts on cohabiting, native species (USFWS 1998).
- Assemble a California freshwater shrimp recovery team (USFWS 1998).

In addition, the 2011 five-year status review identified the following recovery recommendations:

- The recovery plan divided shrimp populations into four drainage units in an effort to preserve potential genetic variability (Service 1998); however, the only genetic analysis to date indicates potential variability within drainage units. Therefore, further genetic analysis should be conducted to determine if significant differences exist within and/or between drainage units. Depending on the results on any future genetic analysis, recovery criteria may need to be updated.
- Conduct a habitat assessment of Santa Rosa Creek to determine if there is sufficient habitat to support a reintroduced population.
- A monitoring and survey program should be developed to determine the current distribution of the species, assess habitat conditions, and population trends rangewide.
- Identify areas where restoration actions could improve habitat quality and quantity.

Environmental Baseline

The California freshwater shrimp only occurs in Marin, Sonoma, and Napa counties, California. Please refer to information above for the environmental baseline.

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Conservancy Fairy Shrimp (*Branchinecta conservatio*) and its Critical Habitat

Listing Status

The Conservancy fairy shrimp was listed as endangered on September 19, 1994 (59 FR 48136). Critical habitat was designated for the Conservancy fairy shrimp on February 10, 2006 (71 FR 7118).

Life History and Habitat

The species is typically associated with large, clay-bottomed vernal pool playas with turbid water (Vollmar 2002); however, three pools in Butte Co. and two pools in Solano Co. at the Montezuma wetlands are atypical because they are relatively small in area and have very low turbidity (Vollmar 2002). This species occupies clay-bottomed vernal pools and vernal lakes, Tuscan and Merhten geological formations, and on Basin Rim landforms. The environmental specificity is very narrow; it is ecologically dependent on the presence or absence and duration of water during specific times of the year, as well as water chemistry (NatureServe 2015). They have been observed in vernal pools ranging in size from 30 to 356,253 square meters (323 to 3,834,675 square feet) (Helm 1998). Conservancy fairy shrimp have been found at elevations ranging from 5 to 1,700 meters (16 to 5,577 feet) (Eriksen and Belk 1999). The species has been found at sites that are low in alkalinity (16 to 47 parts per million) and total dissolved solids (20 to 60 parts per million), with pH near 7 (Eriksen and Belk 1999) (USFWS 2005).

The eggs are dropped from the brooding female to the benthos. The eggs hatch when the vernal pools and swales fill with rainwater and the immature stages rapidly develop into adults. Conservancy fairy shrimp hatch out of tiny cysts within the soil during the first winter rains, and complete their entire lifecycle by early summer. Other life history characteristics include mean days to mature (36.5), mean days to reproduce (46.2), and mean population longevity in days (113.9) (Helm 1998, NatureServe 2015). Conservancy fairy shrimp hatch out of tiny cysts within the soil during the first winter rains, and complete their entire lifecycle by early summer (USFWS 2012).

This species is a detritivore and an invertivore (NatureServe 2015).

Population Status

Rangewide Status of the Species

Conservancy fairy shrimp are endemic to vernal pools in California (USFWS 2012). Its current range is restricted to the California Great Central Valley with one outlying population in Ventura County in the Interior Coast Ranges (Erikson and Belk 1999, NatureServe 2015).

Population Summary

This species has experienced a long-term population trend of a decline < 30% to an increase of 25%. The short-term population trend is stable. It is known in areas spanning a north-south distance of 300 km, but disjunct within this range (NatureServe 2015). This species is only known to occur in ten disjunct populations (USFWS 2012).

Conservancy fairy shrimp are rare, and at the time of listing, six widely separated populations (i.e., clusters of localities) of this species were known (59 FR 48136). The status of one of these six populations is unknown. This particular population was described as being located “south of Chico, Tehama County”. Tehama County is actually north of Chico, and this population was not discussed in either the *Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon* (Recovery Plan) (USFWS 2005), or in the last 5-year review for this species (USFWS 2007). Therefore, this population will not be addressed further in this document. Extensive surveys for fairy shrimp throughout the range of Conservancy fairy shrimp have located five additional populations since the species was listed in 1994.

Currently, the Service is aware of 10 populations of Conservancy fairy shrimp, which include (from north to south): (1) Vina Plains, Butte and Tehama counties; (2) Sacramento National Wildlife Refuge (NWR), Glenn County; (3) Mariner Ranch, Placer County; (4) Yolo Bypass Wildlife Area, Yolo County; (5) Jepson Prairie, Solano County; (6) Mapes Ranch, Stanislaus County; (7) University of California (U.C.) Merced area, Merced County; (8) the Highway 165 area, Merced County; (9) Sandy Mush Road, Merced County; and (10) Los Padres National Forest, Ventura County (USFWS 2012).

As described in the last 5-year review (USFWS 2007), Conservancy fairy shrimp were reported at Beale Air Force Base (Beale) in Yuba County in 1991. The specimens collected at Beale were later identified as vernal pool fairy shrimp (C. Rogers, EcoAnalysts, Inc., pers. comm. 2007). Extensive surveys for vernal pool crustaceans have been conducted at Beale since 1991, and no additional Conservancy fairy shrimp have been detected (Kirsten Christopherson, Beale, pers. comm. 2012.). For these reasons, Conservancy fairy shrimp are not believed to occur at Beale or in Yuba County at this time (USFWS 2012).

Threats

Threats to this species include:

- The primary threats are elimination and degradation of vernal pool habitat in the Central Valley area by urban development, water supply and flood control activities, and conversion of wildlands to agricultural use.
- Climate change is expected to have an effect on vernal pool hydrology through changes in the amount and timing of precipitation inputs to vernal pools and the rate of loss through evaporation and evapotranspiration; and these changes in hydrology will likely affect fairy shrimp species because they are obligate aquatic organisms with life histories dependent on certain hydrologic conditions.
- Non-native herbaceous species occur commonly in vernal pool complexes and have become a threat to native vernal pool species through their capacity to change pool hydrology. It is likely that the lack of fires, coupled with the lack of adequate grazing, has increased the densities of non-native herbaceous vegetation surrounding vernal pools, degrading the habitat (NatureServe 2015).
- It is likely that vernal pools containing Conservancy fairy shrimp have been exposed to harmful pesticides to some degree, but the current effects of contaminants on this species are not known at this time (NatureServe 2015).
- The combination of highly specialized pool type and soil characteristics makes the Conservancy fairy shrimp exceedingly rare (Vollmar 2002). This species is only known to occur in ten disjunct populations, with some populations being comprised of a single vernal pool. Such populations may be highly susceptible to extirpation due to chance events or additional environmental disturbance, such as adverse effects from changes in hydrology or temperatures due to climate change, invasive plant species, and inappropriate grazing regimes. If an extirpation event occurs in an isolated population, the opportunities for recolonization will be greatly reduced due to physical isolation from other source populations (USFWS 2012).
- Inappropriate grazing practices include complete elimination of grazing in areas where nonnative grasses dominate the uplands surrounding vernal pools, and inappropriate timing or intensity of grazing (USFWS 2012).

Five-Year Status Review

There have been three five-year status reviews for this species: one on September 24, 2007, one on June 29, 2012, and one on May 9, 2024. The latest five-year status review conducted the Conservancy fairy shrimp continues to meet the definition of endangered and would remain an endangered species (USFWS 2024).

Critical Habitat

Critical habitat was designated for the Conservancy fairy shrimp on February 10, 2006 (71 FR 7118). Critical habitat units are designated for Butte, Colusa, Mariposa, Merced, Solano, Stanislaus, Tehama, and Ventura counties, California. Critical habitat is designated totaling 161,786 acres. Note that Units 2 and 4 have zero acres of designated critical habitat.

- Unit 1 Tehama County, California.
  - Unit 1A: Tehama County, California. From USGS 1:24,000 topographic quadrangles Richardson Springs, and Acorn Hollow.
  - Unit 1B: Tehama County, California. From USGS 1:24,000 topographic quadrangle Richardson Springs NW.
  - Unit 1C: Tehama County, California. From USGS 1:24,000 topographic quadrangle Richardson Springs NW.
  - Unit 1D: Tehama County, and Butte County, California. From USGS 1:24,000 topographic quadrangles Richardson Springs NW, Campbell Mound, Richardson Springs.
  - Unit 1E: Butte County, California. From USGS 1:24,000 topographic quadrangles Richardson Springs.
- Unit 3: Solano County, California. From USGS 1:24,000 topographic quadrangles Elmira, and Denverton.
- Unit 5: Stanislaus County, California. From USGS 1:24,000 topographic quadrangle Ripon.
- Unit 6: Merced County, and Mariposa County, California. From USGS 1:24,000 topographic quadrangles Snelling, Merced Falls, Winton, Yosemite Lake, Haystack Mtn. Indian Gulch, Merced, Planada, Owens Reservoir, Illinois Hill, Plainsburg, Le Grand, and Raynor Creek.
- Unit 7: Merced County, California.
  - Unit 7A: Merced County, California. From USGS 1:24,000 topographic quadrangles Gustine, Stevinson, San Luis Ranch.
  - Unit 7B: Merced County, California. From USGS 1:24,000 topographic quadrangles Stevinson, San Luis Ranch.
  - Unit 7C: Merced County, California. From USGS 1:24,000 topographic quadrangles Stevinson, Arena, San Luis Ranch, Turner Ranch.
  - Unit 7D: Merced County, California. From USGS 1:24,000 scale quadrangles Arena, Turner Ranch.
  - Unit 7E: Merced County, California. From USGS 1:24,000 scale quadrangles Turner Ranch, Sandy Mush.
  - Unit 7F: Merced County, California. From USGS 1:24,000 scale quadrangles Turner Ranch, Sandy Mush.
- Unit 8: Ventura County, California. From USGS 1:24,000 scale quadrangles San Guillermo, Lockwood Valley, Alamo Mountain, Lion Canyon, Topatopa Mountains.

The primary constituent elements of critical habitat for Conservancy fairy shrimp are the habitat components that provide:

(i) Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described below in paragraph (ii), providing for dispersal and promoting hydroperiods of adequate length in the pools;

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 19 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands;

(iii) Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding; and

(iv) Structure within the pools described above in paragraph (ii), consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the Conservancy fairy shrimp (USFWS 2005).

Reclassification and Delisting Criteria

In the 2012 five-year status review, the downlisting/delisting criteria identified for the Conservancy fairy shrimp include:

1. Habitat Protection: Accomplish habitat protection that promotes vernal pool ecosystem function sufficient to contribute to population viability of the covered species.

    1A. Suitable vernal pool habitat within each prioritized core area for the species is protected.

    1B. Species localities distributed across the species geographic range and genetic range are protected. Protection of extreme edges of populations protects the genetic differences that occur there.

    1C. Reintroduction and introductions must be carried out and meet success criteria.

    1D. Additional localities are permanently protected, if determined essential to recovery goals.

    1E. Habitat protection results in protection of hydrology essential to vernal pool ecosystem function, and monitoring indicates that hydrology that contributes to population viability has been maintained through at least one multi-year period that includes above average, average, and below average local rainfall as defined above, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

2. Adaptive Habitat Management and Monitoring.

    2A. Habitat management and monitoring plans that facilitate maintenance of vernal pool ecosystem function and population viability have been developed and implemented for all habitat protected, as previously discussed in Sections 1 (A-E).

    2B. Mechanisms are in place to provide for management in perpetuity and long-term monitoring of habitat protected in Sections 1 (A-E), as previously discussed (funding, personnel, etc.).

    2C. Monitoring indicates that ecosystem function has been maintained in the areas protected under Sections 1 (A-D) for at least one multi-year period that includes above average,

average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

3. Status Surveys.

   3A. Status surveys, 5-year status reviews, and population monitoring show populations within each vernal pool region where the species occur are viable (e.g., evidence of reproduction and recruitment) and have been maintained (stable or increasing) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

   3B. Status surveys, status reviews, and habitat monitoring show that threats identified during and since the listing process have been ameliorated or eliminated. Site-specific threats identified through standardized site assessments and habitat management planning also must be ameliorated or eliminated.

4. Research.

   4A. Research actions necessary for recovery and conservation of the covered species have been identified (these are research actions that have not been specifically identified in the recovery actions but for which a process to develop them has been identified). Research actions (both specifically identified in the recovery actions and determined through the process) on species biology and ecology, habitat management and restoration, and methods to eliminate or ameliorate threats have been completed and incorporated into habitat protection, habitat management and monitoring, and species monitoring plans, and refinement of recovery criteria and actions.

   4B. Research on genetic structure has been completed (for species where necessary – for reintroduction and introduction, seed banking) and results incorporated into habitat protection plans to ensure that within and among population genetic variation is fully representative by populations protected in the Habitat Protection section of this document, described previously in Sections 1 (A-E).

   4C. Research necessary to determine appropriate parameters to measure population viability for each species have been completed.

5. Participation and Outreach.

   5A. Recovery Implementation Team is established and functioning to oversee rangewide recovery efforts.

   5B. Vernal Pool Regional working groups are established and functioning to oversee regional recovery efforts.

   5C. Participation plans for each vernal pool region have been completed and implemented.

   5D. Vernal Pool Regional working groups have developed and implemented outreach and incentive programs that develop partnerships.

Recovery Actions

- Conduct research and use results to refine recovery actions and criteria, and guide overall recovery and long-term conservation efforts (USFWS 2005).
- Develop and implement participation programs (USFWS 2005).

- Protect vernal pool habitat in the largest blocks possible from loss, fragmentation, degradation, and incompatible uses (USFWS 2005).
- Manage, restore, and monitor vernal pool habitat to promote the recovery of listed species and the long-term conservation of the species of concern (USFWS 2005).
- Conduct range-wide status surveys and status reviews for all species addressed in this recovery plan to determine species status and progress toward achieving recovery of listed species and long-term conservation of species of concern (USFWS 2005).

Environmental Baseline

The Conservancy fairy shrimp and its designated critical habitat only occur in the Great Central Valley with one outlying population in Ventura County in the Interior Coast Ranges, in California. Please refer to information above for the environmental baseline.

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Longhorn Fairy Shrimp (*Branchinecta longiantenna*) and its Critical Habitat

Listing Status

The longhorn fairy shrimp was listed as endangered on September 19, 1994 (59 FR 48136). Critical habitat was designated for the longhorn fairy shrimp on February 10, 2006 (71 FR 7118).

Life History and Habitat

The longhorn fairy shrimp is highly adapted to the unpredictable conditions of vernal pool ecosystems. Although the longhorn fairy shrimp is only known from a few localities, these sites contain very different types of vernal pool habitats. Longhorn fairy shrimp in the Livermore Vernal Pool Region in Contra Costa and Alameda counties live in small, clear, sandstone outcrop vernal pools. These sandstone pools are sometimes no larger than 1 m (3.3 ft.) in diameter, have a pH near neutral, and very low alkalinity and conductivity. Water temperatures in these vernal pools have been measured between 10 to 17.8 °C (50 to 64 °F). In the San Joaquin, Fresno County and Carrizo Vernal Pool regions, the longhorn fairy shrimp is found in clear to turbid grassland pools. These grassland pools may be as large as 62 m (203.4 ft.) in diameter. Water temperatures in the grassland vernal pools are also warmer, between 10 to 28 °C (50 to 82 °F). There is some evidence that temperatures may not be warm enough for the species to mature in the northern portions of the Central Valley. The species was most recently observed in a disturbed roadside ditch near Los Baños. Longhorn fairy shrimp have been found at elevations ranging from 23 m (75.5 ft.) in the San Joaquin Vernal Pool Region to 880.5 m (2,887 ft.) in the Carrizo Vernal Pool Region (USFWS 2007; USFWS 2012). Although longhorn fairy shrimp are adapted to variable vernal pool habitats, longhorn fairy shrimp presumably have evolved to persist under a range of variation in climatic conditions such as rainfall and drought. For population maintenance, vernal pools must last longer, on average, than the time needed for a species to reach maturity and produce viable eggs, and relatively small changes in the timing or amount of precipitation can affect population dynamics. Based on existing data, weather conditions in which vernal pool flooding promotes hatching—but in which pools dry (or become too warm) before embryos are fully developed—are expected to have the greatest negative effect on the resistance and resilience of vernal pool fairy shrimp populations as cyst banks are depleted (USFWS 2007; USFWS 2012).

Female fairy shrimp carry their eggs in a ventral brood sac. The eggs either are dropped to the pool bottom or remain in the brood sac until the mother dies and sinks. When the pool dries out, so do the eggs. Resting fairy shrimp eggs are known as cysts. The cysts remain in the dry pool bed until hatching begins in response to rains and other environmental stimuli such as vernal pool filling up (NatureServe 2015). The cyst bank in the soil may contain cysts from several years of breeding. Cysts can withstand extreme environmental conditions because of their protective coatings. Unless they are smashed or punctured, cysts are not digested when moved down the intestines of animals. When fairy shrimp cyst dry up, they are even more tolerant of extreme conditions and can be subjected to temperatures of up to 65 degrees Celsius (°C) (150 degrees Fahrenheit [°F]), or can be frozen for months. Cysts can also withstand near-vacuum conditions for 10 years without damage to the embryo. The cysts do not hatch until they receive proper environmental signals such as rain (Eriksen and Belk 1999). Hatching can begin in the same week that a pool starts to fill (typically in winter). Larvae of longhorn fairy shrimp hatch soon after rains fill the pools and water reaches around 10 °C (50 °F) (Eriksen and Belk 1999) The minimum time to maturity for longhorn fairy shrimp is 23 days, with an average of 43 days (USFWS 2005). Longhorn fairy shrimp have been collected from December to late April and complete their entire lifecycle by early summer (USFWS 2007). Because only one cohort of eggs is produced each year, longhorn fairy shrimp disappear before their native pools dry. Males die first and appear to be less tolerant of stressful conditions than females (Eriksen and Belk 1999).

Longhorn fairy shrimp are opportunistic filter feeders, and need algae, bacteria, protozoa, rotifers, and bits of detritus present in their environments for feeding (NatureServe 2015). They can face competition from other fairy shrimp species present in their environments, although competition is limited (Eriksen and Belk 1999). Active adult longhorn fairy shrimp have been observed from the same vernal pool as versatile fairy shrimp (*Branchinecta lindahli*) and spadefoot toad tadpoles (Mesobatrachia) on the Carrizo Plain (USFWS 2007).

Longhorn fairy shrimp are nonmigratory and have relatively little ability to disperse on their own. Aquatic birds are the most likely agents of dispersal of longhorn fairy shrimp. Large mammals are also known to act as distributors by wallowing in dirt, getting caught in their fur, and transporting the cysts to another wallow. Also, because cysts can pass through the digestive systems, they can be ingested and then deposited in new habitats when the animal urinates. Less commonly, usual flooding and wind can also transport cysts. Certain fairy shrimp species are restricted in distribution, and adjacent soils may have different or no fairy shrimp. Pools observed after years seem to have the same species and structural and genetic diversity (Eriksen and Belk 1999).

Population Status

Rangewide Status of the Species

The extent of the historical range or variation in vernal pool habitats in which the species occurs is not known (USFWS 2012). The distribution of the longhorn fairy shrimp may never have extended into the northern portion of the Central Valley or into southern California. Extensive surveying of vernal pool habitats in southern California has never revealed populations of longhorn fairy shrimp. However, it is likely that the longhorn fairy shrimp was once more widespread in the regions where it is currently known to occur, and in adjacent areas such as the San Joaquin and Southern Sierra Foothill Vernal Pool Regions, where habitat loss has been extensive (USFWS 2007; USFWS 2012). Longhorn fairy shrimp are restricted to the Central Valley (USFWS 2012).

Longhorn fairy shrimp are extremely rare. The longhorn fairy shrimp is known from only a small number of widely separated populations (USFWS 2005). The five known populations of longhorn fairy shrimp are described in the section below titled Population Summary.

Population Summary

Population dynamics for longhorn fairy shrimp have not been investigated, and USFWS does not know of any studies that have assessed the status of cyst banks in isolated or connected pools. Monitoring has not been sufficient to quantify abundance and identify trends, but rather just presence of the species in surveyed pools. Because of the small population size of longhorn fairy shrimp, they are very susceptible to stochastic events (USFWS 2012). The current population trend is stable, but the population trend has historically varied, from a decline of 30 percent to an increase of 25 percent (NatureServe 2015). Currently, there are five known populations of longhorn fairy shrimp: (1) areas in and adjacent to the Carrizo Plain National Monument, San Luis Obispo County; (2) areas in the San Luis National Wildlife Refuge (NWR) Complex, Merced County; (3) areas in the Brushy Peak Preserve, Alameda County; (4) areas in the Vasco Caves Preserve, near the town of Byron in Contra Costa County; and (5) areas in the proposed Alkali Sink Conservation Bank east of Mendota in Fresno County (USFWS 2012). This species was also detected in 2003 in a roadside ditch 2 miles north of Los Baños, in Merced County. Only one individual was detected in the ditch; this occurrence is considered to be an anomaly and not a sustainable population (USFWS 2012).

Threats

Threats to this species include:

- Urban development and conversion of native habitats to agriculture were noted as major threats for the longhorn fairy shrimp when it was listed as endangered in 1994. At the time of listing, the majority of known populations of this species were protected on public lands. Since the time of listing, additional localities have been detected that are in the same populations as those previously known, but not all of them are on protected land. A new population was detected in Fresno County in an area that is currently being proposed as a conservation bank for vernal pool species. The number of unprotected localities has increased considerably since the previous 5-year review. At this time, there are 20 unprotected localities of longhorn fairy shrimp within portions of the Carrizo Plain population (USFWS 2012). These localities occur on privately owned parcels that are about 20 acres in size.
- Stochastic extinction occurs as a result of random or unpredictable disturbances, and is a continued threat to the longhorn fairy shrimp, due to the rarity of the species. Localities or entire populations may be highly susceptible to extirpation due to stochastic events, such as a series of prolonged catastrophic droughts; or additional environmental disturbances, such as adverse effects from adjacent development or agriculture activities, altered hydrology due to climate change, invasive plant species, or inappropriate grazing regimes. If a catastrophic extirpation event occurs in any locality, the opportunities for re-colonization from other source localities within that population may be reduced, with long-term impacts to the abundance and sustainability of longhorn fairy shrimp in that population. More importantly, populations with a limited number of localities could be extirpated entirely. The U.S. Fish and Wildlife Service (USFWS) considers the loss of long-term viability in any one of the five extant populations a serious threat the species' recovery (USFWS 2012).
- Non-native herbaceous species occur commonly in vernal pool complexes and have become a threat to native vernal pool species through their capacity to change pool hydrology. It is likely that the lack of fires, coupled with the lack of adequate grazing, has increased the densities of non-native herbaceous vegetation surrounding vernal pools, degrading the habitat (NatureServe 2015).
- Longhorn fairy shrimp are dependent on vernal pools that have sufficient water to remain wet throughout the annual reproductive phase of the species. Climate change is expected to change hydrologic conditions in some parts of California. In addition, climate change is expected to influence the amount and timing of precipitation inputs to vernal pools and the rate of loss through evaporation and evapotranspiration, which may result in negative effects to vernal pool crustacean species through altered vernal pool hydrology.

Five-Year Status Review

There have been two five-year status reviews for this species: one on September 28, 2007 and one on June 20, 2012. The latest five-year status review conducted the longhorn fairy shrimp continues to meet the definition of endangered and would remain an endangered species (USFWS 2012).

Critical Habitat

Critical habitat was designated for the longhorn fairy shrimp on February 10, 2006 (71 FR 7118). Critical habitat units are designated for Alameda, Contra Costa, Merced, and San Luis Obispo counties, California. Critical habitat is designated totaling 13,557 acres in three units, as follows:

- Unit 1: Contra Costa County. Unit 1A: Contra Costa County. Unit 1B: Alameda County.
- Unit 2: Merced County.
- Unit 3: San Luis Obispo County.

The primary constituent elements of critical habitat for longhorn fairy shrimp are the habitat components that provide:

(i) Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described below in paragraph (ii), providing for dispersal and promoting hydroperiods of adequate length in the pools;
(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 23 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands;
(iii) Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pool themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding; and
(iv) Structure within the pools described above in paragraph (ii), consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the longhorn fairy shrimp (USFWS 2005).

Reclassification and Delisting Criteria

In the 2012 five-year status review, the downlisting/delisting criteria identified for the Conservancy fairy shrimp include:

1. Habitat Protection: Accomplish habitat protection that promotes vernal pool ecosystem function sufficient to contribute to population viability of the covered species.

    1A. Suitable vernal pool habitat within each prioritized core area for the species is protected.

    1B. Species localities distributed across the species geographic range and genetic range are protected. Protection of extreme edges of populations protects the genetic differences that occur there.

    1C. Reintroduction and introductions must be carried out and meet success criteria.

    1D. Additional localities are permanently protected, if determined essential to recovery goals.

    1E. Habitat protection results in protection of hydrology essential to vernal pool ecosystem function, and monitoring indicates that hydrology that contributes to population viability has been maintained through at least one multi-year period that includes above average, average, and below average local rainfall as defined above, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

2. Adaptive Habitat Management and Monitoring.

    2A. Habitat management and monitoring plans that facilitate maintenance of vernal pool ecosystem function and population viability have been developed and implemented for all habitat protected, as previously discussed in Sections 1 (A-E).

2B. Mechanisms are in place to provide for management in perpetuity and long-term monitoring of habitat protected in Sections 1 (A-E), as previously discussed (funding, personnel, etc.).

2C. Monitoring indicates that ecosystem function has been maintained in the areas protected under Sections 1 (A-D) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

3. Status Surveys.

3A. Status surveys, 5-year status reviews, and population monitoring show populations within each vernal pool region where the species occur are viable (e.g., evidence of reproduction and recruitment) and have been maintained (stable or increasing) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

3B. Status surveys, status reviews, and habitat monitoring show that threats identified during and since the listing process have been ameliorated or eliminated. Site-specific threats identified through standardized site assessments and habitat management planning also must be ameliorated or eliminated.

4. Research.

4A. Research actions necessary for recovery and conservation of the covered species have been identified (these are research actions that have not been specifically identified in the recovery actions but for which a process to develop them has been identified). Research actions (both specifically identified in the recovery actions and determined through the process) on species biology and ecology, habitat management and restoration, and methods to eliminate or ameliorate threats have been completed and incorporated into habitat protection, habitat management and monitoring, and species monitoring plans, and refinement of recovery criteria and actions.

4B. Research on genetic structure has been completed (for species where necessary – for reintroduction and introduction, seed banking) and results incorporated into habitat protection plans to ensure that within and among population genetic variation is fully representative by populations protected in the Habitat Protection section of this document, described previously in Sections 1 (A-E).

4C. Research necessary to determine appropriate parameters to measure population viability for each species have been completed.

5. Participation and Outreach.

5A. Recovery Implementation Team is established and functioning to oversee rangewide recovery efforts.

5B. Vernal Pool Regional working groups are established and functioning to oversee regional recovery efforts.

5C. Participation plans for each vernal pool region have been completed and implemented.

5D. Vernal Pool Regional working groups have developed and implemented outreach and incentive programs that develop partnerships.

Recovery Actions

- Protect vernal pool habitat in the largest blocks possible from loss, fragmentation, degradation, and incompatible uses (USFWS 2005).
- Develop standardized, species-specific guidance for conducting range-wide status surveys for all species addressed in the 2005 Recovery Plan for Vernal Pool Ecosystems of California (USFWS 2005).
- Manage, restore, and monitor vernal pool habitat to promote the recovery of listed species and the long-term conservation of the species of concern (USFWS 2005).
- Conduct research on species addressed in the 2005 Recovery Plan for Vernal Pool Ecosystems of California (USFWS 2005).
- Develop and implement participation programs (USFWS 2005).
- Protection of the known occurrences on private lands in the Carrizo Plain core areas and the currently unprotected Alkali Sink population should be a priority for this species (USFWS 2007, 2012).
- Develop a standardized monitoring method to identify threats and management needs, and to monitor species status and population trends at the Carrizo Plain, San Luis NWR, Vasco Caves Preserve, and Brushy Peak Preserve populations (USFWS 2007, 2012).
- Management and monitoring plans should be prepared for the San Luis NWR Complex and developed for the Alkali Sink conservation bank, the only longhorn fairy shrimp locations remaining without completed management plans. Results from standardized monitoring discussed above, above, should be included in the management plans for all five populations (USFWS 2007, 2012).
- In addition, the following research should be prioritized over the next 5 years: a. Conduct surveys on private lands with a high potential for supporting longhorn fairy shrimp, particularly in areas south of the Brushy Peak and Vasco Caves Preserves and north of the Carrizo Plain, along the western side of the Central Valley; b. Conduct surveys in the area of the Alkali Sink conservation bank; c. Conduct surveys, in the vicinity of Miller Road, north of Los Baños, Merced County, to determine whether or not the single longhorn fairy shrimp found in a road-side ditch represents a self-sustaining population, or represents an anomaly; and, d. Conduct research on vernal pool habitat restoration and longhorn fairy shrimp reintroduction methods to determine the feasibility of introducing longhorn fairy shrimp to biologically appropriate vernal pool regions and soil types (USFWS 2007, 2012).
- Regional vernal pool working groups should be created in regions where longhorn fairy shrimp are known to occur (USFWS 2007, 2012).

Environmental Baseline

The longhorn fairy shrimp only occurs in the Central Valley and its critical habitat was designated in Alameda, Contra Costa, Merced, and San Luis Obispo counties, California. Please refer to information above for the environmental baseline.

Literature Cited

Eriksen, C.H., and D. Belk. 1999. Fairy shrimps of California's puddles, pools and playas. Mad River Press, Eureka, California.

NatureServe. 2015. NatureServe Explorer, An online encyclopedia of life [web application]. Available online at: http://explorer.natureserve.org/.

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USFWS (U.S. Fish and Wildlife Service). 2012. Longhorn Fairy Shrimp (*Branchinecta longiantenna*) 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service Sacramento Fish and Wildlife Office Sacramento, California. June 20. Available online at: https://ecos.fws.gov/docs/tess/species_nonpublish/1926.pdf

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Mount Hermon June Beetle (*Polyphylla barbata*)

Listing Status

The Mount Hermon June beetle was federally listed as endangered in 1997 (62 Federal Register (FR) 3616). The Mount Hermon June beetle was originally listed as an endangered species because of historical loss of habitat and several actual or potential future actions that could further reduce the amount of suitable habitat that supports the species.

Life History and Habitat

The Mount Hermon June beetle is univoltine (i.e., having only one generation per year). As its common name suggests, adult emergence and seasonal activity often begins in June. Historical collection records (Young 1988; BUGGY Database 2003) indicate that adult males have been observed in the months of June, July, August, and September. Specific life history information for the Mount Hermon June beetle is limited, but can be inferred from related species (Buckhorn and Orr 1961; Downes and Andison 1941; Kard and Hain 1990; Lilly and Shorthouse 1971; Van Steenwyk and Rough 1989). Presumably the entire lifecycle (i.e., egg, larva, pupa, and adult) takes 2 to 3 years to complete. The majority of the Mount Hermon June beetle's lifecycle is spent as a subterranean larval stage that feeds on plant roots.

Population Status

The Mount Hermon June beetle is restricted to Zayante sand soils (Bowman and Estrada 1980) derived from ancient sand deposits, known as the Santa Margarita formation (Marangio and Morgan 1987), which are found in the Scotts Valley-Mount Hermon-Felton-Ben Lomond area of the Santa Cruz Mountains. Throughout most of its range, the primary threats to the species are loss of habitat from sand mining and urbanization, and habitat degradation due to invasive plants and unnatural succession. In addition, land uses such as agricultural conversion and recreation (e.g., hiking, horseback riding, mountain biking, and off-road vehicle use) have resulted in loss or degradation of habitat. Herbicide or insecticide use and overcollection by insect collectors are also considered potential threats to the Mount Hermon June beetle and/or its habitat.

Critical Habitat

Critical habitat has not been proposed or designated for this species.

Recovery Plan Information

A recovery plan for the species was published in 1998 (Service 1998). The recovery plan (Service 1998) described three actions necessary to downlist the Mount Hermon June beetle. These actions include: a) protection of the 28 known (as of 1998) collection sites (consisting of 7 discrete areas) of sand parkland habitat through fee-title acquisition, conservation easements, or habitat conservation plans; b) development and implementation of a management plan for the Quail Hollow Ranch County Park; and c) ensuring stable or increasing populations of the Mount Hermon June beetle. The recovery plan states that when the downlisting criteria have been met the species can be considered for delisting if: threats are reduced or eliminated so that populations are capable of persisting without significant human intervention or perpetual endowments are secured for management necessary to maintain the continued existence of the species (Service 1998).

Environmental Baseline

The species only occurs within the State of California, please refer to the information above regarding the species environmental baseline.

Literature Cited

BUGGY Database. 2003. Report of occurrences for the MHJB. Unpublished database maintained by Entomological Consulting Services, Limited. Pleasant Hill, California.

Buckhorn, W.J., and P.W. Orr. 1961. Forest insect conditions in the Pacific Northwest during 1960. Pacific Northwest Forestry and Range Experiment Station. Portland, Oregon. 40 pp.

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Riverside Fairy Shrimp (*Streptocephalus woottoni*)

Listing Status

Riverside fairy shrimp was federally listed as endangered on August 3, 1993, due to habitat loss and degradation due to urban and agricultural development, livestock grazing, off-road vehicle use, trampling, invasion from weedy non-native plants, and other factors (58 FR 41384). Critical habitat was designated on December 4, 2012 (77 FR 72070).

Life History and Habitat

The Riverside fairy shrimp is a small (0.56-0.92 inch) aquatic crustacean in the order Anostraca. The species is generally restricted to vernal pools and other non-vegetated ephemeral (i.e., lasting a short time) pools in Ventura, Riverside, Orange, and San Diego counties of southern California (Service 2021). Vernal pools and vernal swales are often clustered into pool "complexes," and may form dense, interconnected mosaics of small pools, or a sparse scattering of larger pools. Vernal pool complexes that support from one up to many distinct vernal pools are often interconnected by a shared watershed. Both the pool basin and the surrounding watershed are essential for a functioning vernal pool system (Service 2021). The loss of upland vegetation, increased overland water flow due to urban runoff, and alteration of the microtopography can modify the function of vernal pool systems and alter the physiochemical parameters that the Riverside fairy shrimp requires for survival. Because the Riverside fairy shrimp requires ephemerally ponded areas for its conservation, vernal pools are best described from a watershed perspective (Service 2021).

Population Status

Riverside fairy shrimp occurs in 40 vernal pool locations or complexes, including one in Ventura County, five in Orange County, 14 in Riverside County, and 20 in San Diego County (Service 2021). In the 2008 5-year review, we estimated that approximately 45 vernal pool complexes were occupied by Riverside fairy shrimp (Service 2021). The new estimate should not be interpreted as a decrease in the total number of vernal pools or complexes occupied by Riverside fairy shrimp from 2008 to 2021 because of differences in the way pool complexes and occupied habitat have been mapped and tabulated. In fact, we estimate that there are up to nine newly documented Riverside fairy shrimp locations relative to the 2008 review (known as: Tierra Rejada, Fairview Park, Wickerd Road, Lake Skinner Investor, Lake Skinner Multi-Species Reserve, Santa Rosa Plateau, French Valley Donation, Southwest Village Development, and Dennery West) (Service 2021).

Habitat loss and indirect effects from development and fragmentation are ongoing threats but impacts to the species have been reduced in part by the conservation implemented at many locations through regional Habitat Conservation Plans (e.g., City of San Diego Vernal Pool Habitat Conservation Plan and Western Riverside Multiple Species Habitat Conservation Plan). Nonnative plants continue to threaten Riverside fairy shrimp by degrading habitat such that the environmental conditions at some locations may no longer support the species (e.g., expansion of nonnative plants may cause pools to dry more quickly and no longer support the inundation duration needed for Riverside fairy shrimp) (Service 2021).

While Riverside fairy shrimp is protected by the Act, alteration of hydrology remains a threat to the species that was formerly ameliorated to some degree through the implementation of Section 404 of the Clean Water Act. Regulatory changes have eliminated U.S. Army Corps of Engineers oversight of vernal pools and other ephemeral water bodies unless they meet a narrow definition of an adjacent wetland (i.e., water bodies that have a surface connection to a navigable water or territorial sea through flooding in a typical year). Therefore, the Clean Water Act provides less protection against alterations in vernal pools and ephemeral water bodies that may support Riverside fairy shrimp (Service 2021).

Critical Habitat

Designated critical habitat occurs in three units in Ventura, Orange, and San Diego counties, California, for a total of approximately 1,724 acres. The physical and biological features of designated critical habitat include:

1) Ephemeral wetland habitat consisting of vernal pools and ephemeral habitat that have wet and dry periods appropriate for the incubation, maturation, and reproduction of the Riverside fairy shrimp in all but the driest of years, such that the pools: (a) Are inundated (pond) approximately 2 to 8 months during winter and spring, typically filled by rain, and surface and subsurface flow; (b) generally dry down in the late spring to summer months; (c) may not pond every year; and (d) provide the suitable water chemistry characteristics to support the Riverside fairy shrimp. These characteristics include physiochemical factors such as alkalinity, pH, temperature, dissolved solutes, dissolved oxygen, which can vary depending on the amount of recent precipitation, evaporation, or oxygen saturation; time of day; season; and type and depth of soil and subsurface layers. Vernal pool habitat typically exhibits a range of conditions but remains within the physiological tolerance of the species. The general ranges of conditions include, but are not limited to: (i) Dilute, freshwater pools with low levels of total dissolved solids (low ion levels (sodium ion concentrations generally below 70 millimoles per liter (mmol/l))) (ii) Low alkalinity levels (lower than 80 to 1,000 milligrams per liter (mg/l)); and (iii) A range of pH levels from slightly acidic to neutral (typically in range of 6.4–7.1).
2) Intermixed wetland and upland habitats that function as the local watershed, including topographic features characterized by mounds, swales, and low-lying depressions within a matrix of upland habitat that result in intermittently flowing surface and subsurface water in swales, drainages, and pools described in physical and biological feature 1. Associated watersheds provide water to fill the vernal or ephemeral pools in the winter and spring months. Associated watersheds vary in size and therefore cannot be generalized, and they are affected by factors including surface and underground hydrology, the topography of the area surrounding the pool or pools, the vegetative coverage, and the soil substrates in the area. The size of associated watersheds likely varies from a few acres to greater than 100 acres.
3) Soils that support ponding during winter and spring which are found in areas characterized in physical and biological features 1 and 2 that have a clay component or other property that creates an impermeable surface or subsurface layer. Soil series with a clay component or an impermeable surface or subsurface layer typically slow percolation, increase water run-off (at least initially), and contribute to the filling and persistence of ponding of ephemeral wetland habitat where the Riverside fairy shrimp occurs. Soils and soil series known to support vernal pool habitat include, but are not limited to: (a) The Azule, Calleguas, Cropley, and Linne soils series in Ventura County; (b) The Alo, Balcom, Bosanko, Calleguas, Cieneba, and Myford soils series in Orange County; (c) The Cajalco, Claypit, Murrieta, Porterville, Ramona, Traver, and Willows soils series in Riverside County; and (d) The Diablo, Huerhuero, Linne, Placentia, Olivenhain, Redding, Salinas, and Stockpen soils series in San Diego County.

Recovery Plan Information

A recovery plan for Riverside fairy shrimp and other vernal pool species was released on September 3, 1998 (Service 1998) and a clarification to this plan was released on October 1, 2019 (Service 2019). The delisting criteria include the following:

1) All 74 geographic areas and associated vernal pool complexes as identified in Appendices F and G of the 1998 Recovery Plan under each of the specific management areas are protected and managed to ensure long-term viability.

2) The Service must determine that the following factors are no longer present, or continue to adversely affect, Riverside fairy shrimp: (a) the present or threatened destruction, modification, or curtailment of their habitat range; (b) over utilization for commercial, recreational, scientific, or educational purposes; (c) disease or predation; (d) the inadequacy of existing regulatory mechanisms; and (e) other natural and manmade factors affecting their continued existence.
3) Population trends continue to be stable or increasing for 10 consecutive years after threats have been sufficiently ameliorated or managed completion of delisting criterion 2 prior to consideration for delisting.

Environmental Baseline

Since the known occurrences of Riverside fairy shrimp and its designated critical habitat occur entirely within California, the status description above also serves as the baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 1998. Vernal pools of southern California recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon. 113+pp.

Service (U.S. Fish and Wildlife Service). 2019. Recovery plan clarification for the vernal pools of southern California. Department of the Interior. 2 pp.

Service (U.S. Fish and Wildlife Service). 2021. Five-year review: Riverside fairy shrimp (*Streptocephalus woottoni*) 19 pp.

San Diego Fairy Shrimp (*Branchinecta sandiegonensis*)

Listing Status

San Diego fairy shrimp was federally listed as endangered on February 3, 1997, due to habitat destruction and fragmentation from urban development and agricultural conversion, alterations of vernal pool hydrology, off-road vehicle activity, and livestock overgrazing (62 FR 4925). Critical habitat was designated on December 12, 2007 (72 FR 70648).

Life History and Habitat

The San Diego fairy shrimp is a small aquatic crustacean generally restricted to vernal pools in coastal southern California and northwestern Baja California, Mexico. San Diego fairy shrimp are usually observed from January to March when seasonal rainfall fills vernal pools and initiates cyst (egg) hatching. Vernal pools and vernal swales are often clustered into pool "complexes", and may form dense, interconnected mosaics of small pools, or a sparse scattering of larger pools. Vernal pool complexes that support from one up to many distinct vernal pools are often interconnected by a shared watershed. Both the pool basin and the surrounding watershed are essential for a functioning vernal pool system. Loss of upland vegetation, increased overland water flow due to urban runoff, and alteration of the microtopography can modify the function of vernal pool systems, and alter the physiochemical parameters that the San Diego fairy shrimp requires for survival. Because the San Diego fairy shrimp requires ephemerally ponded areas for its conservation, vernal pools are best described from a watershed perspective (Service 2021).

Population Status

There are 51 occurrences of San Diego fairy shrimp that are extant or presumed extant. Since the last status review was conducted in 2008, the distribution of San Diego fairy shrimp has expanded to include one location in Riverside County, where the species was not known to occur previously. This is the first detection of San Diego fairy shrimp east of the coastal range in southern California. Otherwise, the distribution of San Diego fairy shrimp at the county level in the United States has not changed since 2008. The species continues to occur throughout its historic range in San Diego County and Orange County, California. The species was considered extant at two locations in Mexico at the time of listing, known from the general areas of Baja Mar and Valle de las Palmas, but the status of the species at these Mexico locations is unknown (Service 2021).

The magnitude of the threat of development and its associated indirect effects has been reduced through conservation. Conserved lands are areas designated for conservation or are unlikely to be developed due to their inclusion in regional conservation plans, lands conserved by non-profits, and public or quasi-public lands. For example, regional conservation plans include the Southern Subregion and Central/Coastal Habitat Conservation Plans in Orange County and Western Riverside Multiple Species Habitat Conservation Plan (Service 2021).

Off-highway vehicles and human access continue to be threats throughout the range of the species, although fencing to preclude access has occurred at some locations. Non-native plants continue to threaten the species by degrading suitable habitat, and while conservation actions at some locations have alleviated this threat to some degree, it is likely to remain a habitat management challenge in southern California. The threat of habitat fragmentation and the resulting alteration of population dynamics remains due to ongoing development throughout the species range (Service 2021).

Hybridization and competition with *Branchinecta lindahli* may affect San Diego fairy shrimp locations throughout the range of the species. The magnitude of the threat of hybridization and competition, and the ability to manage it, is still being evaluated. Because we understand that *B. lindahli* and hybrids dominate highly disturbed pools (e.g., road ruts), conservation actions should be focused on these degraded habitats, and considerations should be made about whether landowners should remove such features, especially where they exist near intact coastal vernal pools supporting San Diego fairy shrimp. In addition, conservation partners throughout the range of San Diego fairy shrimp should continue to take all necessary precautions to prevent the spread of *B. lindahli* through contaminated equipment and movement of soil (Service 2021).

In addition, a new potential threat of disease has been identified for San Diego fairy shrimp. Wolbachia or similar bacteria can induce cytoplasmic incompatibility. These types of bacteria can also lead to biased sex ratios, parthenogenesis (female asexual reproduction), feminization of males, and a high juvenile male mortality. Because *B. lindahli* can harbor feminizing endoparasitic bacteria, hybridization with San Diego fairy shrimp may lead to genetic and reproduction issues for the listed entity (Service 2021).

While San Diego fairy shrimp is protected by the Act, alteration of hydrology remains a threat to the species that was formerly ameliorated to some degree through the implementation of Section 404 of the Clean Water Act. Regulatory changes have eliminated U.S. Army Corps of Engineers oversight of vernal pools and other ephemeral water bodies unless they meet a narrow definition of an adjacent wetland (i.e., water bodies that have a surface connection to a navigable water or territorial sea through flooding in a typical year). Therefore, San Diego fairy shrimp are more at risk due to alterations in the hydrology of vernal pools and ephemeral water bodies (Service 2021).

Critical Habitat

Designated critical habitat occurs in five units in Orange and San Diego counties, California, for a total of approximately 3,082 acres. The physical and biological features of designated critical habitat include:

1) Vernal pools with shallow to moderate depths (2 to 12 inches) that hold water for sufficient lengths of time (7 to 60 days) necessary for incubation, maturation, and reproduction of the San Diego fairy shrimp, in all but the driest years;
2) Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described in physical and biological feature 1, providing for dispersal and promoting hydroperiods of adequate length in the pools (i.e., the vernal pool watershed); and
3) Flat to gently sloping topography, and any soil type with a clay component and/or an impermeable surface or subsurface layer known to support vernal pool habitat (including Carlsbad, Chesterton, Diablo, Huerhuero, Linne, Olivenhain, Placentia, Redding, and Stockpen soils).

Recovery Plan Information

A recovery plan for San Diego fairy shrimp and other vernal pool species was released on September 3, 1998 (Service 1998) and a clarification to this plan was released on October 1, 2019 (Service 2019). The delisting criteria include the following:

1) All 74 geographic areas and associated vernal pool complexes as identified in Appendices F and G of the 1998 Recovery Plan under each of the specific management areas are protected and managed to ensure long-term viability.
2) The Service must determine that the following factors are no longer present, or continue to adversely affect, San Diego fairy shrimp: (a) the present or threatened destruction, modification, or curtailment of their habitat range; (b) over utilization for commercial, recreational, scientific, or educational purposes; (c) disease or predation; (d) the inadequacy of existing regulatory mechanisms; and (e) other natural and manmade factors affecting their continued existence.

3) Population trends continue to be stable or increasing for 10 consecutive years after threats have been sufficiently ameliorated or managed completion of delisting criterion 2 prior to consideration for delisting.

Environmental Baseline

Since the San Diego fairy shrimp and its designated critical habitat occur mostly within California, except for two potential locations in Mexico for which we have limited information, the status description above also serves as the baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 1998. Vernal pools of southern California recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon. 113+pp.

Service (U.S. Fish and Wildlife Service). 2019. Recovery plan clarification for the vernal pools of southern California. Department of the Interior. 2 pp.

Service (U.S. Fish and Wildlife Service). 2021. Five-year review: San Diego fairy shrimp (*Branchinecta sandiegonensis*) 19 pp.

Smith's Blue Butterfly (*Euphilotes enoptes smithi*)

Listing Status

The Service listed the Smith's blue butterfly as endangered on June 1, 1976 (41 FR 22041 22044). Critical habitat was proposed on February 8, 1977 (42 FR 7972), but was not designated. The decline of the Smith's blue butterfly is attributed to degradation and loss of habitat as a result of urban development, recreational activities in dune habitats, sand mining, military activities, fire suppression in chaparral habitat, and encroachment of exotic plant species.

Life History and Habitat

Smith's blue butterflies co-occur with buckwheat plants that grow in coastal dune, cliffside chaparral, coastal scrub, and coastal grassland communities from the mouth of the Salinas River in Monterey County to San Carpoforo Creek in northern San Luis Obispo County. The Smith's blue butterfly is inextricably dependent upon its host plant species, seacliff buckwheat (*Eriogonum parviflorium*) and coast buckwheat (*Eriogonum latifolium*), during all life stages, except that adults may also feed on nectar from naked buckwheat (*Eriogonum nudum*).

Population Status

Smith's blue butterflies are found within two disjunct areas within their range: 1) a northern area of primarily dune habitats along Monterey Bay north of the Monterey Peninsula, and 2) a southern area of primarily scrub, chaparral, and grassland habitats of the Carmel Valley and Big Sur Coast south of the Monterey Peninsula (Service 2006, p. 6). Long-term monitoring has only been conducted on the Salinas River National Wildlife Refuge since 2015 (Service 2020b, p. 1). Most of our knowledge of the distribution of the Smith's blue butterfly is the result of singular observations made in the past 30 years. Therefore, the number, size, and persistence of colonies throughout the range of the species are poorly understood.

Urban development, recreational activities, and other activities continue to result in habitat loss and degradation. Urban development, introduction of invasive plant species and recreational use have fragmented and continue to fragment habitat for the Smith's blue butterfly. This fragmentation has several ramifications for the Smith's blue butterfly. The quality of the remaining suitable habitat is reduced, the distance dispersing adults must travel to reach the next island of suitable habitat is increased, the entire metapopulation structure is potentially disrupted, and genetic diversity is reduced. Overall, groups of Smith's blue butterflies occupying smaller, more isolated stands of suitable habitat are more likely to be extirpated by stochastic or anthropogenic factors.

Critical Habitat

Critical habitat has not been proposed or designated for this species.

Recovery Plan Information

The Service completed a recovery plan for the species on November 9, 1984 (Service 1984). The Smith's blue butterfly recovery plan objectives focus on protection of those localities that were known when the plan was published (Service 1984). However, due to changes in our knowledge of the subspecies' range and the threats that it faces, the objectives are largely obsolete. The general recovery needs of the Smith's blue butterfly include conserving and managing existing habitat, maintaining and improving connectivity between areas of habitat, and increasing the amount of occupied habitat through restoration efforts. Although the recovery plan is outdated, several of the recovery actions are still valid, including: (1) Revegetating existing blow-out areas with native plants and removing exotic plants; (2) Controlling off-road vehicle use of dunes; (3) Carrying out prescribed burns; (4) Iceplant and Holland dune grass eradication; and, (5) Developing public awareness.

Environmental Baseline

The species only occurs within the State of California, please refer to information above regarding the species environmental baseline.

Literature Cited

[Service] U.S. Fish and Wildlife Service. 2006. Smith's blue butterfly (*Euphilotes enoptes smithi*) 5-year review: Summary and evaluation.

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Valley Elderberry Longhorn Beetle (*Desmocerus californicus dimorphus*) and its Critical Habitat

Listing Status

The Valley elderberry longhorn beetle was listed as threatened on August 8, 1980 (45 FR 52803). Critical habitat was designated for the Valley elderberry longhorn beetle on August 8, 1980 (45 FR 52803).

Life History and Habitat

The valley elderberry longhorn beetle is a habitat specialist and spends almost its entire life history on the sole host plant, blue elderberry. The species is dependent on the blue elderberry plant for larval and adult life stages. Blue elderberries are an important component of riparian ecosystems in California. Within the range of the species, habitats range from lowland riparian forest to foothill oak woodlands, with elevation ranges from 18.3 to 689 m (60 to 2,260 ft.). It has occasionally been found with these plants in more upland habitats, including scrubland and chaparral habitats. The range of the species is bounded by the Cascade Range to the north, Sierra Nevada to the east, Tehachapi Mountains to the south, and coastal ranges and San Francisco Bay to the west (79 FR 55874; NatureServe 2015). Historically, the riparian forests in the Central Valley consisted of several canopy layers with a dense undergrowth, and included Fremont cottonwood (*Populus fremontii*), California sycamore (*Platanus racemosa*), *willows* (Salix sp.), valley oak (*Quercus lobata*), box elder (*Acer negundo* var. *californicum*), Oregon ash (*Fraxinus latifolia*), and several species of vines (e.g., California grape [*Vitis californica*] and poison oak [*Toxicodendron diversilobum*]). These plant communities encompass several remaining natural and semi-natural floristic vegetation alliances and associations in the Great Valley Ecoregion of California. Elderberry shrubs have been found most frequently in mixed plant communities, and in several types of habitat, including non-riparian locations, as both an understory and overstory plant, with valley elderberry longhorn beetle adults and exit holes created by the valley elderberry longhorn beetle found most commonly in riparian woodlands and savannas. The species uses moist valley oak woodlands suitable for blue elderberry plants. Shrub characteristics and other environmental factors appear to have an influence on use by the valley elderberry longhorn beetle in some recent studies, with more exit holes in shrubs in riparian than non-riparian scrub habitat types (USFWS 1984; 79 FR 55874).

The valley elderberry longhorn beetle reproduces through oviparity, with females laying eggs on leaves of the host plant. Females lay eggs singly; the number of eggs are varied, ranging from 8 to 110 in a laboratory setting. In one study, a total of 136 larvae (and an additional 44 eggs that did not hatch) were produced by one captive female valley elderberry longhorn beetle. Hatching success has been estimated at 50 to 67 percent of eggs laid, but survival rates of larvae are unknown. Females lay eggs on elderberry leaves and at the junction of leaf stalks and main stems, with all eggs laid on new growth at the outer tips of elderberry branches. Based on observations of females along the Kings River, females laid eggs at locations on the elderberry branch where the probing ovipositor (i.e., the female's egg-laying organ) could be inserted. In a laboratory setting, the majority of eggs laid were attached to leaves and stems of foliage (provided as food), with a preference for leaf petiole-stem junctions, leaf veins, and other areas containing crevices and depressions. Eggs are approximately 2.3 to 3.0 mm (0.09 to 0.12 in.) long and reddish-brown in color, with longitudinal ridges. Eggs are initially white to bright yellow, then darken to brownish white and reddish (79 FR 55874; USFWS 1984; USFWS 2006). Individuals are very dependent on their host plant, blue elderberry (*Sambucus* spp.). The first instars larvae bore to the center of elderberry stems, where they develop and feed on the pith. Prior to forming their pupae, the elderberry wood boring larvae chew through the bark and then plug the holes with wood shavings. The larvae crawl back to their pupal chamber, which they pack with grass. In the pupal chamber, the larvae metamorphose into their pupae and then into adults, whereupon they emerge between mid-March and mid-June (peak late April to mid- May) and breed. The short adult life stage, including breeding, coincides with the bloom period of the elderberry. The species needs woodland habitat suitable for growing blue elderberry plants for reproduction. Oviposition occurs on stems with diameters greater than about 2.5 cm (1 in.). The larval stage reportedly often takes 2 years inside the host plant; however, a 1-year cycle has been observed in a laboratory setting. Adults live from a few days to a few weeks after emergence, and die within 3 months (79 FR 55874; USFWS 1984; USFWS 2006).

The valley elderberry longhorn beetle is an herbivorous specialist that feeds almost exclusively on blue elderberry (*Sambucus cerulea*) throughout all stages of its life. Adults feed on the foliage and perhaps flowers (and nectar) of the host plant, which are present from March through early June. Larva feed on the pith, and emergence of the adult beetle from the pith of the host is synchronized with the host plant bloom period. The species' food resources are limited in distribution. Adults are active from March until June, while larvae are active year-round. California elderberry longhorn beetle (*D. c. californicus*) may compete with Valley elderberry longhorn beetle, because they can share food sources and their ranges can overlap. The species may also be preyed upon by insectivorous birds, lizards, European earwigs (*Forficula auricularia*), and Argentine ants (*Linepithema humile*). The species is entirely dependent on blue elderberry for feeding, and requires the riparian moist woodlands in which the plant grows. To serve as habitat, the shrubs apparently must have stems 2.5 cm (1 in.) or greater in diameter at ground level, so that larva may bore into them (79 FR 55874; USFWS 1984; USFWS 2006).

The valley elderberry longhorn beetle has very limited dispersal; it usually stays on or near the host plant for the duration of its life. Dispersal distance of an adult valley elderberry longhorn beetle from its emergent site is estimated to be 50 m (164 ft.) or less (USFWS 1984; 79 FR 55874).

Population Status

Rangewide Status of the Species

Although the entire historical distribution of the valley elderberry longhorn beetle is unknown, extensive destruction of riparian forests of the Central Valley during the past 150 years strongly suggests that the beetle's range has decreased and become greatly fragmented. Museum records indicate that the beetle has been collected in four central California counties: Merced, Sacramento, Solano, and Yolo (USFWS 1984).

When the valley elderberry longhorn beetle was listed in 1980, it was known from 10 occurrence records at three locations: the Merced River (Merced County), the American River (Sacramento County), and Putah Creek (Yolo County) of the Central Valley of California. Subsequent surveys throughout the Central Valley discovered more locations and the current presumed historical range is now believed to extend from Shasta County to Madera County below 500 feet in elevation (152.4 meters) (79 FR 55874). Although different ranges for the beetle have been proposed in the past, the current presumed range relies only on verifiable sightings or specimens of adult male Valley elderberry longhorn beetles (79 FR 55874). Previous iterations of the presumed range used both female sightings and exit holes to determine Valley elderberry longhorn beetle presence. Both of these metrics are unreliable as female California elderberry longhorn beetle (*Desmocerus californicus californicus*) and Valley elderberry longhorn beetles are indistinguishable in the field and exit holes cannot be accurately assigned to either species (USFWS 2019).

Population Summary

Occupancy of the valley elderberry longhorn beetle within the presumed historical range over the past 16 years has occurred in approximately 18 hydrologic units and 36 geographical locations in the Central Valley. The overall trend of valley elderberry longhorn beetle occupancy was moderately downward when comparing the 1991 and 1997 survey data. The species trend is an overall decline of approximately 90 percent since the 1800s (79 FR 55874). With regard to population size, no true estimates have been made due to the cryptic nature of the species. Based on a spatial analysis of valley elderberry longhorn beetle populations in the Central Valley, Talley concluded that the several-hundred-meter distances observed between local aggregations of the species supports a limited migration distance for this species. An integrative approach to all three spatial frameworks (patch, gradient, and hierarchical) best defined a population structure for the valley elderberry longhorn beetle. This population structure can be characterized as patchy-dynamic, with regional distributions made up of local aggregations of populations. These localized populations are defined by both broad-scale or continuous factors associated with elderberry shrubs (e.g., shrub age or densities) and environmental variables associated with riparian ecosystems (e.g., elevation, associated trees) that themselves have patch, gradient, and hierarchical structures (79 FR 55874).

Threats

Threats to this species include:

- A significant amount of riparian vegetation (of which a portion contained elderberry shrubs) has been converted to agriculture and urban development since the mid-1800s. Agricultural development has probably reached close to its maximum extent in the Central Valley. However, conversion of agricultural lands into urban development continues at a significant rate, and as a consequence continues to affect beetle habitat by eliminating elderberries along irrigation channels and hedgerows, eliminating the buffering effect, and precluding the potential to restore riparian forest vegetation (79 FR 55874).
- Projects that may have impacted, or could impact, valley elderberry longhorn beetle habitat include: levee construction; bank protection; channelization; facility improvements or ongoing maintenance activities, including clearing and snagging; construction of bypasses; and construction of ancillary features (such as overflow weirs and outfall gates).
- Average temperatures have been rising in the Central Valley of California, and this trend will likely continue because of climate change. Climate change may also affect precipitation and the severity, duration, or periodicity of drought.
- Invasive nonnative plants may be impacting the species through modification or loss of habitat due to competition for space and resources with its host plant, but additional information is needed to evaluate the magnitude of this threat.
- The invasive, nonnative Argentine ant (*Linepithema humile*) has been identified as a potential threat to the valley elderberry longhorn beetle. This ant is both an aggressive competitor with, and predator on, several species of native fauna; it is spreading throughout California riparian areas and displacing assemblages of native arthropods. Although additional studies are needed to better characterize the level of predation threat to the valley elderberry longhorn beetle from Argentine ants, the best available data indicate that this invasive species is a predation threat to the valley elderberry longhorn beetle, and is likely to expand to additional areas within the range of the valley elderberry longhorn beetle in the foreseeable future (79 FR 55874).
- While State and federal laws provide some degree of protection for riparian vegetation and valley elderberry longhorn beetles, other types of local zoning or changes in open space designations in the future could affect the beetle (79 FR 55874).

Many pesticides are commonly used in the valley elderberry longhorn beetle's range. These pesticides include insecticides (most of which are broad-spectrum and likely toxic to the beetle) and herbicides (which may harm or kill its elderberry host plants).

Five-Year Status Review

On September 26, 2009, a 5-year status review was conducted for the Valley elderberry longhorn beetle (USFWS 2006). The USFWS concluded that the delisting of the species was given a reclassification number of "2" indicating that it is an unpetitioned action with a high management impact. On September 17, 2014, the USFWS withdrew the proposed rule to remove the Valley elderberry longhorn beetle from the Federal List of Endangered and Threatened Wildlife under the Endangered Species Act of 1973, as amended (79 FR 55874). On September 26, 2023, the U.S. Fish and Wildlife Service completed a five-year status review of the valley elderberry longhorn beetle and concluded that this species' threatened status would remain unchanged (USFWS 2023).

Critical Habitat

Critical habitat was designated for the Valley elderberry longhorn beetle on August 8, 1980 (45 FR 52803). Primary constituent elements were not defined in this designation.

(1) Sacramento Zone. An area in the city of Sacramento enclosed on the north by the Route 160 Freeway, on the west and southwest by the Western Pacific railroad tracks, and on the east by Commerce Circle and its extension southward to the railroad tracks.
(2) American River Parkway Zone. An area of the American River Parkway on the south bank of the American River, bounded on the north by latitude 30"37'30"N, on the west and southwest by Elmanto Drive from its junction with Ambassador Drive to its extension to latitude 38"37'30"N, and on the south and east by Ambassador Drive and its extension north to latitude 38'37'30"N. Goethe Park, and that portion of the American River Parkway northeast of Goethe Park, west of the Jedediah Smith Memorial Bicycle Trail, and north to a line extended eastward from Palm Drive.

Recovery Plan Information

On June 28, 1984, the USFWS issued the Recovery Plan for the Valley elderberry longhorn beetle (USFWS 1984). On October 4, 2019, the USFWS issued the Revised Recovery Plan for the Valley elderberry longhorn beetle (USFWS 2019).

Recovery Actions

- Acquire, enhance, restore, and protect suitable habitat for the Valley elderberry longhorn beetle. This action involves land acquisition, habitat management, and site improvements.
- Develop management and monitoring plans for protected riparian areas that consider the threats and needs of the Valley elderberry longhorn beetle. Plans should include status and demographic monitoring, non-native predator control, habitat enhancement, and other needed activities that may increase the resilience of the Valley elderberry longhorn beetle.
- Include Valley elderberry longhorn beetle conservation as a component of state and local programs to protect riparian habitat.
- Complete studies that focus on: habitat patch size, elderberry density, and connectivity that influence the viability of individual Valley elderberry beetle populations; influences on demography and reproductive rates of the Valley elderberry longhorn beetle; and factors that influence or limit adult dispersal.
- Conduct surveys for the Valley elderberry longhorn beetle in each HUC8 subbasin to monitor and assess the health of known populations and to locate new populations.

Environmental Baseline

The Valley elderberry longhorn beetle and its designated critical habitat only occur in the Central Valley, California. Please refer to information above for the environmental baseline.

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Vernal Pool Fairy Shrimp (*Branchinecta lynchi*)

Listing Status

The vernal pool fairy shrimp was listed as threatened on September 19, 1994 (59 FR 48136).

Life History and Habitat

Physical Description

The vernal pool fairy shrimp (*Branchinecta lynchi*) is a small freshwater crustacean, varying in size from 3 to 38 millimeters (0.12 to 1.5 inches [in.] long) and belonging to an ancient order of branchiopods, the Anostraca. Like other anostracans, it has stalked compound eyes and eleven pairs of phyllopods (swimming legs that also function as gills). The vernal pool fairy shrimp is genetically distinct from other *Branchinecta* species, and is distinguished by the morphology of the male's second antenna and the female's third thoracic segment (on the middle part of its body) (USFWS 2007).

Habitat

Vernal pool fairy shrimp have an ephemeral lifecycle and exist only in vernal pools or vernal pool-like habitats; the species does not occur in riverine, marine, or other permanent bodies of water. The vernal pool fairy shrimp is endemic to California and the Agate Desert of southern Oregon. It has the widest geographic range of the federally-listed vernal pool crustaceans, but it is seldom abundant where found, especially where it co-occurs with other species.The vernal pool fairy shrimp occupies a variety of different vernal pool habitats, from small, clear, sandstone rock pools to large, turbid, alkaline, grassland valley floor pools (USFWS 2005). The vernal pool fairy shrimp occurs only in cool-water pools. Whatever the habitat, the wetlands in which this species is found are small (less than 200 square meters [m2] [2,153 square feet (sq. ft.)]) and shallow (mean 5 centimeters [cm] [2 in.]); however, this species occasionally inhabits large (44,534 m2 [478,371 sq. ft.]) and very deep (122 cm [48 in.]) habitats (NatureServe 2015). Although the vernal pool fairy shrimp has been collected from large vernal pools, including one exceeding 10 hectares (ha) (25 acres [ac.]) in area, it tends to occur primarily in smaller pools, and is most frequently found in pools measuring less than 0.02 ha (0.05 ac.) in area. The vernal pool fairy shrimp typically occurs at elevations from 10 meters (m) (33 feet [ft.]) to 1,220 m (4,003 ft.), although two sites in the Los Padres National Forest have been found to contain the species at an elevation of 1,700 m (5,600 ft.).The vernal pool fairy shrimp has been collected at water temperatures as low as 4.5°C (40°F), and has not been found in water temperatures above about 24°C (75°F). The species is typically found in pools with low to moderate amounts of salinity or total dissolved solids. Vernal pools are mostly rain-fed, resulting in low nutrient levels and dramatic daily fluctuations in pH, dissolved oxygen, and carbon dioxide. Although there are many observations of the environmental conditions where vernal pool fairy shrimp have been found, there have been no experimental studies investigating the specific habitat requirements of this species. In Oregon, the vernal pool fairy shrimp is found in two distinct vernal pool habitats. The species occurs on alluvial fan terraces associated with Agate-Winlo soils on the Agate Desert, and in the Table Rocks area on Randcore-Shoat soils underlain by lava bedrock. These vernal pool habitats represent the northern extent of the vernal pool fairy shrimp. In the Western Riverside County and Santa Barbara vernal pool regions, the vernal pool fairy shrimp occurs on inland mesas and valleys, on weak to strongly alkaline soils. In the Los Padres National Forest in Ventura County, it is known to occur in atypical habitats that consist of vernal pools located under a Jeffrey pine (*Pinus jeffreyi*) canopy that does not possess a grass understory. In general, the vernal pool fairy shrimp has a sporadic distribution in the vernal pool complexes, with most pools being uninhabited by the species (USFWS 2007). The thermal and chemical properties of vernal pool waters are two of the primary factors affecting the distributions of specific fairy shrimp species (including the vernal pool fairy shrimp), or their appearance from year to year. Different species may appear in pools from one year to the next, depending on whether the pools fill at a different time of the year. In years with warm winter rains, vernal pool fairy shrimp do not hatch in at least a portion of their range. In years with low amounts of precipitation or atypical timing of precipitation (or in substandard habitat), vernal pool species may die off before reproducing (Eriksen and Belk 1999). In some cases, vernal pool fairy shrimp will cease to be found in pools where they were formerly found (USFWS 2007).

Taxonomy

The vernal pool fairy shrimp was first collected between 1874 and 1941, when it was described incorrectly as Colorado fairy shrimp (*Branchinecta coloradensis*). Its identity as a separate species was resolved in 1990. Subsequent genetic analysis has confirmed that the vernal pool fairy shrimp is a distinct species (USFWS 2007). The species was named in honor of James B. Lynch, a systematist of North American fairy shrimp (USFWS 2005). Vernal pool fairy shrimp closely resemble Colorado fairy shrimp (*Branchinecta coloradensis*). However, there are differences in the shape of a small mound-like feature at the base of the male's antennae, called the pulvillus. The Colorado fairy shrimp has a round pulvillus, while the vernal pool fairy shrimp's pulvillus is elongate. The vernal pool fairy shrimp can also be identified by the shape of a bulge on the distal, or more distant end, of the antennae. This bulge is smaller and less spiny on the vernal pool fairy shrimp. The female Colorado fairy shrimp's brood pouch is longer and more cylindrical than the vernal pool fairy shrimp's. Female vernal pool fairy shrimp also closely resemble female midvalley fairy shrimp. These two species can be distinguished by the number and placement of lobes on their backs, called dorsolateral thoracic protuberances. Vernal pool fairy shrimp have paired dorsolateral thoracic protuberances on the third thoracic segment that are not found in the midvalley fairy shrimp (USFWS 2005).

Current Range

Since the vernal pool fairy shrimp’s listing, surveys of vernal pools and other temporary waters throughout the western United States have resulted in an increase in the shrimp’s known range. In 1998, the shrimp was discovered in two distinct vernal pool habitats in Jackson County, Oregon. The known range of the vernal pool fairy shrimp was also extended due to its detection in one pool at the Napa Airport at the southeastern edge of the Lake-Napa Vernal Pool Region (USFWS 2007). The vernal pool fairy shrimp is currently found in 28 counties across the Central Valley and coast ranges of California, and in Jackson County in southern Oregon. The species occupies a variety of vernal pool habitats, and occurs in 11 of the 17 vernal pool regions and 45 of the 85 core recovery areas identified in California (USFWS 2005).

Population Status

The vernal pool fairy shrimp is much less restricted in range than other species of fairy shrimp; however, it is not abundant at any site (NatureServe 2015). Surveys (and monitoring) of vernal pool fairy shrimp generally only record presence/absence in pools and do not provide information on shrimp abundance in pools. At the time of listing in 1994, the populations represented either geographic clusters of occurrence records or single occurrences from areas with extant vernal pool habitat. The 32 extant populations were described for the following counties, with the number of populations in parentheses: Shasta County (1), Tehama County (4), Glenn County (1), Butte County (1), Yuba County (1), Placer County (1), El Dorado County (1), Sacramento County (2), Solano County (1), Contra Costs County (1), Alameda County (1), Merced County (4), Madera County (2), Fresno County (2), San Benito County (1), Tulare County (4), San Luis Obispo County (1), Santa Barbara County (1), and Riverside County (2) (USFWS 2007).

Currently, the vernal pool fairy shrimp is known from 13 pool regions. At the time of listing, 178 extant occurrences were known from 32 putative populations, based on proximity of known occurrences. There are currently 400 recorded occurrences (USFWS 2007).The USFWS has information to indicate that the shrimp is still extant in most of the putative populations, although loss and fragmentation of vernal pool habitat has occurred in and around most of the 1994 populations, potentially decreasing their viability. Without species specific monitoring, the USFWS does not know whether populations of vernal pool fairy shrimp are declining (USFWS 2007).

Critical Habitat

The Fish and Wildlife Service (Service) designated approximately 858,846 acres (ac) (347,563 hectares (ha)) of critical habitat for 4 vernal pool crustaceans and 11 vernal pool plants in 34 counties in California and 1 county in southern Oregon in a final rule of August 11, 2005 (70 FR 46924). That rule designated critical habitat for the 15 vernal pool species collectively. Pursuant to that rule, on February 10, 2006, the Service published species-specific unit descriptions and maps for the 15 species. This rule specifically identifies the critical habitat for each individual species identified in the August 11, 2005, final rule.35 units are designated as critical habitat, totaling 597,821 acres:

- Unit 1: Jackson County, Oregon. Unit 1A: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Shady Grove. Unit 1B: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Shady Grove. Unit 1C: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Shady Grove. Unit 1D: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 1E: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Shady Grove. Unit 1F: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Shady Grove. Unit 1G: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point.
- Unit 2: Jackson County, Oregon. Unit 2A: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 2B: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 2C: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 2D. Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 2E: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 2E: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point.
- Unit 3: Jackson County, Oregon. Unit 3A: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point. Unit 3B: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Eagle Point, Sams Valley. Unit 3C: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Sams Valley.
- Unit 4: Jackson County, Oregon. Unit 4A: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Sams Valley. Unit 4B: Jackson County, Oregon. From USGS 1:24,000 scale quadrangle Sams Valley.
- Unit 5: Shasta County, California. From USGS 1:24,000 scale quadrangle Palo Cedro, Enterprise, Balls Ferry, Cottonwood.
- Unit 6: Tehama County, California. From USGS 1:24,000 scale quadrangle Red Bluff East, Red Bluff West, Gerber, West of Gerber, Corning, Henleyville.
- Unit 7: Tehama County, California. Unit 7A: Tehama County, California. From USGS 1:24,000 scale quadrangle Acorn Hollow and Richardson Springs NW. Unit 7B: Tehama County, California. From USGS 1:24,000 scale quadrangle Sloughhouse. Unit 7C: Tehama County, California. From USGS 1:24,000 scale quadrangle Richard Springs NW. Unit 7D: Tehama and Butte counties, California. From USGS 1:24,000 scale quadrangle Campbell Mound, Richardson Springs, and Richardson Springs NW. Unit 7E: Butte County, California. From USGS 1:24,000 scale quadrangle Richardson Springs. Unit 7F: Butte County, California, California. From USGS 1:24,000 scale quadrangle Richardson Springs.
- Unit 8: Tehama and Glenn counties, California. From USGS 1:24,000 scale quadrangle Kirkwood and Black Butte Dam.

- Unit 9: Butte County, California. From USGS 1:24,000 scale quadrangle Chico.
- Unit 11: Yuba County, California. From USGS 1:24,000 scale quadrangle Browns Valley and Wheatland.
- Unit 12: Placer County, California. Unit 12A: Placer County, California. From USGS 1:24,000 scale quadrangle Lincoln. Unit 12B: Placer County, California. From USGS 1:24,000 scale quadrangle Lincoln.
- Unit 13: Sacramento County, California. From USGS 1:24,000 scale quadrangle Carmichael.
- Unit 14: Sacramento and Amador County, California. Unit 14A: Sacramento and Amador County, California. From USGS 1:24,000 scale quadrangle Carbondale, Sloughhouse, Goose Creek, and Clay. Unit 14B: Sacramento County, California. From USGS 1:24,000 scale quadrangle Sloughhouse.
- Unit 16: Solano County, California. Unit 16A: Solano County, California. From USGS 1:24,000 scale quadrangle Elmira, Denverton, and Fairfield South. Unit 16B: Solano County, California. From USGS 1:24,000 scale quadrangle Elmira and Denverton. Unit 16C: Solano County, California. From USGS 1:24,000 scale quadrangle Elmira. Unit 16D: Solano County, California. From USGS 1:24,000 scale quadrangle Dozier.
- Unit 17: Napa County, California. From USGS 1:24,000 scale quadrangle Cuttings Wharf.
- Unit 18: San Joaquin County, California. From USGS 1:24,000 scale quadrangle Valley Springs SW, Linden, Farmington, and Peters.
- Unit 19: Contra Costa County, California. Unit 19A: Contra Costa County, California. From USGS 1:24,000 scale quadrangle Brentwood and Antioch South. Unit 19B: Contra Costa County, California. From USGS 1:24,000 scale quadrangle Clifton Court Forebay and Byron Hot Springs. Unit 19C: Alameda County, California. From USGS 1:24,000 scale quadrangle Altamont and Livermore.
- Unit 20: Stanislaus County, California. From USGS 1:24,000 scale quadrangle Ripon.
- Unit 21: Stanislaus County, California. Unit 21A: Stanislaus County, California. From USGS 1:24,000 scale quadrangle Paulsell and Montpelier. Unit 21B: Stanislaus, Merced, and Mariposa counties, California. From USGS 1:24,000 scale quadrangle La Grange, Cooperstown, Paulsell, Turlock Lake, Snelling, Montpelier and Merced Falls. Unit 21C: Merced County, California. From USGS 1:24,000 scale quadrangle Turlock Lake.
- Unit 22: Merced County, California. From USGS 1:24,000 scale quadrangle Merced Falls, Snelling, Indian Gulch, Haystack Mtn., Yosemite Lake, Winton, Owens Reservoir, Planada, Le Grand, Plainsburg, and Merced.

Primary Constituent Elements/Physical or Biological Features

Critical habitat units are designated for Jackson County, Oregon, and Alameda, Amador, Butte, Contra Costa, Fresno, Kings, Madera, Mariposa, Merced, Monterey, Napa, Placer, Sacramento, San Benito, San Joaquin, San Luis Obispo, Santa Barbara, Shasta, Solano, Stanislaus, Tehama, Tulare, Ventura, and Yuba counties, California. The primary constituent elements of critical habitat for vernal pool fairy shrimp (*Branchinecta lynchi*) are the habitat components that provide:

(i) Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described below in paragraph (ii), providing for dispersal and promoting hydroperiods of adequate length in the pools;

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 18 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands;

(iii) Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding; and

(iv) Structure within the pools described above in paragraph (ii), consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

Recovery Plan Information

Recovery Actions

Recovery actions for this species include the following:

- Protect vernal pool habitat in the largest blocks possible from loss, fragmentation, degradation, and incompatible uses (USFWS 2005).
- Manage, restore, and monitor vernal pool habitat to promote the recovery of listed species and the long-term conservation of the species of concern (USFWS 2005).
- Conduct range-wide status surveys and status reviews for all species addressed in this recovery plan to determine species status and progress toward achieving recovery of listed species and long-term conservation of species of concern (USFWS 2005).
- Conduct research and use results to refine recovery actions and criteria, and guide overall recovery and long-term conservation efforts (USFWS 2005).
- Develop and implement participation programs (USFWS 2005).
- Research: Conduct coordinated research for the vernal pool fairy shrimp that assesses the number of demographically independent units that are persisting, directly estimates levels of migration between units (to determine likelihood of recolonization), determines long- term trends in population growth, and experimentally measures probabilities of local extinction and recolonization. Research should address egg bank dynamics and trends in egg bank abundance over time. Comparisons between isolated pools, pools in fragmented habitat, pools in intact vernal pool complexes, and a variety of created pools should also be assessed. The long-term effects on the hydrology of vernal pools from development-related alterations to vernal pool sub-watersheds should be assessed. Efforts should lead to determinations of appropriate hydrology (or upland) buffers. Additional research needs include a systematic survey to update the status of known California Natural Diversity Database occurrences. The probability of detecting the species under USFWS' survey guidelines for vernal pool crustaceans should also be conducted (USFWS 2007).
- Recovery: Additional preservation of known extant occurrences is needed to reduce habitat threats and reach recovery goals outlined in the 2005 Recovery Plan. Preservation of large blocks of vernal pool habitat that contain complete or large portions of vernal pool complexes is needed for this species. USFWS should also work with private landowners for the conservation of habitat for the vernal pool fairy shrimp through conservation easements or other methods (USFWS 2007).
- Monitoring: Develop and implement a standardized formal monitoring program that collects data in sufficient detail to evaluate species status, and examine changes in population dynamics and community composition (USFWS 2007).
- Habitat Management: Develop management indicators for identifying potential problems and assessing ecosystem health as it pertains to vernal pool crustaceans. Establish requirements for appropriate management of vernal pool landscapes. Establish improved guidelines, monitoring

protocols, and success criteria for appropriate management of vernal pool landscapes and constructed and restored pools (USFWS 2007).

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Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)

Listing Status

The vernal pool tadpole shrimp was listed as endangered on September 19, 1994 (59 FR 48136).

Population Status

There are 226 occurrences of this species within 19 counties; however, the number of populations represented (species occurrences with a separation of greater than 0.25 mile [mi.]), is unknown (USFWS 2007).

Although vernal pool tadpole shrimp are spread over a wide geographic range, their habitat is highly fragmented and they are uncommon where they are found (USFWS 2007). Several to several hundred individuals can be found in any given water body (NatureServe 2015). At the time of listing in 1994, vernal pool tadpole shrimp were known from 18 populations, extending from east of Redding, Shasta County, southward to the San Luis NWR, Merced County, in the Central Valley, with a disjunct population at the San Francisco NWR, Alameda County (59 FR 48136). However, the precise location and extent of those populations and the number of counties occupied at that time are not known (USFWS 2005). There are 226 occurrences within 19 counties; however, the number of populations represented (species occurrences with a separation of greater than 0.25 mi.), is unknown. A given pool may support several to several hundred individuals within a given water body (NatureServe 2015). Annual surveys have not occurred at all sites with known vernal pool tadpole shrimp occurrences. Where surveys have been conducted for vernal pool tadpole shrimp, they were designed for the purpose of determining the presence of species in the areas of proposed development or road projects, and have generally been limited in scope, focusing on a single parcel or occurrence. Surveys are generally not conducted in a manner to facilitate determination of the population trends of this species. No trends either downward or upward have been reported at any of the monitored sites; however, the accelerated loss and fragmentation of vernal pool tadpole shrimp habitat, particularly in the Southeastern Sacramento Valley Vernal Pool Region, is expected to result in markedly decreased long-term viability of this species. Populations in the Vina Plains in Tehama County may be susceptible, as described in the 1994 final rule, to decreased fecundity due to parasitization by flukes (Trematoda) of an undetermined species (USFWS 2007).

Current Range

The vernal pool tadpole shrimp is currently distributed across the Central Valley of California and in the San Francisco Bay Area. The species' distribution has been greatly reduced from historical times as a result of widespread destruction and degradation of its vernal pool habitat. Vernal pool habitats in the Central Valley now represent only about 25 percent of their former area, and remaining habitats are considerably more fragmented and isolated than during historical times. Vernal pool tadpole shrimp are uncommon even where vernal pool habitats occur (USFWS 2005). The vernal pool tadpole shrimp has a patchy distribution across the Central Valley of California, from Shasta County southward to northwestern Tulare County, with isolated occurrences in Alameda and Contra Costa counties. The California Natural Diversity Database (CNDDB) currently reports 226 occurrences of vernal pool tadpole shrimp in the following 19 counties: Alameda, Butte, Colusa, Contra Costa, Fresno, Glenn, Kings, Merced, Placer, Sacramento, San Joaquin, Shasta, Solano, Stanislaus, Sutter, Tehama, Tulare, Yolo, and Yuba. Sacramento County contains 28 percent, the greatest amount, of the known occurrences (USFWS 2007).

Critical Habitat

Critical habitat for this species was originally designated on August 6, 2003. On August 11, 2005, the Fish and Wildlife Service (Service), re-evaluated the economic exclusions made to the previous final rule

(68 FR 46683; August 6, 2003), which designated critical habitat pursuant to the Endangered Species Act of 1973, as amended (Act), for 4 vernal pool crustaceans and 11 vernal pool plants. A total of approximately 858,846 acres (ac) (347,563 hectares (ha)) of land are now designated critical habitat. This reflects exclusion of lands from the final designation for economic reasons, pursuant to section 4(b)(2) of the Act. This designation also reflects the lands previously confirmed for exclusion under 4(b)(2) of the Act for noneconomic reasons (70 FR 11140; March 8, 2005). The non-economic exclusions include the boundaries of various Habitat Conservation Plans, National Wildlife Refuges and National fish hatchery lands (33,097 ac (13,394 ha)), State lands within ecological reserves and wildlife management areas (20,933 ac (8,471 ha)), Department of Defense lands within Beale and Travis Air Force Bases as well as Fort Hunter Liggett and Camp Roberts Army installations (64,259 ac (26,005 ha)), Tribal lands managed by the Mechoopda Tribe (644 ac (261 ha)), and the Santa Rosa Plateau Ecological Reserve (10,200 ac (4,128 ha)) from the final designation.

Critical habitat for the vernal pool tadpole shrimp (*Lepidurus packardi*) in California consists of the following areas:

(1) Subunit 5A; Siskiyou County, California. From USGS 1:24,000 scale quadrangle Timbered Crater.
(2) Subunit 5B; Modoc and Shasta County, California. From USGS 1:24,000 scale quadrangle Day, Timbered Crater.
(3) Subunit 5C; Shasta County, California. From USGS 1:24,000 scale quadrangle Dana, Burney Falls.
(4) Subunit 5D; Shasta County, California. From USGS 1:24,000 scale quadrangle Burney.
(5) Subunit 5E; Shasta County, California. From USGS 1:24,000 scale quadrangle Burney.
(6) Subunit 5F; Shasta County, California. From USGS 1:24,000 scale quadrangle Merken Bench.
(7) Subunit 5G; Shasta County, California. From USGS 1:24,000 scale quadrangle Murken Bench, Old Station.
(8) Subunit 5H; Lassen County, California. From USGS 1:24,000 scale quadrangle Poison Lake, Swains Hole.
(9) Subunit 5I; Lassen and Shasta County, California. From USGS 1:24,000 scale quadrangle Swains Hole.
(10) Subunit 5J; Lassen County, California. From USGS 1:24,000 scale quadrangle Harvey Mtn., Poison Lake, Pine Creek Valley, Bogard Buttes.
(11) Subunit 5K; Shasta County, California. From USGS 1:24,000 scale quadrangle Old Station, West Prospect Peak.
(12) Subunit 5L; Plumas County, California. From USGS 1:24,000 scale quadrangle Almanor.
(13) Subunit 6A; Shasta County, California. From USGS 1:24,000 scale quadrangle Enterprise.
(14) Subunit 6B; Shasta County, California. From USGS 1:24,000 scale quadrangle Enterprise, Cottonwood.
(15) Subunit 6C; Shasta County, California. From USGS 1:24,000 scale quadrangles Balls Ferry, Cottonwood, Enterprise, and Palo Cedro.
(16) Subunit 6D; Shasta County, California. From USGS 1:24,000 scale quadrangle Palo Cedro, Balls Ferry.
(17) Subunit 6E; Tehama County, California. From USGS 1:24,000 scale quadrangle Henleyville, Corning, West of Gerber, Gerber, Red Bluff West, Red Bluff East.
(18) Subunit 6F; Glenn and Tehama counties, California. From USGS 1:24,000 scale quadrangle Black Butte Dam and Kirkwood.
(19) Subunit 7A; Shasta County, Tehama County, California. From USGS 1:24,000 scale quadrangle Balls Ferry.

(20) Subunit 7B; Shasta and Tehama County, California. From USGS 1:24,000 scale quadrangles Tuscan Buttes NE, Balls Ferry, Shingletown, Dales, Bend, Red Bluff East.
(21) Subunit 7C; Butte County, Tehama County, California. From USGS 1:24,000 scale quadrangles Acorn Hollow, Campbell Mound, Richardson Springs Northwest, and Vina.
(22) Subunit 7D; Butte County, California. From USGS 1:24,000 scale quadrangle Richardson Springs.
(23) Subunit 7E; Butte County, California. From USGS 1:24,000 scale quadrangle Richardson Springs.
(24) Subunit 7F; Butte County, California. From USGS 1:24,000 scale quadrangle Paradise West, Richardson Springs, Chico.
(25) Subunit 7G; Butte County, California. From USGS 1:24,000 scale quadrangle Hamlin Canyon, Chico.
(26) Subunit 7H; Butte County, California. From USGS 1:24,000 scale quadrangle Cherokee, Hamlin Canyon.
(27) Subunit 7I; Butte County, California. From USGS 1:24,000 scale quadrangle Hamlin Canyon, Shipee.
(28) Subunit 7J; Butte County, California. From USGS 1:24,000 scale quadrangle Cherokee, Oroville, Shippee.
(29) Subunit 7K; Butte County, California. From USGS 1:24,000 scale quadrangles Oroville, and Shippee.
(30) Subunit 7L; Butte County, California. From USGS 1:24,000 scale quadrangle Hamlin Canyon, Shippee.
(31) Subunit 7M; Butte County, California. From USGS 1:24,000 scale quadrangle Cherokee, Oroville, Shippee.
(32) Subunit 7N; Butte County, California. From USGS 1:24,000 scale quadrangle Oroville, Shippee.
(33) Subunit 8A; Mendocino County, California. From USGS 1:24,000 scale quadrangle Point Arena.
(34) Subunit 9A; Lake County, California. From USGS 1:24,000 scale quadrangle Kelseyville, The Geysers.
(35) Subunit 9B; Lake County, California. From USGS 1:24,000 scale quadrangle Middletown.
(36) Subunit 9C; Napa County, California. From USGS 1:24,000 scale quadrangle Capell Valley, Yountville.
(37) Subunit 10A; Colusa County, California. From USGS 1:24,000 scale quadrangle Meridian, Colusa.
(38) Subunit 10B; Yolo County, California. From USGS 1:24,000 scale quadrangles Davis, and Saxon.
(39) Subunit 10C; Solano County, California. From USGS 1:24,000 scale quadrangle Dozier.
(40) Subunit 10D; Solano County, California. From USGS 1:24,000 scale quadrangle Elmira.
(41) Subunit 10E; Solano County, California. From USGS 1:24,000 scale quadrangles Denverton, and Elmira.
(42) Subunit 10F; Solano County, California. From USGS 1:24,000 scale quadrangles Denverton, Elmira, and Fairfield South.
(43) Subunit 10G; Solano County, California. From USGS 1:24,000 scale quadrangle Fairfield South.
(44) Subunit 10H; Solano County, California. From USGS 1:24,000 scale quadrangle Fairfield South.

(45) Subunit 11A; Yuba County, California. From USGS 1:24,000 scale quadrangles Browns Valley, and Wheatland.
(46) Subunit 11B; Placer County, California. From USGS 1:24,000 scale quadrangle Lincoln.
(47) Subunit 11C; Placer County, California. From USGS 1:24,000 scale quadrangle Lincoln.
(48) Subunit 11D; Sacramento County, California. From USGS 1:24,000 scale quadrangle Folsom.
(49) Subunit 11E; Sacramento County, California. From USGS 1:24,000 scale quadrangle Carmichael.
(50) Subunit 11F; Sacramento County, California. From USGS 1:24,000 scale quadrangle Sloughhouse.
(51) Subunit 11G; Amador County, Sacramento County, California. From USGS 1:24,000 scale quadrangles Carbondale, Clay, Goose Creek, and Sloughhouse.
(52) Subunit 11H; Sacramento, San Joaquin County, California. From USGS 1:24,000 scale quadrangle Lockeford, Clay.
(53) Subunit 12A; Napa County, California. From USGS 1:24,000 scale quadrangle Napa, Cuttings Wharf.
(54) Subunit 12B; Napa County, California. From USGS 1:24,000 scale quadrangle Cuttings Wharf.
(55) Subunit 12C; Contra Costa County, California. From USGS 1:24,000 scale quadrangle Benicia, Mare Island.
(56) Subunit 13A; Contra Costa County, California. From USGS 1:24,000 scale quadrangle Antioch South, Brentwood.
(57) Subunit 13B; Contra Costa County, California. From USGS 1:24,000 scale quadrangle Byron Hot Springs, Clifton Court Forebay.
(58) Subunit 13C; Contra Costa County, California. From USGS 1:24,000 scale quadrangle Byron Hot Springs.
(59) Subunit 13D; Alameda County, California. From USGS 1:24,000 scale quadrangle Byron Hot Springs.
(60) Subunit 13E; Alameda County, California. From USGS 1:24,000 scale quadrangle Altamont, Livermore.
(61) Subunit 14A; Stanislaus County, California. From USGS 1:24,000 scale quadrangle Ripon.
(62) Subunit 14B; Merced County, California. From USGS 1:24,000 scale quadrangles Gustine, San Luis Ranch, and Stevinson.
(63) Subunit 14C; Merced County, California. From USGS 1:24,000 scale quadrangles San Luis Ranch, and Stevinson.
(64) Subunit 14D; Merced County, California. From USGS 1:24,000 scale quadrangles Arena, San Luis Ranch, Stevinson, and Turner Ranch.
(65) Subunit 14E; Merced County, California. From USGS 1:24,000 scale quadrangles Arena, and Turner Ranch.
(66) Subunit 14F; Merced County, California. From USGS 1:24,000 scale quadrangles Sandy Mush, and Turner Ranch.
(67) Subunit 14G; Merced County, California. From USGS 1:24,000 scale quadrangles Sandy Mush and Turner Ranch.
(68) Subunit 14H; Merced County, California. From USGS 1:24,000 scale quadrangle Sandy Mush.
(69) Subunit 14I; Merced County, California. From USGS 1:24,000 scale quadrangles El Nido, and Sandy Mush.
(70) Subunit 14J; Merced County, California. From USGS 1:24,000 scale quadrangle Sandy Mush.
(71) Subunit 14K; Merced County, California. From USGS 1:24,000 scale quadrangle El Nido.

[(89) omitted]

(90) Subunit 14L; Merced County, California. From USGS 1:24,000 scale quadrangles El Nido, and Plainsburg.
(91) Subunit 14M; Kings County and Tulare County, California. From USGS 1:24,000 scale quadrangles Burris Park, Monson, Remnoy, and Traver.
(92) Subunit 14N; Tulare County, California. From USGS 1:24,000 scale quadrangles Alpaugh, Cocoran, and Taylor Weir.
(93) Subunit 14O; Tulare County, California. From USGS 1:24,000 scale quadrangles Alpaugh, and Pixley.
(94) Subunit 14P; Tulare County, California. From USGS 1:24,000 scale quadrangles Alpaugh, and Pixley.
(95) Subunit 14Q; Tulare County, California. From USGS 1:24,000 scale quadrangle Delano West.
(96) Subunit 15A; San Joaquin County, California. From USGS 1:24,000 scale quadrangle Peters, Farmington, Linden, Valley Springs SW.
(97) Subunit 15B; Tuolumne and Stanislaus County, California. From USGS 1:24,000 scale quadrangle Keystone, Knights Ferry.
(98) Subunit 15C; Stanislaus County, California. From USGS 1:24,000 scale quadrangles Paulsell, and Waterford.
(99) Subunit 15D; Stanislaus County, California. From USGS 1:24,000 scale quadrangle Paulsell.
(100) Subunit 15E; Stanislaus County, Tuolumne County, California. From USGS 1:24,000 scale quadrangles Cooperstown, Keystone, La Grange, and Paulsell.
(101) Subunit 15F; Stanislaus County, California. From USGS 1:24,000 scale quadrangle Paulsell.
(102) Subunit 15G; Stanislaus County, California. From USGS 1:24,000 scale quadrangles Montpelier, and Paulsell.
(103) Subunit 15H; Merced County, Stanislaus County, California. From USGS 1:24,000 scale quadrangles Cooperstown, La Grange, Merced Falls, Montpelier, Paulsell, and Turlock Lake.
(104) Subunit 15I; Merced County, California. From USGS 1:24,000 scale quadrangle Turlock Lake.
(105) Subunit 15J; Madera County, Mariposa County, Merced County, California. From USGS 1:24,000 scale quadrangles Haystack Mountain, Illinois Hill, Indian Gulch, Le Grand, Merced, Merced Falls, Owens Reservoir, Plainsburg, Planada, Raynor Creek, Snelling, Winton, and Yosemite Lake.
(105) Subunit 15J; Madera County, Mariposa County, Merced County, California. From USGS 1:24,000 scale quadrangles Haystack Mountain, Illinois Hill, Indian Gulch, Le Grand, Merced, Merced Falls, Owens Reservoir, Plainsburg, Planada, Raynor Creek, Snelling, Winton, and Yosemite Lake.
(107) Subunit 15L; Fresno County, and Madera County, California. From USGS 1:24,000 scale quadrangles Daulton, Friant, Gregg, Lanes Bridge, Little Table Mountain, and Millerton Lake West.
(108) Subunit 15M; Madera County, California. From USGS 1:24,000 scale quadrangles Millerton Lake East, and North Fork.
(109) Subunit 15N; Fresno County, California. From USGS 1:24,000 scale quadrangles Academy, and Millerton Lake East.
(110) Subunit 15O; Fresno County, California. From USGS 1:24,000 scale quadrangles Academy, Friant, and Round Mountain.
(111) Subunit 15P; Fresno County, California. From USGS 1:24,000 scale quadrangle Clovis.
(112) Subunit 15Q; Fresno County, California. From USGS 1:24,000 scale quadrangle Clovis.
(113) Subunit 15R; Tulare County, California. From USGS 1:24,000 scale quadrangles Ivanhoe, and Stokes Mountain.

(114) Subunit 15S; Tulare County, California. From USGS 1:24,000 scale quadrangles Auckland, Ivanhoe, Stokes Mountain, and Woodlake.
(115) Subunit 15T; Tulare County, California. From USGS 1:24,000 scale quadrangle Woodlake.
(116) Subunit 15U; Tulare County, California. From USGS 1:24,000 scale quadrangle Monson.
(117) Subunit 15V; Tulare County, California. From USGS 1:24,000 scale quadrangle Monson.
(118) Subunit 15W; Tulare County, California. From USGS 1:24,000 scale quadrangle Monson.
(119) Subunit 16B; Alameda County, California. From USGS 1:24,000 scale quadrangle Niles, Milpitas.
(120) Subunit 17A; San Benito, Monterey counties, California. From USGS 1:24,000 scale quadrangle Llanada, San Benito, Hernandez Reservoir, Rock Springs Peak, Topo Valley, Hepsedam Peak, Lonoak, Pinalito Canyon, Monarch Peak, Nattrass Valley.
(121) Subunit 18A; Monterey County, California. From USGS 1:24,000 scale quadrangle Williams Hill, Jolon, Valleton, Bradley, San Miguel, Wunpost.
(122) Subunit 19A; Monterey County, California. From USGS 1:24,000 scale quadrangle Bradley, San Miguel, Wunpost, Valleton.
(123) Subunit 19B; Monterey, San Luis Obispo counties, California. From USGS 1:24,000 scale quadrangle Bradley.
(124) Subunit 19C; Monterey, San Luis Obispo counties, California. From USGS 1:24,000 scale quadrangle San Miguel.
(125) Subunit 19D; San Luis Obispo County, California. From USGS 1:24,000 scale quadrangle San Miguel.
(126) Subunit 19E; San Luis Obispo County, California. From USGS 1:24,000 scale quadrangle Paso Robles, and San Miguel.
(127) Subunit 19F; San Luis Obispo County, California. From USGS 1:24,000 scale quadrangle Paso Robles, Adelaida.
(128) Subunit 19G; Monterey and San Luis Obispo counties, California. From USGS 1:24,000 scale quadrangle Creston, Paso Robles, Estrella, Ranchito Canyon, Cholame Hills.
(129) Subunit 20A; San Luis Obispo, California. From USGS 1:24,000 scale quadrangle Simmler.
(130) Subunit 21A; Santa Barbara County, California. From USGS 1:24,000 scale quadrangle Santa Ynez, Lake Cachuma, Los Olivos, Figueroa Mtn.
(131) Subunit 22A; Ventura County, California. From USGS 1:24,000 scale quadrangles Alamo Mountain, Lion Canyon, Lockwood Valley, San Guillermo, and Topatopa Mountains.

Primary Constituent Elements/Physical or Biological Features

The primary constituent elements of critical habitat for vernal pool tadpole shrimp (*Lepidurus packardi*) are the habitat components that provide:

(i) Topographic features characterized by mounds and swales and depressions within a matrix of surrounding uplands that result in complexes of continuously, or intermittently, flowing surface water in the swales connecting the pools described in paragraph (ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools;
(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water for a minimum of 41 days, in all but the driest years; thereby providing adequate water for incubation, maturation, and reproduction. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands;

(iii) Sources of food, expected to be detritus occurring in the pools, contributed by overland flow from the pools' watershed, or the results of biological processes within the pools themselves, such as single-celled bacteria, algae, and dead organic matter, to provide for feeding; and

(iv) Structure within the pools described in paragraph (ii) of this section, consisting of organic and inorganic materials, such as living and dead plants from plant species adapted to seasonally inundated environments, rocks, and other inorganic debris that may be washed, blown, or otherwise transported into the pools, that provide shelter.

Recovery Plan Information

Recovery Actions

- Protect vernal pool habitat in the largest blocks possible from loss, fragmentation, degradation, and incompatible uses (USFWS 2005).
- Manage, restore, and monitor vernal pool habitat to promote the recovery of listed species and the long-term conservation of the species of concern (USFWS 2005).
- Conduct range-wide status surveys and status reviews for all species addressed in this recovery plan to determine species status and progress toward achieving recovery of listed species and long-term conservation of species of concern (USFWS 2005).
- Conduct research and use results to refine recovery actions and criteria, and guide overall recovery and long-term conservation efforts (USFWS 2005).
- Develop and implement participation programs (USFWS 2005).
- Additional preservation of known extant occurrences is needed to reduce threats and reach recovery goals outlined in the Recovery Plan. Therefore, preservation of Zone 1 and 2 core areas should be pursued. The areas requiring the highest conservation action due to loss of habitat and/or lack of protected areas include the Northwestern Sacramento Valley (where there are limited protected areas, limited restoration possibilities, and rapid urban expansion, particularly in the Redding area); the Northeastern Sacramento Valley (where, despite the presence of some large preserves, there are limited protected areas in much of the region, a high number of sensitive species, and a high urban-conversion rate); the Southeastern Sacramento Valley (where there are limited protected areas and a high urban-conversion rate); the San Joaquin Valley (where greater emphasis on pool conservation is needed in the northeastern and southern portions of the valley); and the Southern Sierra Foothills (where large areas of the region are being urbanized or converted to agriculture without vernal pool resource mitigation). USFWS should work with private landowners for the conservation of vernal pool tadpole shrimp through conservation easements or other methods (USFWS 2007).
- A standardized formal monitoring program should be developed and implemented to collect data in sufficient detail to evaluate species status, and examine changes in population dynamics and community composition. Monitoring should be conducted in areas with known occurrences throughout the range of this species, including revisiting historical survey sites. Many occurrences reported in the CNDDB (2007) have not been visited in more than a decade. An updated status-review of all known occurrences should be completed. In addition, a statewide vernal pool habitat mapping inventory should be implemented to quantify the actual acreage of vernal pools and acres protected (USFWS 2007).
- Research should be conducted on the extant distribution of the vernal pool tadpole shrimp, to better understand why it is absent from seemingly suitable vernal pools between areas that are known to be occupied by this species, and to understand the specifics of pools where this species occurs. Additional research should be conducted at regularly surveyed sites to incorporate research recommendations outlined in the Recovery Plan (USFWS 2007).
- Results from monitoring and research should be included in the management plans for protected sites supporting occurrences of this species. There is a need to develop management indicators for identifying potential problems and assessing ecosystem health as it pertains to vernal pool

crustaceans. Requirements for appropriate management of vernal pool landscapes also must be established. Because of urban encroachment and resulting hydrological changes, conservation efforts should be focused on managing for unseasonable sources of water that infiltrate vernal pool preserves, resulting in changed site hydrology. Improved guidelines and success criteria also should be established for the monitoring of constructed and restored pools (USFWS 2007).

- Presence-absence survey guidelines should be improved. The current methodology is not always effective for documenting the presence of the species with confidence, given the species' adaptations to environmental fluctuations. Surveys, monitoring of conservation areas, and reporting should be standardized so that data can be systematically compared across sites (USFWS 2007).

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Fish

Delta Smelt (*Hypomesus transpacificus*) and its Critical Habitat

Listing Status

The USFWS listed the delta smelt as threatened on March 5, 1993 (USFWS 1993), and designated critical habitat for the species on December 19, 1994 (USFWS 1994). The delta smelt was one of eight fish species addressed in the Recovery Plan for the Sacramento-San Joaquin Delta Native Fishes (USFWS 1996). A 5-year status review of the delta smelt was completed on March 31, 2004 (USFWS 2004). The review concluded that delta smelt remained a threatened species. A subsequent 5-year status review recommended uplisting delta smelt from threatened to endangered (USFWS 2010a). A 12-month finding on a petition to reclassify the delta smelt as an endangered species was completed on April 7, 2010 (USFWS 2010b). After reviewing all available scientific and commercial information, the Service determined that re-classifying the delta smelt from a threatened to an endangered species was warranted but precluded by other higher priority listing actions (Service 2010c). The Service reviews the status and uplisting recommendation for delta smelt during its Candidate Notice of Review (CNOR) process. Each year it has been published, the CNOR has recommended the uplisting from threatened to endangered. Electronic copies of these documents are available at https://ecos.fws.gov/ecp/species/321.

Life History and Habitat

The delta smelt has a fairly simple life history because a large majority of individuals live only one year (Bennett 2005; Moyle *et al.* 2016) and because it is an endemic species (Moyle 2002), comprising only one genetic population (Fisch *et al.* 2011), that completes its full lifecycle in the northern reaches of the San Francisco Bay-Delta (Merz *et al.* 2011). Most spawning occurs from February through May in various places from the Napa River and locations to the east including much of the Sacramento-San Joaquin Delta. Larvae hatch and enter the plankton primarily from March through May, and most individuals have metamorphosed into the juvenile life stage by June or early July. Most of the juvenile fish continue to rear in habitats from Suisun Bay and marsh and locations east principally along the Sacramento River-Cache Slough corridor (recently dubbed the 'North Delta Arc'; Moyle *et al.* 2010). The juvenile fish (or 'sub-adults') begin to develop into maturing adults in the late fall. Thereafter, the population spatial distribution expands with the onset of early winter storms and the first individuals begin to reach sexual maturity by January in some years, but most often in February (Damon *et al.* 2016; Kurobe *et al.* 2016). Delta smelt do not reach sexual maturity until they grow to at least 55 mm in length (~ 2 inches) and 50% of individuals are sexually mature at 60 to 65 mm in length (Rose *et al.* 2013). The spawning microhabitats of the delta smelt are unknown, but based on adult distribution data (Damon *et al.* 2016; Polansky *et al.* 2018) and the evaluation of otolith microchemistry (Hobbs *et al.* 2007; Bush 2017), most delta smelt spawn in freshwater to slightly brackish-water habitats under tidal influence. Most individuals die after spawning, but as is typical for annual fishes, when conditions allow, some individuals can spawn more than once during their single spawning season (Damon *et al.* 2016).

Population Status

The 2021 California Department of Fish and Wildlife (CDFW) Fall Midwater Trawl (FMWT) Index was 0 for the fourth year in a row. The CDFW Spring Kodiak Trawl (SKT) monitors the adult spawning stock of delta smelt and serves as an indication for the relative number and distribution of spawners in the system. The 2021 SKT Relative Abundance Index was 0, the lowest on record.

The CDFW methods generate abundance indices from each survey but each index is on a different numeric scale. This means the index number generated by a given survey only has quantitative meaning relative to other indices generated by the same survey. Further, the CDFW indices lack estimates of uncertainty (variability) which limits interpretation of abundance changes from year to year even within each sampling program. In 2019, the USFWS completed a new delta smelt abundance indexing procedure using data from all four CDFW surveys (FMWT, summer townet, 20-mm, and SKT) (Polansky *et al.* 2019). The USFWS method improves upon the CDFW method because it generates abundance indices in units of numbers of fish, including attempts to correct for different sampling efficiencies among surveys, and the method includes measures of uncertainty. USFWS indices of spawner abundance based on combined January and February SKT sampling are listed with their confidence intervals in Table 1. The estimates show the most recent 20 years of the delta smelt's longer-term decline in numbers of fish as best as they can be approximated with currently available information.

The USFWS' Enhanced Delta Smelt Monitoring (EDSM) program is designed to complete Delta wide surveys at a weekly time scale while SKT does this at a monthly scale, so the USFWS calculated EDSM abundance estimates using all weekly survey data within the January-February time interval (Table 2). For both surveys, data collected from January and February of each year were combined to derive a single abundance estimate. Beginning in 2022, estimates include cultured delta smelt released in the Delta between December 2021 and February 2022 described below. The effects of survey specific sampling times and locations in relation to release times and locations have not been fully evaluated.

In December 2021, the USFWS, along with CDFW, DWR, and Reclamation, began experimentally releasing captively produced delta smelt into the Sacramento-San Joaquin River Delta in an experiment intended to help inform future supplementation of the species in the wild. A total of 5 releases totaling 55,733 brood year 2021 marked (adipose fin clip or Visible Implant Elastomer (VIE) delta smelt from UC Davis' Fish Conservation and Culture Laboratory. The first release of 12,800 delta smelt occurred over December 14 and 15, 2021 in Rio Vista. The second release of 12,800 delta smelt occurred over January 11 and 12, 2022 in Rio Vista. The third release of 6,400 delta smelt occurred on February 3, 2022 in the Sacramento Deep Water Ship Channel. The fourth release of 12,800 delta smelt occurred over February 9 and 10, 2022 in Suisun Marsh. The fifth release of 10,933 delta smelt occurred over February 16 and 17 in the Sacramento Deep Water Ship Channel. A subsample of those marked fish have been recaptured in the Deepwater Shipping Channel, central Delta, south Delta, and Suisun March by EDSM, Chipps Island Trawl, SKT, Bay Study, and in the Central Valley Project salvage facility.

Table 1. Spring Kodiak Trawl (SKT) Survey abundance estimates and related statistics and data summaries. The Year-to-Year Ratio column shows the population growth rate from one year to the next, calculated as the ratio of abundances from consecutive years. *Data from only February was used because SKT sampling did not take place in January.

			95% Confidence Interval		**Total delta smelt caught (total tows) by the SKT survey**		
Year	Abundance Estimate	Standard Error	Lower Bound	Upper Bound	January	February	Year-to-Year Ratio
2002	1,093,244	195,329	760,332	1,523,294	262 (35)	394(39)	NA
2003*	996,055	261,205	581,197	1,597,198	NA (0)	232 (39)	0.91
2004	966,981	262,190	553,729	1,573,002	380 (39)	300 (34)	0.97
2005	715,858	147,190	470,572	1,044,828	220 (39)	218 (40)	0.74
2006	272,327	42,400	198,681	364,438	44 (40)	84 (40)	0.38
2007	449,466	128,731	249,216	749,168	109 (40)	107 (39)	1.65

2008	509,428	188,396	236,859	963,839	132 (40)	36 (39)	1.13
2009	1,166,145	523,856	459,083	2,464,804	579 (40)	61 (42)	2.29
2010	251,863	54,580	161,753	374,582	88 (41)	57 (41)	0.22
2011	461,599	202,547	185,712	962,088	177 (42)	128 (40)	1.83
2012	1,177,201	328,682	662,728	1,939,836	320 (42)	287 (42)	2.55
2013	333,682	89,809	191,886	541,064	100 (41)	125 (41)	0.28
2014	308,972	91,474	167,858	522,884	148 (40)	55 (40)	0.93
2015	213,345	76,639	101,434	397,439	21 (39)	68 (39)	0.69
2016	25,445	9,584	11,661	48,622	7 (40)	6 (39)	0.12
2017	73,331	23,342	38,010	128,459	18 (38)	8 (41)	2.88
2018	26,649	21,397	5,215	82,805	10 (40)	4 (41)	0.36
2019	5,610	4,395	1,138	17,135	1 (40)	1 (39)	0.21
2020	5,213	3,644	1,241	14,710	1 (39)	1 (40)	0.93
2021	0	Not Defined	Not Defined	Not Defined	0 (39)	0 (36)	0
2022	12,679	9,033	2,942	36,250	0 (36)	5 (40)	NA

Table 2. Enhanced Delta Smelt Monitoring (EDSM) Survey abundance estimates with columns as in Table 1.

			95% Confidence Interval		**Total delta smelt caught (total tows) by the EDSM survey**		
Year	Abundance Estimate	Standard Error	Lower Bound	Upper Bound	January	February	Year-to-Year Ratio
2017	85,162	21,362	50,902	134,047	54 (401)	33 (684)	NA
2018	6,821	2,778	2,931	13,614	10 (727)	3 (610)	0.08
2019	4,500	1,075	2,758	6,947	17 (724)	7 (518)	0.66
2020	1,079	544	379	2,448	3 (625)	2 (606)	0.23
2021	267	189	63	760	2 (327)	0 (466)	0.26
2022	4,909	2,232	1,911	10,450	6 (468)	12 (484)	18.39

Critical Habitat

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

The Service's primary objective in designating critical habitat was to identify the key components of delta smelt habitat that support successful completion of the lifecycle, including spawning, larval and juvenile transport, rearing, and adult migration back to spawning sites. Delta smelt are endemic to the Bay-Delta and the vast majority only live one year. Thus, regardless of annual hydrology, the Bay-Delta estuary must provide suitable habitat all year, every year. The primary constituent elements (PCEs) essential to the conservation of the delta smelt are physical habitat, water, river flow, and salinity concentrations required to maintain delta smelt habitat for spawning, larval and juvenile transport, rearing, and adult migration (USFWS 1994).

The Service's primary objective in designating critical habitat was to identify the key components of delta smelt habitat that support successful completion of the lifecycle.

The delta smelt's critical habitat is currently not adequately serving its intended conservation role and function because there are very few locations that consistently provide all the needed habitat attributes for larval and juvenile rearing at the same times and in the same places. The Service's review indicates it is rearing habitat that remains most impacted by ecological changes in the estuary, both before and since the delta smelt's listing under the ESA. Those changes have stemmed from chronic low outflow, changes in the seasonal timing of Delta inflow, and lower flow variability, species invasions and associated changes in how the upper estuary food web functions, declining prey availability, high water temperatures, declining water turbidity, and localized contaminant exposure and accumulation by delta smelt.

Recovery Plan Information

The delta smelt was one of eight fish species addressed in the Recovery Plan for the Sacramento-San Joaquin Delta Native Fishes (USFWS 1996). The USFWS has used the most up-to-date, best available information to outline the recovery needs of delta smelt. Based on available resources, the USFWS proposes that, in order to recover, delta smelt need a substantially more abundant population, an increase in the quantity and quality of habitat, and other needs as further outlined below:

Abundance **-** a substantially more abundant population, which is notably linked to the success of recruitment between life stages. Abundance is affected by entrainment, predation, feeding, competition, demographics, reproductive success, and fish condition and health.

Entrainment and Impingement Risk

- A reduction in entrainment and impingement of adult, juvenile, and larval individuals and their food supply at Central Valley Project and State Water Project pumping facilities, over and above reductions achieved under real-time operations of the 2008 USFWS biological opinion on the Long-Term Operations of the Central Valley Project and State Water Project, to increase the abundance of the spawning adult population and the potential for recruitment of larvae and juveniles into the adult population. This can be done through OMR modified actions to increase protection among life stages.
- A reduction in entrainment and impingement from other water diversion-related structures within delta smelt critical habitat where delta smelt adults, larvae, or juveniles are known or are likely to be impinged or entrained to increase the adult population and the potential for recruitment of juveniles into the adult population.
- A reduction in entrained food supply within delta smelt critical habitat.

Predation

- Increased escape cover (*i.e.*, sufficient habitat to reduce/avoid predation from observed increases in water clarity).
- Reduction in predators in the Bay-Delta ecosystem to increase survival of adults, larvae, and juveniles from an overall increase in relative abundance of predator species system-wide.

Feeding

- Increased copepod production.

Competition

- Reduction in competition and food web alteration from non-native fish and invertebrates.

Demographic/Genetic

- Maintain or increase genetic diversity within the population and Allee effects (*e.g.*, reduced schooling ability, reduced ability to find mates).

Reproductive Success

- Restoration of migratory and spawning cues from reductions in the spawning season window and modification of natural flow regimes.
- Increase the condition of spawning individuals, such as fish size (*e.g.*, weight, length), fat storage, sufficient calorie intake, and lipid energy.
- Improve delta smelt vital rates, including higher growth rates and higher fecundity levels.
- Improve the sex ratio (males to females) with recognition that there is uncertainty associated with this need and therefore is identified as needing additional research and monitoring.

Fish Body Condition/Health

- Improve physical health through a reduction in contaminants exposure and other pollutants (*e.g.,* metals, pesticides, CEC's [endocrine disruptors], etc.) within its habitat to increase survival of adults, larvae and juveniles.

Habitat **-** an increase in the quality and quantity of suitable migratory, spawning, and rearing habitat. Improved habitat quality within the Bay-Delta should enhance delta smelt reproduction and allow for recruitment success necessary to the species to survive. Suitable habitat conditions require habitat diversity, water quality, and flow.

Habitat Diversity

- Increase habitat complexity (*e.g.*, reduction in dead end sloughs) and heterogeneity.
- Increase in the quality and quantity of suitable spawning habitat and substrate (*i.e.*, sandy beaches with sufficient water velocities, available for direct use) due to reductions in sandy beaches system-wide.
- Maintain or increase (*i.e.*, protect, restore, create, or enhance) suitable habitat within designated critical habitat (*i.e.*, with PCEs), further preventing reductions in habitat.

Water Quality

- Improve water quality – suitable water quality constituents within optimal range (*i.e.*, turbidity, DO levels, water temperature, pH, salinity).

Flow

- Improve flow conditions – suitable flow conditions (*i.e.*, velocity, timing, [delta] freshwater outflow, salinity, tidal energy, flow suitable for spawning migration, to trigger movement to spawning areas, and egg incubation). These can be achieved as a result of active or passive management of water and sediment processes in the San Francisco Bay-Delta ecosystem that mimics more natural (*i.e.,* pre-water development) conditions.

Other Needs – Other factors that affect delta smelt include climate change, aquatic invasive macrophytes, harmful cyanobacteria blooms (*Microcystis*), disease, and exposure to in-water work activities.

Climate Change

- Maintain and increase sufficient suitable habitat from threats of ecosystem changes (community and habitat shifts).
- Prevent reductions/shifts in suitable habitat due to sea-level rise and increased droughts and temperatures.
- Maximize delta smelt population resilience in the face of the potential adverse effects of ongoing climate change that are occurring in the Bay-Delta ecosystem.

Aquatic Invasive Macrophytes

- Reduce aquatic invasive macrophytes due to increased predator habitat from changes in water quality as a result of increased water clarity, residence times, and flow reductions.

Harmful Cyanobacteria Blooms (*i.e., Microcystis*)

- Reduce harmful cyanobacteria blooms from increased water residence time/flow reductions and increased anthropogenic nutrient inputs.

Disease

- Reductions in disease to increase survival of adults, larvae, and juveniles.

Risk to Individuals from Exposure to In-water Work Activities (*e.g.*, dredging riprapping, suction dredging, agricultural diversions)

- Reduce sources of harassment, harm, or mortality to delta smelt individuals, habitat loss, and effects to prey density (*i.e.*, modification of food supply).

Supplementation – The very low abundance of delta smelt has increased the urgency toward development of a program for supplementing the wild population of delta smelt (Lessard *et al*. 2018). Studies are currently underway to help develop a program for using cultured delta smelt for supplementation efforts. In order for a supplementation program to be fully successful, fish must be released into an environment that provides ample food, low levels of toxic compounds, and low entrainment losses (USFWS 1996).

Environmental Baseline

The delta smelt and its designated critical habitat only occur within the State of California. Please refer to the information above.

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Lahontan Cutthroat Trout (*Oncorhynchus clarkii henshawi*)

Listing Status

On October 13, 1970, Lahontan cutthroat trout (LCT) were federally listed as endangered under the Endangered Species Conservation Act of 1969 (USFWS 1970). On July 16, 1975, LCT were reclassified as threatened under the ESA in conjunction with a special 4(d) rule to facilitate management by the states and allow state-permitted sport harvest (USFWS 1975). The 4(d) rule for LCT exempts the take of LCT from the section 9 prohibitions of the ESA when such take is in accordance with applicable state law (50 § CFR 17.44(a)(1)). Critical habitat is not designated for LCT.

The USFWS' 5-year Review included a rangewide evaluation of threats to LCT (USFWS 2009). The 5-Year Review identified nonnative species; habitat fragmentation and isolation; small populations; degraded habitat conditions from land use activities such as road management, water management actions, mining, and livestock grazing; and impacts from climate change such as increased temperatures and increasing frequency and severity of drought and fire as the primary threats affecting the species' long-term persistence.

Life History and Habitat

LCT historically occupied large freshwater and alkaline lakes, small mountain streams and lakes, small tributary streams, and major rivers of the Lahontan Basin of northern Nevada, eastern California, and southern Oregon, including the Truckee, Carson, Walker, Susan, Humboldt, Quinn, Summit Lake/Black Rock Desert, and Coyote Lake watersheds (USFWS 1995). LCT evolved in a variety of habitats which resulted in resident, fluvial, and lacustrine life histories (USFWS 1995). Like most salmonids, LCT require relatively clear, cold waters to maintain viable populations. LCT reproduce in the spring and are obligatory stream spawners, sometimes migrating large distances to find adequate spawning areas. Unlike most freshwater fish species, LCT tolerate relatively high alkalinity and total dissolved solid levels found in some lake environments. LCT evolved in the absence of other trout, and they are highly susceptible to hybridization and competition from introduced trout species.

Population Status

Within the estimated historical range of LCT (circa 1800), approximately 68.0% of stream and lake habitat provide occupied and/or potentially suitable habitat for LCT today (LCT Coordinating Committee 2019). The estimated 32.0% loss over time of potentially suitable habitat across the historical range is due to climatic and anthropogenic factors that have resulted in either the complete loss of habitat or increased stream temperatures within habitats at lower elevations. Habitat considered unsuitable for LCT today can possibly be restored and made suitable in the future because the potentially suitable habitat category reflects a snapshot of habitat conditions and can change to provide better habitat for the species (LCT Coordinating Committee 2019).

As of 2019, 72 "self-sustaining" LCT populations exist in about 15.0 percent of the remaining potentially suitable habitat (LCT Coordinating Committee 2019). Approximately 80.0% of the existing populations occur in smaller, isolated habitat fragments and/or have lower abundances (LCT Coordinating Committee 2019). As a result, these isolated populations are not likely resilient in the long-term (LCT Coordinating Committee 2019) and will require some level of active management in perpetuity. The remaining 20.0% of LCT populations were considered resilient in early 2019; however, very recent preliminary data indicates that at least a third of those are directly threatened by hybridization with rainbow trout (*Oncorhynchus mykiss*). There is little evidence that habitat conditions supporting most LCT populations are improving, indicating that habitat degradation and isolation due to current land management practices still actively threaten many of the existing populations. This information, in combination with the additional threats on the landscape, as discussed below, indicates that the status of LCT is not improving rangewide.

Critical Habitat

Critical habitat has not been designated for LCT.

Recovery Plan Information

A Recovery Plan for the Lahontan Cutthroat Trout was completed in 1995 (USFWS 1995). Information in the Recovery Plan was updated in the following document: Updated Goals and Objectives for the Conservation of Lahontan Cutthroat Trout (Updated Goals and Objectives; LCT Coordinating Committee 2019). The Updated Goals and Objectives, in short, divides the range of LCT into 10 Management Units, where focus was placed on conserving the adaptive capacity of the species by ensuring its life-history characteristics and genetic diversity are conserved in the variable geographic and ecological settings in which the subspecies evolved. This can be accomplished by ensuring LCT populations are represented (i.e., conserve genetic and behavioral diversity within a variety of ecological and geographic settings), resilient (i.e., contain enough individuals in larger, more diverse habitat fragments), and redundant (i.e., spread the risk of extirpation due to catastrophic events) within each Management Unit. For more information regarding how the 3 Rs are guiding LCT recovery efforts today, please see the Updated Goals and Objectives (LCT Coordinating Committee 2019).

Environmental Baseline

In the State of California, the Carson, Truckee, and Walker watersheds contain LCT populations within the historical range (USFWS 1995). In the Carson watershed, some occupied waters include: Heenan, Poison, Murray, and Golden Canyon Creeks, East Fork Carson River, and Heenan Lake. Within the Truckee/Tahoe Watershed: Pole and Sagehen Creeks, Upper Truckee River and tributaries, Little Truckee River, Lake Tahoe, Fallen Leaf Lake, Cascade Lake, Donner Lake, Independence Lake (Gerstung 1988), and several small alpine lakes. Within the Walker Basin: By-Day, Mill, Murphy, Slinkard, Silver, and Wolf Creeks.

Lahontan cutthroat trout have been stocked in out of historical range locations to create refuge populations including upper Mokelumne River (Pacific, Marshal Canyon, and Mill Ranch Creek), upper San Joaquin River (Portuguese Creek), upper Stanislaus River (Disaster Creek), and Yuba River (East Fork and Macklin Creeks) (USFWS 2009).

To provide angling opportunities, LCT have been stocked by CDFW at multiple locations primarily in Sierra Nevada streams, lakes and reservoirs, both within historic watersheds and out-of-basin waters. California counties that contain the majority of these stockings include: Alpine, El Dorado, Mono, Nevada, and Sierra. Stocking locations and numbers vary annually dependent on hatchery production with some locations discontinued or not stocked for multiple years (CDFW unpublished stocking data).

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Longfin Smelt (*Spirinchus thaleichthys*), San Francisco Bay-Delta DPS

Listing Status

The Service published a proposed rule to list the San Francisco Bay-Delta DPS of the longfin smelt as endangered on October 7, 2022 (87 FR 60957). A species status assessment was issued in June 2024 (Service 2024), compiling biological information and conditions of the DPS. The Service published a final rule listing the San Francisco Bay-Delta DPS as endangered on July 30, 2024 (89 FR 61029). The Service proposed designation of critical habitat for the San Francisco Bay-Delta DPS of the longfin smelt on January 15, 2025 (90 CFR 3765). Critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Life History and Habitat

Longfin smelt habitat encompasses the San Francisco Bay, Suisun Bay and Marsh, the Delta, and many freshwater tributaries to these water bodies, and it includes the coastal Pacific Ocean out to the Farallon Islands and waters along the beaches to the north and south of the Golden Gate (City of San Francisco 1985 and 1986; Garwood 2017; Young et al. 2024). Thus, the DPS occupies the full freshwater to marine salinity gradient. Longfin smelt are distributed across these habitats depending on age class and season with warm water temperatures influencing all juvenile and adult fish to move into deeper more saline water where it is cooler.

The core habitat for spawning and early life-stage rearing of longfin smelt is estuarine. Tidal flows that enter the estuary from the Pacific Ocean, and inflowing river water, present opposing forces because these two sources of water have inherent differences in density; saltwater is denser than fresh water. The ocean delivers more force during spring tides than neap tides, and the rivers deliver more force when they run high than when they run low. These opposing hydraulic forces interact with the estuary's bathymetry to create extremely variable and complex hydrodynamic mixing of fresh- and saltwater. Where saltwater and freshwater come into contact, vertical and lateral mixing currents are intermittently generated (Stacey *et al*. 2001). During periods of high outflow, salinity can be very different at the water surface than at the bottom due to freshwater flowing over the top of saltier water (MacWilliams *et al*. 2015). This stronger vertical salinity gradient when Delta outflow is elevated creates more surface area for mixing of fresh and brackish water and can result in salty water near the bottom having a net flow landward while overlying fresher water has a net flow toward the ocean (Monismith *et al.* 2002).

Longfin smelt are generally adapted to cold- and cool-water habitats so elements of their life cycle within the Bay-Delta are influenced by seasonal water temperature variation (Jeffries et al. 2016; Yanagitsuru et al. 2021). The spawning-ready adults migrate into low-salinity waters in the fall (increasingly, late fall). Current information indicates the annual spawning run may be comprised of multiple age classes rather than dominated solely by Age-2 fish as was previously believed. Most spawning and larval hatching occurs from December through March. Most if not all adults die after spawning. Longfin smelt spawn in Suisun Bay/Marsh and parts of the Delta every year (Gross et al. 2022). When Delta outflow and Bay tributary flows are high enough, longfin smelt can spawn successfully in the Napa and Petaluma rivers (Gross et al. 2022) and other Bay Area streams like Coyote Creek in South San Francisco Bay (Lewis et at. 2019). The larvae and young juveniles rear primarily in the low-salinity zone with a center of distribution near X2 (Dege and Brown 2004), which means the spatial location of the Age-0 population can be quite variable from year to year (Grimaldo et al. 2020; Gross et al. 2022). Limited information on the habitat affinities of these young fish indicates that density is comparable in nearshore and offshore habitats (Grimaldo et al. 2017). As water warms in the later spring and early summer, the young fish move to cooler, higher salinity habitats and an unknown fraction enter the Pacific Ocean (Young et al. 2024). The fish remain predominantly in these higher salinity habitats until they are ready to spawn.

Population Status

The longfin smelt DPS has been in general decline for many decades due mainly to alterations of the estuary flow regime and food web (see Service 2022b for further details). A population viability analysis of the DPS indicated it had a high risk of quasi-extinction in the near future (2025-2040; Tobias et al. 2023). The Service is currently leading a life cycle modeling effort as part of the CDFW's Longfin Smelt Science Plan. The proposed action is scheduled to run through 2027 and produce improved estimates of abundance, drivers of abundance, and population viability that will inform the next iteration of this consultation.

The Bay-Delta DPS has plausibly been declining for over 50 years and that decline is presently at circa 3–4 orders of magnitude below initial observations. The Fall Midwater Trawl (FMWT) index is the most used metric of longfin smelt recruitment in the scientific literature. It is generally an index of age-0 fish but the adult longfin smelt population is also declining (Rosenfield and Baxter 2007; Nobriga and Rosenfield 2016), which in turn is limiting how many eggs can be produced. The reason the longfin smelt population keeps appearing to have a 'step decline' or 'change in intercept' of its outflow relationship is at least in part because analyses that do not account for the declining abundances of the parental generations are based on an improper population model that ignores the influence of adult egg supply on how many recruits can be produced.

Critical Habitat

The Service proposed designation of critical habitat for the San Francisco Bay-Delta DPS of the longfin smelt. However, critical habitat will not be analyzed for this species until publication of a final rule in the Federal Register.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The entirety of the San Francisco Bay-Delta DPS of the longfin smelt is within California, please refer to the information above regarding the species environmental baseline.

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Tidewater Goby (*Eucyclogobius newberryi*) and its Critical Habitat

Listing Status

The Service listed the tidewater goby as endangered on March 7, 1994 (59 FR 5494) and designated critical habitat for the tidewater goby on February 6, 2013 (78 FR 8745).

Life History and Habitat

The tidewater goby is endemic to California and is one of the only species of fish to live exclusively in brackish water coastal lagoons, estuaries, and marshes in California (Swift et al. 1989, Moyle 2002). Tidewater goby habitat is characterized by fairly still, but not stagnant, brackish water. They can withstand a wide range of habitat conditions and have been documented in waters with salinity levels that range from 0 to 42 parts per thousand (ppt), temperatures ranging from 46 to 77 degrees Fahrenheit and water depths from 10 to 79 inches (Irwin and Soltz 1984, Swift et al. 1989). Tidewater gobies often migrate upstream and are commonly found up to 0.6 mile up from a lagoon or estuary (Service 2005), and have been recorded as far as 3 to 5 miles upstream of tidal areas (Irwin and Soltz 1984).

Population Status

Historically, the tidewater goby occurred in at least 150 California coastal lagoons and estuaries, from Tillas Slough near the Oregon/California border south to Agua Hedionda Lagoon in northern San Diego County (Swift et al. 1989); the southern extent of its distribution has been reduced by several miles after the mouth of Agua Hedionda Lagoon was permanently modified to be open to the ocean and no longer supports tidewater gobies. The species is currently known to occur in 103 localities, although the number of sites fluctuates with climatic conditions and the current status is unknown in 12 localities. Currently, the most stable populations are in lagoons and estuaries of intermediate size (5 to 124 acres) that are relatively unaffected by human activities (Service 2005).

The decline of the tidewater goby is attributed primarily to habitat loss or degradation resulting from urban, agricultural, and industrial development in and around coastal wetlands, lagoons, and estuaries (Irwin and Soltz 1984). High flows naturally and periodically breach lagoon barriers and expose tidewater gobies to tidal conditions, but artificial breaching has been observed to cause tidewater goby stranding and mortality (Swift et al. 2018). The tidewater goby remains listed as endangered and its overall population and range is currently stable, but still faces ongoing and likely increasing threats of urbanization, artificial breaching, stochastic environmental conditions, and introduced predators. The southernmost population of tidewater goby remains critically endangered because this species has become extirpated from 5 of the 13 historical localities, 4 of which cannot be restored.

Critical Habitat

Approximately 12,156 acres fall within the boundaries of the 65 critical habitat units designated by the 2013 final revised critical habitat rule. Revised critical habitat for the tidewater goby now occurs in 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. Overall, the critical habitat for this species has remained stable but is still threatened by coastal development.

PBF 1: Persistent, shallow (in the range of approximately 0.3 to 6.6 feet), still-to-slow-moving water in lagoons, estuaries, and coastal streams with salinity up to 12 ppt, which provide adequate space for normal behavior and individual and population growth that contain one or more of the following:

- PBF 1a: Substrates (e.g., sand, silt, mud) suitable for the construction of burrows for reproduction;

- PBF 1b: Submerged and emergent aquatic vegetation, such as *Potamogeton pectinatus*, *Ruppia maritima*, *Typha latifola*, and *Scirpus* spp., that provides protection from predators and high flow events; or
- PBF 1c: Presence of a sandbar(s) across the mouth of a lagoon or estuary during the late spring, summer, and fall that closes or partially closes the lagoon or estuary, thereby providing relatively stable water levels and salinity.

Recovery Plan Information

The goal of the tidewater goby recovery plan (Service 2005) is to conserve and recover the tidewater goby throughout its range by managing threats and maintaining viable metapopulations within each recovery unit while retaining morphological and genetic adaptations to regional and local environmental conditions. The recovery plan identifies six recovery units: North Coast Unit, Greater Bay Unit, Central Coast Unit, Conception Unit, Los Angeles/Ventura Unit, and South Coast Unit.

Environmental Baseline

The species only occurs within the State of California, please refer to the above information regarding the species environmental baseline.

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Unarmored Threespine Stickleback (*Gasterosteus aculeatus williamsoni*)

Listing Status

The Service listed the unarmored threespine stickleback as endangered on October 13, 1970 (35 FR 16047). Channelization and other habitat modifications result in the destruction and degradation of unarmored threespine stickleback habitat. Rivers and streams that once supported unarmored threespine sticklebacks have been either severely altered or reduced for the most part to concrete-lined drains. Stream channelization can diminish the side channels and backwater pool habitat used by unarmored threespine stickleback, and by scouring of stream channels, which may eliminate or reduce the substrate needed for nests (Baskin 1974, p. 58).

Life History and Habitat

The unarmored threespine stickleback is a small (up to 2.36 inches), scaleless, freshwater fish inhabiting slow moving reaches, or quiet water microhabitats of streams and rivers. Favorable habitats for the unarmored threespine stickleback are usually shaded by dense and abundant vegetation. Unarmored threespine sticklebacks feed primarily on benthic insects, small crustaceans, and snails, and to a lesser degree, on flat worms, nematodes, and terrestrial insects. They reproduce throughout the year, but breeding activity is reduced from October to January. Reproduction occurs in areas with adequate aquatic vegetation and gentle flow of water where males establishand vigorously defend territories (Moyle 2002, p. 342; Swift 1999, p. 22).

Population Status

Unarmored threespine sticklebacks were historically distributed throughout southern California, including low-gradient portions of the Los Angeles, San Gabriel, and Santa Ana Rivers, and from a few localities in Santa Barbara County. At the time of listing in 1970, however, they wereonly known to occur in the upper reaches of the Santa Clara River, including Soledad Canyon (Baskin 1974, pp. 3, 7). Current extant populations are restricted to the upper Santa Clara River and its tributaries in Los Angeles County, San Antonio Creek on Vandenberg Air Force Base in Santa Barbara County, Shay Creek (tributary to Baldwin Lake) in San Bernardino County, and San Felipe Creek in San Diego County.

The unarmored threespine stickleback faces a series of threats that include channelization and other habitat modifications associated with urbanization, agricultural practices, and recreation; agricultural, industrial, and municipal water pollution; stream flow alterations caused by water diversion and ground water pumping; the introduction of competing and predatory species; and hybridization with partially armored threespine stickleback.

At the time of listing, there was no abundance data for the unarmored threespine stickleback. Even now, no rangewide, long-term monitoring program is currently being conducted for the subspecies, and data on population dynamics are limited. Despite the availability of survey methods that can estimate constant variability in local abundance (i.e., annual and seasonal changes in distribution and abundance hamper efforts to estimate population size for this short-lived subspecies), estimates of population size are generally lacking due to minimal survey efforts. Unarmored threespine stickleback populations also vary with between-year changes in environmental conditions, such as drought. While unarmored threespine sticklebacks may be seasonally abundant in most years, the subspecies' restricted distribution renders it vulnerable to catastrophic extirpation.

Recovery Plan Information

The Service first issueda recovery plan for the unarmored threespine stickleback in 1977 (Service 1977), which was revised in December 1985 (Service 1985). The revised recovery plan for the unarmored threespine stickleback designated three areas as very important for the survival and recovery of the subspecies: (1) two disjunct reaches of the Santa Clara River in Los Angeles County; (2) a short reach of San Francisquito Canyon; and (3) the lowermost 8.4 miles in San Antonio Creek in Santa Barbara County (Service 1985). The recovery plan states that the subspecies could be considered recovered when: (1) habitat conditions for each of the known remnant populations have been stabilized at or near historical carrying capacities; (2) the other known threats have been addressed in a manner that assures the continued existence of these populations; and (3) at least five self-sustaining populations have been maintained within the historical range of unarmored threespine stickleback for a period of 5 consecutive years without significant threats to their continued existence. The recovery strategy for the unarmored threespine stickleback, as defined in the recovery plan, includes the following actions: (1) close regulation of removal (take) of the subspecies; (2) monitoring and appropriate management of habitat conditions; (3) implementation of contingency plans to protect the subspecies from natural or man-made disasters; and (4) establishment of additional populations in suitable reintroduction sites as needed.

Environmental Baseline

The species only occurs within the State of California, please refer to information above regarding the species environmental baseline.

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Vernal Pool Plants

Butte County Meadowfoam (*Limnanthes floccosa* ssp. *californica*) and its Critical Habitat

Listing Status

The Butte County meadowfoam (*Limnanthes floccosa* ssp. *californica*) was listed as endangered on June 8, 1992 (57 FR 24192). Critical habitat was designated for the Butte County meadowfoam on February 10, 2006 (71 FR 7118).

Life History and Habitat

The Butte County meadowfoam inhabits valley and foothill grasslands (mesic soils). It grows in three types of seasonal wetlands: ephemeral drainages, vernal pool depressions in ephemeral drainages, and occasionally around the edges of isolated vernal pools (57 FR 24192, NatureServe 2015). This species occurs on alluvial terraces in annual grasslands with mima mound topography. The occurrences are found at 165 to 1,167 feet in elevation (CNDDB 2007). *Limnanthes floccosa* ssp. *californica* occurs in different soils on Tuscan-Igo-Anita Complex Fan terraces of 0-3 percent slope, 0-50 percent rock cobble with an underlying clay durapan. According to the 2006 Butte Area Soil Survey, L *Limnanthes floccosa* ssp. *californica* is found on 32 different "Musym" classes of soil, but always with an underlying durapan, rock cobble and common hydrological factors. *Limnanthes floccosa* ssp. *californica* has also been found occasionally in disturbed areas, such as drainage ditches, firebreaks, and graded sites (USFWS 2008).

This is an annual plant. *Limnanthes floccosa* ssp. *californica* typically begins flowering in February, reaches peak flowering in March, and may continue into April if conditions are suitable. Nutlets are produced in March and April, and the plants die back by early May. *Limnanthes floccosa* ssp. *californica* has floral adaptations that allow for cross-pollination by insects, but self-pollination mechanisms take over to ensure seed set if insect pollination is unsuccessful. The particular pollinators of *Limnanthes floccosa* ssp. *californica* have not been identified; however, other meadowfoam species are pollinated by the native burrowing bees *Andrena limnanthis* and *Panurginus occidentalis* and by honeybees, beetles, flies, true bugs (order Hemiptera), butterflies, and moths (USFWS 2008).

Nutlets of *Limnanthes floccosa* ssp. *californica* are apparently dispersed by water and can remain afloat for up to 3 days. Most meadowfoam nutlets are dispersed only short distances. Birds and livestock are potential sources of long-distance seed dispersal, but specific instances of such dispersal have not been documented (USFWS 2008).

Population Status

Rangewide Status of the Species

This species is endemic to California, only known from Butte County. Known historically and currently to occur only in Butte County within the Northeast Sacramento Valley Vernal Pool Region (USFWS 2008). At least eight new occurrences of *Limnanthes floccosa* ssp. *californica* have been discovered since 1988 (USFWS 2005).

Population Summary

When listed, there were 18 known extant occurrences of this subspecies (57 FR 24192). In 1989, less than 200,000 plants likely existed in the censused populations (57 FR 24192, NatureServe 2015). Quantitative information on the numbers of plants and area occupied by *Limnanthes floccosa* ssp. *californica* has not been collected in a consistent and systematic manner at all occurrences since the time of listing; therefore, definitive range-wide abundance and population trend information is not yet available (USFWS 2008).

Some surveys have been conducted on individual locations with varying results. Surveys conducted in 2004 for *Limnanthes floccosa* ssp. *californica* indicate that some of the locations may be decreasing in numbers of plants. However, at least one occurrence, Rancho Arroyo (also known as Foothill Park East Preserve), was reported to have increased in area and in number of plants beginning in approximately 2005. Surveys conducted at Tuscan Preserve and Doe Mill Preserve over 15 years showed that numbers of plants fluctuated annually, reflecting the weather conditions (USFWS 2008).

Threats

Threats to this species include:

- 11 occurrences are located on privately owned land and are unprotected. Habitat loss or degradation from urbanization continues to be the greatest threat to all occurrences of the subspecies, even to those that are protected from development (USFWS 2008).
- The Draft Land Management Plan for the Doe Mill Preserve noted that the occurrence of *Limnanthes floccosa* ssp. *californica* was "healthy" in 1991 but was reduced in numbers in 1996 and stressed from competition with the nonnative grass, *Taeniatherum caput-medusae* (medusa-head). *Glyceria declinata* (waxy manna grass) is a nonnative, perennial grass which may become a threat to *Limnanthes floccosa* ssp. *californica*. *Glyceria declinata* forms dense stands and is able to invade vernal pool habitat and displace native plants (USFWS 2008).
- Maintenance of the natural hydrology of these wetlands is necessary for the survival and recovery of this subspecies. Drought or flood conditions will place additional strains on the vernal pool ecosystems supporting *Limnanthes floccosa* ssp. *californica* occurrences. Climate change is also a stressor (USFWS 2008).
- Impacts from off-road vehicles continue to threaten to the subspecies (USFWS 2008).

Five-Year Status Review

On July 10, 2008, the USFWS issued a five-year status review of the Butte County meadowfoam, which resulted in no change in listing status (USFWS 2008). On September 26, 2023, the U.S. Fish and Wildlife Service completed a five-year status review of Butte County meadowfoam and concluded that this species' endangered status would remain unchanged (USFWS 2022).

Critical Habitat

Critical habitat was designated for the Butte County meadowfoam on February 10, 2006 (71 FR 7118). Critical habitat units are depicted for Tehama and Butte counties, California. Critical habitat is designated in four units totaling 16,636 acres.

The primary constituent elements of critical habitat for the Butte County meadowfoam (*Limnanthes floccosa* ssp. *californica*) are the habitat components that provide (71 FR 7118):

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described in paragraph (ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools; and

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the Butte County meadowfoam (USFWS 2005).

Recovery Actions

- Establish a range-wide recovery implementation team (USFWS 2005).
- Establish working groups and develop participation plans for each vernal pool region (USFWS 2005).
- Develop and implement adaptive management plans based on monitoring data and best available science (USFWS 2005).
- Assist local governments in developing habitat conservation plans and developing land use protection measures (USFWS 2005).
- Assist private landowners in developing landowner agreements (USFWS 2005).
- Acquire habitat, where necessary (USFWS 2005).
- Track losses and protection of suitable habitat and occurrences within core areas (USFWS 2005).
- Ensure mechanisms are in place to provide for the perpetual management and monitoring of core areas, vernal pool regions, or for each management unit within a vernal pool region, as appropriate (USFWS 2005).

Environmental Baseline

The Butte County meadowfoam and its designated critical habitat only occur in Butte and Tehama counties, California. Please refer to information above for the environmental baseline.

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NatureServe. 2015. NatureServe Explorer, An online encyclopedia of life [web application]. Available online at: http://explorer.natureserve.org/.

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71 FR 7118. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for Four Vernal Pool Crustaceans and Eleven Vernal Pool Plants. Final Rule, Administrative Revisions. Vol. 71, No. 28. Federal Register 7118. February 10, 2006. Available online at: https://www.govinfo.gov/content/pkg/FR-2006-02-10/pdf/06-1080.pdf#page=2

California Orcutt Grass (*Orcuttia californica*)

Listing Status

California Orcutt grass was federally listed as endangered on August 3, 1993 due to habitat loss and degradation from urban and agricultural development, livestock grazing, off-road vehicle use, trampling, invasion from weedy non-native plants, and other factors (58 FR 41384).

Life History and Habitat

California Orcutt grass is a tufted annual grass, 2 to 8 inches tall. Its seeds germinate in the saturated and/or submerged soil of vernal pools, and plants are at first nearly prostrate. The plants produce more erect glandular pubescent stems when they are exposed as the pool dries up and subsequently produce flowers and seeds. California Orcutt grass seeds germinate while the pool is inundated, and plants appear prostrate during this phase of their life history. The plant's stems become more erect as the ephemeral pool dries out by evaporation, at which time the plants flower, usually between April and June, and set seed. It is doubtful that any significant amount of germination occurs in the absence of the pool being inundated. Like most grasses, its flowers are wind pollinated; however, it relies on fungi to play a role in stimulating germination (Service 2011).

Population Status

At the time of listing, California Orcutt grass was thought to be restricted to four general localities in California, located in Riverside and San Diego counties. These localities were the Santa Rosa Plateau, Skunk Hollow, and Salt Creek (now identified as the Stowe Pools) in Riverside County, and Otay Mesa in San Diego County. At the time, it was thought to be extirpated from Los Angeles County (Service 2011).

California Orcutt grass is currently considered to be extant at 28 occurrences: three occurrences in Ventura County, three occurrences in Los Angeles County, nine occurrences in Riverside County, and 13 occurrences in San Diego County. Since listing, California Orcutt grass was rediscovered at two occurrences in Los Angeles County and detected for the first time at three occurrences in Ventura County. These occurrences extend the range of the species by about 87 miles to the northwest. California Orcutt grass is still considered to be extant at the Santa Rosa Plateau, Skunk Hollow, and Upper Salt Creek (Stowe Pools) in Riverside County. Since listing, four previously unknown occurrences of the species have been found in Riverside County, and at least nine previously unknown occurrences have been found in San Diego County. In Baja California, Mexico, California Orcutt grass had been found historically on Mesa de Colonet and at San Quintin; however, there is no current knowledge confirming the contemporary existence of the species in this area (Service 2011).

All remaining California Orcutt grass habitat is threatened, to varying degrees, by many of the original threats. However, trampling associated with immigrant travel, military activities, and mowing and plowing of extant habitat have nearly been eliminated as threats. All other delineated threats remain, including rangewide threats associated with small population size and climate change, and may disrupt the presence and population dynamics of the species. Twelve occurrences face threats to the habitat from urban or agricultural development and off-highway vehicle traffic. Grazing remains as a threat to four of the occurrences, and nonnative plants threaten five occurrences. Outside of continued urbanization and direct/indirect effects associated with this threat, climate change may have the longest lasting potential for degrading the species long-term persistence, setting back potential recovery, or causing extinction. Protections afforded by the Act and corresponding cooperative endeavors with private landowners, universities, and local and State governments, have reduced or ameliorated several of these threats since listing. As a result, conservation efforts afford protection to 11 of the 28 (39 percent) extant occurrences of California Orcutt grass from direct habitat loss due to development (Service 2011).

Recovery Plan Information

A recovery plan for California Orcutt grass and other vernal pool species was released on September 3, 1998 (Service 1998) and a clarification to this plan was released on October 1, 2019 (Service 2019). The delisting criteria include the following:

1) All 74 geographic areas and associated vernal pool complexes as identified in Appendices F and G of the 1998 Recovery Plan under each of the specific management areas are protected and managed to ensure long-term viability.
2) The Service must determine that the following factors are no longer present, or continue to adversely affect, California Orcutt grass: (a) the present or threatened destruction, modification, or curtailment of their habitat range; (b) over utilization for commercial, recreational, scientific, or educational purposes; (c) disease or predation; (d) the inadequacy of existing regulatory mechanisms; and (e) other natural and manmade factors affecting their continued existence.
3) Population trends continue to be stable or increasing for 10 consecutive years after threats have been sufficiently ameliorated or managed completion of delisting criterion 2 prior to consideration for delisting.

None of the criteria in the recovery plan have been completely met at this time, and many threats continue to impact the species. A better estimate of the population size in each pool complex is still needed to ensure the long-term persistence of the species. In addition, population trends also need to be monitored and must be stable or increasing for a minimum of 10 years prior to reclassification (Service 2011).

Environmental Baseline

Since California Orcutt grass occurs mostly within California, except for two potential locations in Mexico for which there is limited information available, the status description above also serves as the baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 1998. Vernal pools of southern California recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon. 113+pp.

Service (U.S. Fish and Wildlife Service). 2011. (*Orcuttia californica*) California Orcutt grass 5-year review: summary and evaluation. 44 pp.

Service (U.S. Fish and Wildlife Service). 2019. Recovery plan clarification for the vernal pools of southern California. Department of the Interior. 2 pp.

Contra Costa Goldfields (*Lasthenia conjugens*) and its Critical Habitat

Listing Status

The Contra Costa goldfields (*Lasthenia conjugens*) was listed as endangered on June 18, 1997 (62 FR 33029). Critical habitat was designated for the Butte County meadowfoam on February 10, 2006 (71 FR 7118).

Life History and Habitat

The Contra Costa goldfields inhabit vernal pools in open grassy areas at elevations up to 470 m (NatureServe 2015). *Lasthenia conjugens* typically grows in vernal pools, swales, and low depressions in open valley and foothill grasslands and have been found in three types of vernal pools: Northern Basalt Flow, Northern Claypan, and Northern Volcanic Ashflow (Sawyer and Keeler-Wolf 1995). This species is commonly found at elevations less than 61 meters (m) (200 feet (ft)) but has been documented at 445 m (1465ft) in Napa County and at 137 m (450ft) in Monterey County (USFWS 2013).

Seed dispersal mechanisms in *Lasthenia conjugens* are unknown. However, the lack of a pappus or even hairs on the achenes makes wind dispersal unlikely (USFWS 2005).

Lasthenia conjugens flowers from March to June and is self- incompatible (USFWS 2013). Although *L. conjugens* has not been the subject of pollinator studies, observations suggest that the same insects visit all outcrossed species of *Lasthenia*, rather than concentrating on any particular species. Insect visitors to flowers of Lasthenia belong to five orders: Coleoptera, Diptera, Hemiptera (true bugs), Hymenoptera, and Lepidoptera. Most of these insects are generalist pollinators. All of the specialist pollinators of *Lasthenia* are solitary bees (family Andrenidae); these pollinators include two species in the subgenus Diandrena (*Andrena submoesta* and *A. puthua*) and five or six species in the subgenus Hesperandrena (*Andrena baeriae, A. duboisi, A. lativentris*, and two or three undescribed species) (USFWS 2005).

Population Status

<u>Rangewide Status of the Species</u>

Historically, *Lasthenia conjugens* occurred in seven vernal pool regions: Central Coast, Lake-Napa, Livermore, Mendocino, Santa Barbara, Santa Rosa, and Solano-Colusa. In addition, several historical occurrences in Contra Costa County are outside of the defined vernal pool regions. Ornduff (1966) reported collections from 13 sites in Alameda, Contra Costa, Mendocino, Napa, Santa Barbara, Santa Clara and Solano counties. Although he cited three specimens each from Contra Costa and Santa Barbara counties, Ornduff (1966; 1979) noted that the species was most common in Solano County. One additional site in Alameda County was documented in 1959 by G. Thomas Robbins, who collected a specimen (# 3963, housed at the Jepson Herbarium) on the “shore of the San Francisco Bay” south of Russell (USFWS 2005; USFWS 2013).

Lasthenia conjugens has been reported in ten counties within California: Alameda, Contra Costa, Marin, Mendocino, Monterey, Napa, Santa Barbara, Santa Clara, Solano, and Sonoma (USFWS 2013).

<u>Population Summary</u>

Of the 23 presumed extant records, four occurrences may now be extirpated: (1) an occurrence in Mendocino County has not been observed since 1937; (2) an occurrence in Alameda County has not been observed since 1959; (3) in 1987, a single plant was observed in Napa County and has not been documented since; (4) an occurrence in Solano County was noted on a field checklist in1996 and the location is unknown. Ramp Neale et al. (2008) found high levels of genetic diversity and moderate levels of differentiation among populations (USFWS 2013).

Threats

Threats to this species include:

- One of the primary threats to *L. conjugens* is conversion of land use, for example residential and industrial development, wetland drainage, and agricultural land conversion (including vineyards) (USFWS 2008). Since 65% of this species occurs on private land and is not protected, this is an ongoing problem (USFWS 2008).
- Inadequacy of existing regulatory mechanisms (USFWS 2013).
- Competition from invasive plant species poses a primary threat to this species. Non-native grasses occur commonly in vernal pool complexes and have become a threat to native vernal pool species through their capacity to change pool hydrology. Non-native grasses maintain dominance at pool edges, sequestering light and soil moisture. *Lolium multiflorum* and *Glyceria declinata* (waxy mannagrass) increase thatch buildup, which leads to increased oxygen depletion in the pools and contributes to the shortening of inundation periods through increased evapotranspiration. As vernal pool complexes become surrounded by residential development and disturbed habitat, the likelihood of invasion by nonnative plants increases (USFWS 2013).
- Both lack of grazing and excessive grazing may cause an increase in organic matter in the habitat that can eliminate the natural vernal pool invertebrate community and promote opportunistic and invasive nonnative species, such as Lolium spp., that outcompete the obligate vernal pool species. The cessation of cattle grazing has been found to exacerbate the negative effects of invasive non-native plants on vernal pool inundation period. Appropriate levels of grazing may help maintain soil conditions and limit the amount of thatch accumulation near vernal pools. Increased grass cover in and around ungrazed pools may lead to an increase in evapotranspiration rates, resulting in a decreased hydroperiod. In areas where long-term grazing has been in effect, moderate grazing (in both stocking numbers and amount of time) may be an important tool in combating non-native plant species, when burning is not an option. Moderate grazing may be a necessary tool to maintain the species diversity of the natural vernal pool ecosystem (USFWS 2013).
- Climate change is another threat to this species.

Five-Year Status Review

There have been three five-year status reviews for this species: one on September 30, 2008, one on February 21, 2013, and one on February 27, 2024. The latest five-year status review concluded that Contra Costa goldfields to meet the definition of endangered and would remain an endangered species (USFWS 2024).

Critical Habitat

The critical habitat designation for *Lasthenia conjugens* includes eight units in Alameda, Contra Costa, Mendocino, Napa, and Solano counties, California. This species critical habitat encompasses approximately 14,730 acres (71 FR 7118).

- Unit 1: Mendocino County, California. From USGS 1:24,000 scale quadrangle Point Arena.
- Unit 2: Napa County, California. From USGS 1:24,000 scale quadrangles Yountville, Capell Valley. Unit 3: Napa County, California. From USGS 1:24,000 scale quadrangles Napa, Cuttings Wharf.
- Unit 4: Solano County, California. (i) Unit 4A: Solano County, California. From USGS 1:24,000 scale quadrangle Fairfield South. (ii) Unit 4B: Solano County, California. From USGS 1:24,000 scale quadrangles Fairfield South. (iii) Unit 4C: Solano County, California. From USGS 1:24,000 scale quadrangles Elmira, Denverton.

- Unit 5: Solano County, California. (i) Unit 5A: Solano County, California. From USGS 1:24,000 scale quadrangle Elmira. (ii) Unit 5B: Solano County, California. From USGS 1:24,000 scale quadrangles Elmira, Denverton.
- Unit 6: Contra Costa County, California. From USGS 1:24,000 scale quadrangle Benicia.
- Unit 7: Contra Costa County, California. From USGS 1:24,000 scale quadrangles Byron Hot Springs, Clifton Court Forebay.
- Unit 8: Alameda County, California. (i) Unit 8A: Alameda County, California. (ii) Unit 8B: Alameda County, California. From USGS 1:24,000 scale quadrangles Milpitas, Niles.

The primary constituent elements of critical habitat for the Contra Costa goldfields (*Lasthenia conjugens*) are the habitat components that provide (71 FR 7118):

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described below in paragraph (ii), providing for dispersal and promoting hydroperiods of adequate length in the pools;

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the Contra Costa goldfields (USFWS 2005).

Recovery Actions

- Protect vernal pool habitat in the largest blocks possible from loss, fragmentation, degradation, and incompatible uses (USFWS 2005).
- Manage, restore, and monitor vernal pool habitat to promote the recovery of listed species and the long-term conservation of the species of concern (USFWS 2005).
- Conduct range-wide status surveys and status reviews for all species addressed in this recovery plan to determine species status and progress toward achieving recovery of listed species and long-term conservation of species of concern (USFWS 2005).
- Conduct research and use results to refine recovery actions and criteria, and guide overall recovery and long-term conservation efforts (USFWS 2005).
- Develop and implement participation programs (USFWS 2005).

Environmental Baseline

The Contra Costa goldfields only occurs in ten counties within California, and its designated critical habitat in Alameda, Contra Costa, Mendocino, Napa, and Solano counties, California. Please refer to information above for the environmental baseline.

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USFWS (U.S. Fish and Wildlife Service). 2005. Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon. Portland, Oregon. xxvi + 606 pages. December 15.

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Few-flowered Navarretia (*Navarretia leucocephala* ssp. *pauciflora* (= *N. pauciflora*))

Listing Status

The few-flowered navarretia (*Navarretia leucocephala* ssp. *pauciflora* (= *N. pauciflora*)) was listed as endangered on June 18, 1997 (62 FR 33029). No critical habitat has been designated for the few-flowered navarretia.

Life History and Habitat

The few-flowered navarretia is extremely rare. This species is dependent on vernal pools for survival and its life history is closely linked to the hydrology of these wetlands. This species is found only on vernal pools on substrates of volcanic origin, specifically in Northern Basalt Flow and Northern Volcanic Ashflow Vernal Pools. Extant localities in Lake County are in "flats" of recent alluvium in mountainous areas; site specific details are not known for Napa County sites (USFWS 2008).

The few-flowered navarretia inhabits vernal pools with a volcanic ash substrate in chaparral, grassland, or mixed coniferous forest communities (NatureServe 2015).

Population Status

Rangewide Status of the Species

The few-flowered navarretia is found in Lake and Napa counties, in the Lake-Napa Vernal Pool Region (USFWS 2008).

Population Summary

All occurrences are within an approximately 20-square mile area. The CNDDB reports eight known occurrences of this species; six in Lake County and two in Napa County (USFWS 2008). However, it is difficult to determine the actual number of localities because of some plants exhibit characteristics that are intermediate between the few-flowered navarretia and many-flowered navarretia (*Navarretia leucocephala* ssp. *plieantha*) because some occurrences historically reported have very vague location descriptions and these locations may represent known sites by different names (USFWS 2008).

Only 1-5 populations of the few-flowered navarretia are known for a total of 1000-2500 individuals of this species (NatureServe 2015).

Threats

Threats to this species include:

- Threats to the habitat of few-flowered navarretia include alteration of hydrology, effects from road maintenance activities, agriculture land conversion, construction of a stock pond, off-road vehicle use, inappropriate grazing regimes, and competition from invasive weedy plant species (USFWS 2008).
- Competition from invasive plant species continues to pose a threat to this species. The localities at Hesse Flat and Manning Flat have been reported to be threatened by invasive plant species such as yellow star thistle (*Centaurea solstitialis*). Although specific information regarding adverse effects from invasive plant species is not available for all sites, it is likely that many of the localities of few-flowered navarretia are currently threatened by invasive plants to some degree. Further research and monitoring are necessary to determine the degree that this species is threatened by non-native invasive plant species (USFWS 2008).
- The small number of localities makes it difficult for this species to persist while sustaining the impacts from competition from nonnative plant species, intensive grazing, changes in hydrology, adjacent development, drought, or other unknown factors. Such populations may be highly

susceptible to extirpation due to chance environmental disturbances. If a locality of few-flowered navarretia has several consecutive years of poor rainfall, intensive grazing, changes in hydrology from adjacent development, or intense competition from other plant species, it is possible that the locality will become extirpated. Populations that decline to zero may not always be capable of rebounding from the soil seed bank and the population is likely to become extirpated (USFWS 2008).

- Climate change is another threat to this species.

Five-Year Status Review

On July 10, 2008, the USFWS issued a five-year status review of the few-flowered navarretia, which resulted in no change in listing status (USFWS 2008). On August 31, 2023, the U.S. Fish and Wildlife Service completed another five-year status review of the few-flowered navarretia, and concluded that this species' endangered status would remain unchanged (USFWS 2023).

Critical Habitat

No critical habitat has been designated for the few-flowered navarretia.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the few-flowered Navarretia (USFWS 2005).

Environmental Baseline

The few-flowered Navarretia only occurs in Lake and Napa counties, California. Please refer to information above for the environmental baseline.

Literature Cited

NatureServe. 2015. NatureServe Explorer, An online encyclopedia of life [web application]. Available online at: http://explorer.natureserve.org/.

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USFWS (U.S. Fish and Wildlife Service). 2023. Few-flowered navarretia (*Navarretia leucocephala* ssp. *pauciflora*) 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service Sacramento Fish and Wildlife Office Sacramento, California. August 31.

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Fleshy Owl's-clover (*Castilleja campestris* ssp. *succulenta*) and its Critical Habitat

Listing Status

The fleshy owl's-clover (*Castilleja campestris* ssp. *succulenta*) was listed as threatened on March 26, 1997 (62 FR 14338). Critical habitat was designated for the fleshy owl's-clover on February 10, 2006 (71 FR 7118).

Life History and Habitat

Castilleja campestris ssp. *succulenta* is found primarily in vernal pools, and only in the lower rolling foothill areas of the eastern San Joaquin Valley in the Southern Sierra Foothills Vernal Pool Region (USFWS 2005). Soil textures at those sites range from extremely stony loam to loamy clay. At the UC Merced site and the surrounding community planning area, 81.4% of the individual pools where this taxon was found were on Redding gravelly loam, 9.5% were on Corning gravelly sandy loam, 6.4% were on Corning gravelly loam, 1.7% were on Keyes gravelly loam, 0.7% was on Keyes gravelly clay loam, and 0.3% was on Pentz loam (USFWS 2011). Self-pollinating species of Castilleja typically occur as widely scattered individuals, rather than dense colonies (USFWS 2011). Populations of *Castilleja campestris* spp. *succulenta* have been reported from elevations of 24.0 m (80 feet) at the San Joaquin County site to 700.0 m (2,300 feet) at Kennedy Table in Madera County (USFWS 2011; NatureServe, 2015).

Castilleja campestris ssp. *succulenta* is an annual plant. Seeds of the *C. campestris* ssp. *succulenta* do not require the presence of a host to germinate, as they form root connections only after reaching a seedling stage.

Population Status

<u>*Rangewide Status of the Species*</u>

The historical distribution between 1937 and 1986 was reported from 33 occurrences, all in the Southern Sierra Foothills Vernal Pool Region (USFWS 2011). Sixteen of those occurrences, including the type locality, were in eastern Merced County. Six occurrences each were in Fresno and Madera counties and five others were in Stanislaus County (USFWS 2011).

The fleshy owl's-clover is found primarily in vernal pools along the lower rolling foothill grasslands in the eastern San Joaquin Valley of the Southern Sierra Foothills Vernal Pool Region (USFWS 2011).

<u>*Population Summary*</u>

At the time of the listing in 1997, there were 36 extant occurrences of *Castilleja campestris* ssp. *succulenta* and currently there are 90 presumed extant occurrences (USFWS 2011). The increase in occurrences is most likely a result of an increased number of surveys. Since the final listing rule, an additional threat to *Castilleja campestris* ssp. *succulenta* is that many of its populations are small in number. A small population size makes a population more vulnerable to extirpation from chance events (USFWS 2011).

<u>*Threats*</u>

Threats to this species include:

- The 1997 final rule stated that nearly half of the extant *Castilleja campestris* ssp. *succulenta* occurrences are threatened by man-made activities such as urbanization, agricultural land conversion, discing, trampling due to overgrazing, mining, and a proposed road expansion project. The threats presented in the listing rule are still relevant. The habitat of this species has

been reduced and fragmented throughout its range and vernal pools continue to be removed by the factors previously noted. Lands on the Central Valley floor are closer to existing cities and agricultural lands than the valley rim, which is steeper, less fertile and more removed from cities. As a result, valley floor vernal pools, along with open rangeland, have been and continue to be favored for urban and agricultural development (USFWS 2011).
- Since the final listing rule, an additional threat to *Castilleja campestris* ssp. *succulenta* is that many of its populations are small in number. A small population size makes a population more vulnerable to extirpation from chance events as noted in the 2005 Recovery Plan.
- This taxon is very cyclical and is somewhat scarce in normal or below normal rainfall years but large populations may be evident in wet years at the known sites (USFWS 2011).
- Climate change is another threat to this species.

Five-Year Status Review

On September 8, 2011, the USFWS issued a five-year status review of the fleshy owl's-clover, which resulted in no change in listing status (USFWS 2011). On August 30, 2023, the U.S. Fish and Wildlife Service completed another five-year status review of the fleshy owls-clover, and concluded that this species' threatened status would remain unchanged (USFWS 2023).

Critical Habitat

Critical habitat was designated for the fleshy owl's-clover on February 10, 2006 (71 FR 7118). The critical habitat designation for *Castilleja campestris* ssp. *succulenta* includes six units (some with multiple parts) in Fresno, Madera, Mariposa, Merced, San Joaquin, Stanislaus, and Tuolumne counties, California. This species critical habitat encompasses approximately 175,873 acres (71 FR 7118).

- Unit 1: Sacramento and San Joaquin counties, California. From USGS 1:24,000 scale quadrangles Clay and Lockeford.
- Unit 2: Tuolumne and Stanislaus counties, California. From USGS 1:24,000 scale quadrangles Keystone, La Grange, Cooperstown and Paulsell.
- Unit 3: Mariposa and Merced counties, California. (i) Unit 3A: Mariposa and Merced counties, California. From USGS 1:24,000 scale quadrangles Merced Falls and Snelling.
- Unit 3B: Mariposa and Merced counties, California. From USGS 1:24,000 scale quadrangles Merced Falls, Snelling, Indian Gulch, Haystack Mountain, Yosemite Lake, Winton, Owen's Reservoir, Planada and Merced.
- Unit 4: Madera and Merced counties, California. (i) Unit 4A: Madera and Merced counties, California. From USGS 1:24,000 scale quadrangle Raynor Creek.
- Unit 4C: Madera and Fresno counties, California. From USGS 1:24,000 scale quadrangles Millerton Lake West, Little Table Mountain, Daulton, Friant, Lanes Bridge and Gregg.
- Unit 5: Fresno County, California. (i) Unit 5A: Fresno County, California. From USGS 1:24,000 scale quadrangles Friant and Round Mountain.
- Unit 5B: Fresno County, California. From USGS 1:24,000 scale quadrangle Clovis.
- Unit 6: Fresno County, California. (i) Unit 6A: Fresno County, California. From USGS 1:24,000 scale quadrangles Millerton Lake East and Academy.
- Unit 6B: Madera County, California. From USGS 1:24,000 scale quadrangles North Fork and Millerton Lake East.

Primary constituent elements (PCEs) are the physical and biological features of critical habitat essential to a species' conservation. The primary constituent elements of critical habitat for *Castilleja campestris* ssp. *succulenta* (Fleshy owl's-clover) are the habitat components that provide:

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described in paragraph (ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools; and

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the fleshy owl's-cover (USFWS 2005).

<u>*Recovery Actions*</u>

- Conduct standardized vernal pool habitat site assessments for both the Southeastern Sacramento Valley and Southern Sierra Foothills vernal pool regions (USFWS 2011).
- Establish management and monitoring plans which include criteria for frequent surveys in order to capture the blooming period for this species. The *Castilleja campestris* ssp. *succulenta* population numbers vary widely from year to year depending on habitat conditions and rainfall (Vollmar 2002). Therefore, the Service should encourage bank owners and preserve managers to perform surveys on a frequent schedule in order to gather additional data which will increase knowledge. The additional information will be utilized for future 5-year reviews (USFWS 2011).
- The Vernal Pool Regional working group should formulate a plan to reach out and educate private landowners as to the value of federally-listed species on their lands, with a particular focus on plants. The Vernal Pool Regional group also should provide guidance to assist landowners on how to better manage their lands for the overall benefit of this species (USFWS 2011).
- The Service should encourage collection of seeds and storage in approved seed banks from extant occurrences, in each core area, to aid in the establishment of a seed bank (USFWS 2011).
- The Service should encourage County and local governments to consider developing Habitat Conservation Plans (HCPs) to include vernal pool species. Take of a federally-listed invertebrate species would be permitted on private land, and any habitat acquisition to compensate for invertebrate species could include the *Castilleja campestris* ssp. *succulenta* if appropriate. Fresno County has been awarded Federal funds for the development of an HCP and additional funds may be available in the future for counties who apply for them (USFWS 2011).
- Efforts to protect vernal pool species should include conservation efforts on a landscape scale (Vollmar 2002). Landscape Conservation Cooperatives provide Federal scientific and technical support for conservation on a landscape scale which is the entire range of an identified priority species. These cooperatives also have a role in helping partners identify common goals and priorities to target the right science for efficient and effective conservation (USFWS 2011).

Environmental Baseline

The fleshy owl's-clover and its designated critical habitat only occur in the eastern San Joaquin Valley in the Southern Sierra Foothills Vernal Pool Region, in California. Please refer to information above for the environmental baseline.

Literature Cited

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Vollmar, J.E., (editor). 2002. Wildlife and rare plant ecology of Eastern Merced County's vernal pool grasslands. Vollmar Consulting, Berkeley, California. pages 75-80. 62 FR 14338. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for Three Plants and Threatened Status for Five Plants From Vernal Pools in the Central Valley of California. Final Rule. Vol 62, No. 58. Federal Register 14338. March 26, 1997. Available online at: https://www.govinfo.gov/content/pkg/FR-1997-03-26/pdf/97-7619.pdf#page=1

71 FR 7118. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for Four Vernal Pool Crustaceans and Eleven Vernal Pool Plants. Final Rule, Administrative Revisions. Vol. 71, No. 28. Federal Register 7118. February 10, 2006. Available online at: https://www.govinfo.gov/content/pkg/FR-2006-02-10/pdf/06-1080.pdf#page=2

Hairy Orcutt Grass (*Orcuttia pilosa*) and its Critical Habitat

Listing Status

The hairy Orcutt grass (*Orcuttia pilosa*) was listed as endangered on March 26, 1997 (62 FR 14338). Critical habitat was designated for the hairy Orcutt grass on February 10, 2006 (71 FR 7118).

Life History and Habitat

This species grows in vernal pools occurring on the eastern side of the Central Valley. The plant germinates underwater and blooms after drydown (NatureServe 2015).

Other members of the genus are known to be wind pollinated and dispersed by water (floating) and adhering to fur and feet with the sticky exudate. Given the close similarity of congeners, it is likely *Orcuttia pilosa* does the same. *O. pilosa* germinates in standing water and flowers after pool bottom is dry. *O. pilosa* is often the only living plant remaining in the dry and cracked vernal pool bed in late summer. This species appears to need fairly constant water levels during the winter. This seems to limit distribution more than the size of the vernal pool. *O. pilosa* seem to be poor competitors. Cocklebur (*Xanthum* sp.) competes directly by shading. In some years cocklebur forms 100% cover during the peak of *O. pilosa*. The hairy Orcutt grass may tolerate light to moderate grazing. Plants require a well developed soil. Habitat creation is probably impossible because of soil requirements; Predominantly outcrossing (NatureServe 2015).

Population Status

Rangewide Status of the Species

Orcuttia pilosa occurs over a 490 km stretch on the eastern margin of the San Joaquin and Sacramento Valleys from Tehama County south through Merced and Mariposa counties, California (NatureServe 2015).

Population Summary

Of 36 occurrences of *Orcuttia pilosa*, 12 are known to be extirpated, 9 are of unknown condition and only 6 are considered stable (NatureServe 2015).

Threats

Threats (USFWS 2009) to this species include:

- Urbanization.
- Agricultural conversion.
- Highway expansion.
- Off-road vehicle use.
- Livestock grazing (and trampling).
- Invasive plants.
- Inadequacy of existing regulatory mechanisms.
- Drought and climate change.

Five-Year Status Review

On June 15, 2009, the USFWS issued a five-year status review of the hairy Orcutt grass, which resulted in no change in listing status (USFWS 2009). On August 8, 2024, the U.S. Fish and Wildlife Service completed another five-year status review of the hairy Orcutt grass, and concluded that this species' endangered status would remain unchanged (USFWS 2024).

Critical Habitat

Critical habitat was designated for the hairy Orcutt grass on February 10, 2006 (71 FR 7118). The critical habitat designation for *Orcuttia pilosa* is in Butte, Fresno, Madera, Mariposa, Merced, Stanislaus, and Tehama counties, California. This species critical habitat encompasses approximately 79,608 acres (71 FR 7118).

- Unit 1: Tehama County, California. From USGS 1:24,000 topographic quadrangles Acorn Hollow and Richardson Springs NW.
- Unit 2: Butte County, California. From USGS 1:24,000 topographic quadrangle Hamlin Canyon.
- Unit 4: Merced, Mariposa, and Stanislaus counties, California. (i) Unit 4A: Merced, Mariposa, and Stanislaus counties, California. From USGS 1:24,000 topographic quadrangles Paulsell, Cooperstown, Le Grange, Montpelier, Turlock Lake, Snelling, and Merced Falls. (ii) Unit 4B: Stanislaus County, California. From USGS 1:24,000 topographic quadrangles Paulsell and Montpelier. (iii) Unit 4C: Merced County, California. From USGS 1:24,000 topographic quadrangle Turlock Lake.
- Unit 5: Madera County, California. (i) Unit 5A: Madera County, California. From USGS 1:24,000 topographic quadrangle Daulton. Unit 5B: Madera County, California. From USGS 1:24,000 topographic quadrangle Daulton.
- Unit 6: Madera County, California. From USGS 1:24,000 topographic quadrangles Daulton, Little Table Mountain, Gregg, and Lanes Bridge.

Primary constituent elements (PCEs) are the physical and biological features of critical habitat essential to a species' conservation. The primary constituent elements of critical habitat for the hairy Orcutt grass (*Orcuttia pilosa*) are the habitat components that provide:

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described in paragraph (ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools; and

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the hairy Orcutt grass (USFWS 2005).

Recovery Actions

Recovery actions (USFWS 2009) for this species include the following:

1. Habitat protection: Accomplish habitat protection that promotes vernal pool ecosystem function sufficient to contribute to population viability of the covered species.

1A. Suitable vernal pool habitat within each prioritized core area for the species is protected.

1B. Species occurrences distributed across the species geographic range and genetic range are protected. Protection of extreme edges of populations protects the genetic differences that occur there.

1C. Reintroductions must be carried out and meet success criteria established in the recovery plan.

1D. Additional occurrences identified through future site assessments, GIS and other analyses, and status surveys that are determined essential to recovery are protected. Any newly found occurrences may count towards recovery goals if the occurrences are permanently protected as described in the recovery plan.

1E. Habitat protection results in protection of hydrology essential to vernal pool ecosystem function, and monitoring indicates that hydrology that contributes to population viability has been maintained through at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

2. Adaptive Habitat Management and Monitoring:

2A. Habitat management and monitoring plans that facilitate maintenance of vernal pool ecosystem function and population viability have been developed and implemented for all habitat protected, as previously discussed in Sections 1 (A-E).

2B. Mechanisms are in place to provide for management in perpetuity and long-term monitoring of habitat protected in Sections 1 (A-E), as previously discussed (funding, personnel, etc.).

2C. Monitoring indicates that ecosystem function has been maintained in the areas protected under Sections 1 (A-D) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

2D. Seed banking actions have been completed for species that would require it as insurance against risk of stochastic extirpations or that will require reintroductions or introductions to contribute to meeting recovery criteria.

3. Status Surveys:

3A. Status surveys, 5-year status reviews, and population monitoring show populations within each vernal pool region where the species occur are viable (e.g., evidence of reproduction and recruitment) and have been maintained (stable or increasing) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

3B. Status surveys, status reviews, and habitat monitoring show that threats identified during and since the listing process have been ameliorated or eliminated. Site-specific threats identified through standardized site assessments and habitat management planning also must be ameliorated or eliminated.

4. Research:

    4A. Research actions necessary for recovery and conservation of the covered species have been identified (these are research actions that have not been specifically identified in the recovery actions but for which a process to develop them has been identified). Research actions (both specifically identified in the recovery actions and determined through the process) on species biology and ecology, habitat management and restoration, and methods to eliminate or ameliorate threats have been completed and incorporated into habitat protection, habitat management and monitoring, and species monitoring plans, and refinement of recovery criteria and actions.

    4B. Research on genetic structure has been completed (for species where necessary – for reintroduction and introduction, seed banking) and results incorporated into habitat protection plans to ensure that within and among population genetic variation is fully representative by populations protected in the Habitat Protection section of this document, described previously in Sections 1 (A-E).

    4C. Research necessary to determine appropriate parameters to measure population viability for each species have been completed.

5. Participation and outreach:

    5A. Recovery Implementation Team is established and functioning to oversee rangewide recovery efforts.

    5B. Vernal pool regional working groups are established and functioning to oversee regional recovery efforts.

    5C. Participation plans for each vernal pool region have been completed and implemented.

    5D. Vernal pool region working groups have developed and implemented outreach and incentive programs that develop partnerships contributing to achieving recovery criteria 1-4.

Environmental Baseline

The hairy Orcutt grass and its designated critical habitat only occur San Joaquin and Sacramento Valleys from Tehama County south through Merced and Mariposa counties, California. Please refer to information above for the environmental baseline.

Literature Cited

NatureServe. 2015. NatureServe Explorer, An online encyclopedia of life [web application]. Available online at: http://explorer.natureserve.org/.

USFWS (U.S. Fish and Wildlife Service). 2005. Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon. Portland, Oregon. xxvi + 606 pages. December 15.

USFWS (U.S. Fish and Wildlife Service). 2009. *Orcuttia pilosa* (Hairy Orcutt Grass) 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service Sacramento Fish and Wildlife Office Sacramento, California. June 15.

USFWS (U.S. Fish and Wildlife Service). 2024. *Orcuttia pilosa* (Hairy Orcutt Grass) 5-Year Review: Summary and Evaluation. U.S. Fish and Wildlife Service Sacramento Fish and Wildlife Office Sacramento, California. August 8.

62 FR 14338. Endangered and Threatened Wildlife and Plants; Determination of Endangered Status for Three Plants and Threatened Status for Five Plants From Vernal Pools in the Central Valley of

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71 FR 7118. Endangered and Threatened Wildlife and Plants; Designation of Critical Habitat for Four Vernal Pool Crustaceans and Eleven Vernal Pool Plants. Final Rule, Administrative Revisions. Vol. 71, No. 28. Federal Register 7118. February 10, 2006. Available online at: https://www.govinfo.gov/content/pkg/FR-2006-02-10/pdf/06-1080.pdf#page=2

Hoover's Spurge (*Chamaesyce hooveri*) and its Critical Habitat

Listing Status

The Hoover's spurge (*Chamaesyce hooveri*) was listed as threatened on March 26, 1997 (62 FR 14338). Critical habitat was designated for the Hoover's spurge on February 10, 2006 (71 FR 7118).

Life History and Habitat

Chamaesyce hooveri is restricted to vernal pools. Deeper pools apparently provide better habitat for this species because the duration of inundation is longer and the deeper portions are nearly devoid of other vegetation, thus limiting competition from other plants. However, the plant appears to adapted to a wide variety of soils, which range in texture from clay to sandy loam (USFWS 2005).

Chamaesyce hooveri is a summer annual, but few details of its life history are known. Populations in Merced and Tulare counties typically flower from late May through July, whereas those farther north in Stanislaus County and the Sacramento Valley flower from mid-June into October. Beetles (order Coleoptera), flies (order Diptera), bees and wasps (order Hymenoptera), and butterflies and moths (order Lepidoptera) have been observed visiting the flowers of *Chamaesyce hooveri* and may potentially serve as pollinators (USFWS 2005).

Population Status

Rangewide Status of the Species

For decades, *Chamaesyce hooveri* was known from only three localities: near Yettem and Visalia in Tulare County, and near Vina in Tehama County. Collections were made from these three areas in the late 1930s and early 1940s. From 1974 through 1987, 21 additional occurrences of *C. hooveri* were reported. The majority of these (15) were in Tehama County. One to three occurrences were discovered during this period in each of Butte, Merced, Stanislaus, and Tulare counties. The historical localities for this species were in the Northeastern Sacramento Valley, San Joaquin Valley, Solano-Colusa, and Southern Sierra Foothills Vernal Pool Regions (USFWS 2005).

Of the 26 occurrences presumed to be extant, only 3 have been observed within the past decades. The main remaining area of concentration for *Chamaesyce hooveri* is within the Northeastern Sacramento Valley Vernal Pool Region. The Vina Plains of Tehama and Butte counties contain 14 (53.8 percent) of the 26 known extant occurrences for C. hooveri in an area of about 91 square kilometers (35 square miles). One other site in the same region is near Chico in Butte County. Seven of the extant occurrences are in the Southern Sierra Foothills Vernal Pool Region, including five in the Visalia-Yettem area of Tulare County and two in the Hickman-La Grange area of Stanislaus County. Three other occurrences are on the Sacramento National Wildlife Refuge in Glenn County, which is in the Solano-Colusa Vernal Pool Region. The one other extant occurrence is on the Bert Crane Ranch in Merced County, which is within the San Joaquin Valley Vernal Pool Region (USFWS 2005).

Population Summary

The Sacramento National Wildlife Refuge populations have been monitored annually since 1992. *Chamaesyce hooveri* is known to have occurred in 11 pools on the Refuge between 1992 and 2006. It is not seen in all the pools every year. In 2006, it was observed in 4 pools totaling over 1,200 plants. Population numbers have ranged from less than 100 plants seen in 2001 to over 2,500 plants seen in 1993 (USFWS 2009). Of the 31 known occurrences and sites, 27 are presumed to be extant (USFWS 2009).

Threats

Threats to this species include:

- Habitat loss occurs from direct destruction and modification of pools due to filling, grading, discing, leveling, paving, and other activities, as well as modification of surrounding uplands, which alters vernal pool watersheds and the supporting upland ecosystem. Fifty-five percent of presumed extant sites of *C. hooveri* are on private land and are not protected (USFWS 2009).
- During the 30 years prior to listing, agricultural land conversion was known to have caused the extirpation of one population and threatened two more populations of *C. hooveri* in Tulare County (USFWS 2009).
- Vernal pool habitats in the Central Valley now represent approximately 9 percent of their former area, and remaining habitats are considerably more fragmented and isolated than historically and during the recent past (USFWS 2009).
- Competition from invasive native or non-native plant species threatens nine of the extant occurrences, including eight in the Vina Plains and one on the Sacramento National Wildlife Refuge in Glenn County (USFWS 2009).
- *Chamaesyce hooveri* is an obligate wetland species found only in vernal pools, typically on alluvial fans or terraces of ancient rivers or streams, with a few on the rim of the Central Valley basin. Therefore, maintenance of the natural hydrology of the pools is necessary for the survival and recovery of this species. Drought or flood conditions will place additional strains on the vernal pool ecosystem supporting *C. hooveri* occurrences, some of which are already fragmented or reduced by agricultural conversion and development. Where occurrences persist on only marginal habitat, the addition of extreme drought conditions is likely to result in higher rates of mortality in the short term with the effects of low reproductive output and survivorship persisting after the drought has ceased (USFWS 2009).
- Small population size poses a serious threat for at least four of the known occurrences, which total fewer than 100 individuals even in favorable years (USFWS 2009). Such small populations are subject to extirpation from random events such as extended drought and genetic drift. Small population size makes it difficult for this species to persist while sustaining the impacts of habitat fragmentation. Such populations may be highly susceptible to extirpation due to chance events, inbreeding depression, or additional environmental disturbance (USFWS 2009).

Five-Year Status Review

On February 4, 2009, the USFWS issued a five-year status review of the Hoover's spurge, which resulted in no change in listing status (USFWS 2009). On September 9, 2023, the U.S. Fish and Wildlife Service completed another five-year status review of Hoover's spurge, and concluded that this species' threatened status would remain unchanged (USFWS 2023).

Critical Habitat

Critical habitat was designated for the Hoover's spurge on February 10, 2006 (71 FR 7118). The critical habitat designation for *Chamaesyce hooveri* includes seven units in Merced, Stanislaus, Tehama, Tulare, and Tuolumne counties, California. This species critical habitat encompasses approximately 114,713 acres (46,423 hectares) (71 FR 7118).

- Unit 1: Tehama County, California. From USGS 24,000 topographic quad Acorn Hollow, Richardson Springs NW.
- Unit 2: Butte County, California. From USGS 24,000 topographic quad Hamlin Canyon.
- Unit 4: Stanislaus and Tuolumne counties.
- Unit 5: Stanislaus and Merced counties. (i) Unit 5A: Stanislaus and Merced counties. From USGS 24,000 topographic quads Paulsell, Cooperstown, Le Grange, Montpelier, Turlock Lake, Snelling, Merced Fall. (ii) Unit 5B: Merced County. From USGS 24,000 topographic quad

Turlock Lake. (iii) Unit 5C: Stanislaus County. From USGS 24,000 topographic quads Paulsell, Montpelier.

- Unit 6: Merced County. (i) Unit 6A: Merced County. USGS 24,000 topographic quads Stevinson, San Luis Ranch. Unit 6B: Merced County. From USGS 24,000 topographic quad Stevinson, Arena, San Luis Ranch, Turner Ranch. Unit 6C: Merced County. From USGS 24,000 topographic quad Arena, Turner Ranch. Unit 6D: Merced County. USGS 24,000 topographic quad Turner Ranch, Sandy Mush. Unit 6E: Merced County. USGS 24,000 topographic quad Turner Ranch, Sandy Mush.
- Unit 7: Tulare County. (i) Unit 7A: Tulare County. From USGS 24,000 topographic quads Stokes Mtn., Ivanhoe. (ii) Unit 7B: Tulare County. From USGS 24,000 topographic quads Ivanhoe. (iii) Unit 7C: Tulare County. From USGS 24,000 topographic quads Stokes Mtn., Auckland, Ivanhoe, Woodlake. Unit 7D: Tulare County. From USGS 24,000 topographic quad Woodlake. Unit 7E: Tulare County. From USGS 24,000 topographic quad Monson. Unit 7F: Tulare County. USGS 24,000 topographic quad Monson. Unit 7G: Tulare County. USGS 24,000 topographic quad Monson.
- Unit 3 (excluded): Glenn and Colusa counties, California. This unit was excluded from the designation pursuant to Section 4(b)(2) of the Act (see Exclusions under 4(b)(2) in the final critical habitat rule (70 FR 46924).

Primary constituent elements (PCEs) are the physical and biological features of critical habitat essential to a species' conservation. The PCEs of *Chamaesyce hooveri* critical habitat consists of two components (71 FR 7118).

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described below in paragraph (ii), providing for dispersal and promoting hydroperiods of adequate length in the pools;

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the Hoover's spurge (USFWS 2005).

Recovery Actions

- Protect vernal pool habitat from being destroyed or modified by development, agriculture, or other activities. Acquiring conservation easements or fee title to habitat lands are some ways that conservators can help guarantee protection of the species in perpetuity (USFWS 2009).
- Develop standardized population trend survey protocols and implement to complete updated status surveys, especially for populations on private lands where trends have not been recently updated (USFWS 2009).
- Manage invasive plants on preserves. Management should include research to determine effective eradication methods of nonnative competitors, and pool conditions that favor one plant over another (USFWS 2009).

- Create and convene regional vernal pool working groups in regions where *Chamaesyce hooveri* occurs. Regional vernal pool working groups will be important for the tracking the progress of recovery efforts, including the amount of suitable habitat protected for each of the species in the core areas (USFWS 2009).
- Collect seeds from each core area following the Center for Plant Conservation Guidelines (1991). Seed collections should be stored in at least two sites, including the National Center for Genetic Resources in Fort Collins, Colorado, and a facility certified by the Center for Plant Conservation (USFWS 2009).

Environmental Baseline

The Hoover's spurge and its designated critical habitat occur in the Central Valley and Southern Sierra Foothills Vernal Pool Regions, California. Please refer to information above for the environmental baseline.

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USFWS (U.S. Fish and Wildlife Service). 2005. Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon. Portland, Oregon. xxvi + 606 pages. December 15.

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Otay Mesa-mint (Pogogyne nudiuscula)

Listing Status

Otay mesa-mint was federally listed as endangered on August 3, 1993, due to habitat loss and degradation from urban and agricultural development, livestock grazing, off-road vehicle use, trampling, invasion from weedy non-native plants, and other factors (58 FR 41384).

Life History and Habitat

Otay mesa-mint is an annual herb in the Lamiaceae (mint family) that is restricted to vernal pools in southern San Diego County, California. Plants can reach one foot or more in height with purple flowers arranged in whorls that typically bloom from May or June through early July (Service 2021). Vernal pools and vernal swales are often clustered into pool "complexes," and may form dense, interconnected mosaics of small pools, or a sparse scattering of larger pools. Vernal pool complexes that support from one up to many distinct vernal pools are often interconnected by a shared watershed. Both the pool basin and the surrounding watershed are essential for a functioning vernal pool system (Service 2021).

Population Status

There are 24 Otay mesa-mint locations: 17 are extant, two are presumed extant, three are historically extirpated, and two have questionable identification. There are five new occurrences since the 2010 5-year review, and no locations have been extirpated since listing. It is possible that Otay mesa-mint occurs at other locations that have not been surveyed (Service 2021).

Threats such as development, nonnative plants, human access and disturbance, and fire and fire suppression are currently impacting Otay mesa-mint. However, the number of vernal pool complexes threatened by development, compaction of soils, altered hydrology, road projects, human disturbance, and off-highway vehicles have decreased due to land conservation and restoration efforts. Competition with nonnative plants remains a threat at many occurrences and is managed to some degree by partners (Service 2021).

Recovery Plan Information

A recovery plan for Otay mesa-mint and other vernal pool species was released on September 3, 1998 (Service 1998) and a clarification to this plan was released on October 1, 2019 (Service 2019). The delisting criteria include the following:

1) All 74 geographic areas and associated vernal pool complexes as identified in Appendices F and G of the 1998 Recovery Plan under each of the specific management areas are protected and managed to ensure long-term viability.
2) The Service must determine that the following factors are no longer present, or continue to adversely affect, Otay mesa-mint: (a) the present or threatened destruction, modification, or curtailment of their habitat range; (b) over utilization for commercial, recreational, scientific, or educational purposes; (c) disease or predation; (d) the inadequacy of existing regulatory mechanisms; and (e) other natural and manmade factors affecting their continued existence.
3) Population trends continue to be stable or increasing for 10 consecutive years after threats have been sufficiently ameliorated or managed completion of delisting criterion 2 prior to consideration for delisting.

Environmental Baseline

Since the known occurrences of Otay mesa-mint occur entirely within California, the status description above also serves as the baseline for this consultation.

Literature Cited

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Sacramento Orcutt Grass (*Orcuttia viscida*) and its Critical Habitat

Listing Status

The Sacramento Orcutt grass (*Orcuttia viscida*) was listed as endangered on March 26, 1997 (62 FR 14338). Critical habitat was designated for the Sacramento Orcutt grass on February 10, 2006 (71 FR 7118).

Life History and Habitat

Orcuttia viscida is known only from vernal pool habitats in a 22-square-mile area in Sacramento County, California. *O. viscida* requires a very well-developed soil with a silica-iron hardpan layer 2-10 feet below ground level. This impermeable hardpan causes water to perch above ground. Habitat creation for the genus *Orcuttia* is probably impossible because of its specific soil requirements (NatureServe 2015).

Other members of the genus are known to be wind pollinated and dispersed by water and by adhering to feet and fur with the sticky exudate. Given the similarity between congeners, it is likely O. viscida shares these characteristics (NatureServe 2015).

Other members of the genus are known to be wind pollinated and dispersed by water and by adhering to feet and fur with the sticky exudate. Given the similarity between congeners, it is likely O. viscida shares these characteristics.; Genus Orcuttia forms a distinct group within the grass family with no apparent affinities to any other grasses, probably of ancient origin. Common associates include coyote thistle (*Eryngium* spp.), spike rush (*Eleocharis* spp.), Carter's buttercup (*Ramnunculus alveolatus*), double-horned downingia (*Downingia bicornata*), white-flowered navarretia (*Navarrettia leucocephala*), and annual checkerbloom (*Sidalcea calycosa*). *O. viscida* requires enough standing water to allow the growth of an anaerobic fungus over the seed coat to break dormancy. In drier years the seeds remain dormant. Seeds may remain viable for many years. Orcuttia seem to be poor competitors and only grow in areas where prolonged (but not constant) inundation drowns out competitors; Predominantly outcrossing (NatureServe 2015).

Population Status

<u>Rangewide Status of the Species</u>

The Sacramento Orcutt grass is known only from Sacramento County, California in two main clumps. The two areas add up to approximately 22 square miles of range extent (NatureServe 2015).

<u>Population Summary</u>

The Sacramento Orcutt grass is highly vulnerable. Long term trend probably has been one of moderate to substantial decline, of approximately 30-70%. In a good year, there can be as many as greater than 2 million total plants. But plant numbers are not very informative here. Known from 9 total occurrences, one of which is historical and extirpated (NatureServe 2015). Low redundancy, resiliency and representation are inferred based on the low number of populations and restricted geography of this species.

The current population trend information (numbers of plants) for *Orcuttia viscida* indicates this species appears to be stable at five of the nine occurrences. No quantitative information is available for the other four locations. However, threats to *Orcuttia viscida* from loss of habitat, primarily from urbanization and land conversion to agriculture, continue at the single unprotected occurrence located east of Grantline Road. Competition from nonnative, aggressive plant species, especially *Glyceria declinata* (waxy manna grass), threatens at least five occurrences of *Orcuttia viscida*. *Parentucellia viscosa* (sticky bartsia) has become established at Kiefer Landfill Wetland Preserve and likely threatens the *Orcuttia viscida* occurrences there (USFWS 2008).

California Natural Diversity Database reports the existence of nine extant occurrences of *Orcuttia viscida*, whereas the recovery plan reported eight occurrences. The location of the most recently recorded occurrence, at Arroyo Seco Conservation Bank, which was not included in the Recovery Plan, is within the known range of the species and is approximately 6.4 kilometers (4 miles) from another extant occurrence (USFWS 2008). Therefore, this additional occurrence does not substantially increase the amount of known occupied habitat and is not a range extension. Although the occurrences which have been monitored appear to be stable, many of the occurrences occupy small areas and have a small number of plants. For example, *Orcuttia viscida* at the Rancho Seco occurrence occupied two vernal pools in previous years but only 17 plants in a single pool could be found in 2005 (USFWS 2008).

Threats

Threats to this species include:

- Urbanization continues to be the greatest threat to the single, unprotected occurrence, located east of Grantline Road (USFWS 2008).
- Proposed expansion of Kiefer Landfill is listed as a threat to this species (USFWS 2008).
- Proposed gravel and aggregate mining (62 FR 14338) is listed as a threat to this species (USFWS 2008).
- It is estimated that if the *Glyceria declinata* populations in *Orcuttia viscida* habitat grow at the rate of the San Joaquin or Phoenix Park populations, *O. viscida* could be completely displaced by *G. declinata* in 10 years or less. Voluntary efforts to remove *G. declinata* at Phoenix Park by handpulling have been the only efforts to control the species in *O. viscida* habitat. At Kiefer Landfill Wetland Preserve, sticky bartsia (*Parentucellia viscosa*) is invading the upper edges of the vernal pools that surround the vernal pools supporting *Orcuttia viscida*. The effects of this species on *Orcuttia viscida* are currently unknown; however, this species warrants observation (USFWS 2008).
- Habitat for *Orcuttia viscida* continues to be highly fragmented throughout its range due to conversion of natural habitat for urban and agricultural uses. This fragmentation has resulted in small, isolated populations of this species. For example, at least three occurrences are each found in single vernal pools. Such populations may be highly susceptible to extirpation due to chance events, inbreeding depression, or additional environmental disturbance. If an extirpation event occurs in a population that has been fragmented, the opportunities for recolonization will be greatly reduced due to physical isolation from other source populations (USFWS 2008).
- Climate change is a threat to this species (USFWS 2008).

Five-Year Status Review

On June 15, 2008, the USFWS issued a five-year status review of the Sacramento Orcutt grass, which resulted in no change in listing status (USFWS 2008). On September 19, 2024, the U.S. Fish and Wildlife Service completed another five-year status review of the Sacramento Orcutt grass, and concluded that this species' endangered status would remain unchanged (USFWS 2024).

Critical Habitat

Critical habitat was designated for the Sacramento Orcutt grass on February 10, 2006 (71 FR 7118). The critical habitat designation for *Orcuttia viscida* includes three units in Amador and Sacramento counties, California. This species critical habitat encompasses approximately 33,273 acres (ac) (13,465 hectares (ha)) (71 FR 7118).

- Unit 1: Sacramento County, California. From USGS 1:24,000 topographic quadrangle Folsom.
- Unit 2: Sacramento County, California. From USGS 1:24,000 topographic quadrangle Carmichael.
- Unit 3: Sacramento and Amador counties, California. From USGS 1:24,000 topographic quadrangles Sloughhouse, Carbondale, Clay, and Goose Creek.

Primary constituent elements (PCEs) are the physical and biological features of critical habitat essential to a species' conservation. The PCEs of critical habitat for Sacramento Orcutt grass (*Orcuttia viscida*) are the habitat components that provide (71 FR 7118):

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described in paragraph ((ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools; and

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the Sacramento Orcutt grass (USFWS 2005).

Recovery Actions

- Conduct a study to identify methods to control the dispersal of the invasive grass, *Glyceria declinata*, in vernal pool habitat (USFWS 2008).
- Develop and implement a management plan for control of nonnative, competitive plants, particularly *Glyceria declinata*. Phoenix Park, Phoenix Field, and Kiefer Landfill Wetland Preserve should be targeted for immediate control of *Glyceria declinata*. All remaining *Orcuttia viscida* occurrences should be surveyed for presence of *Glyceria declinata* and managed accordingly (USFWS 2008).
- Introduce appropriate levels of grazing at the Rancho Seco site to benefit the *Orcuttia viscida* occurrence (USFWS 2008).
- Work with SMUD to permanently protect the *Orcuttia viscida* plants and habitat, facilitate livestock watering improvements, and improve the cattle grazing regime to benefit *Orcuttia viscida* (USFWS 2008).
- Conduct genetic research on *Glyceria declinata* to clarify its taxonomy (USFWS 2008).

Environmental Baseline

The Sacramento Orcutt grass and its designated critical habitat only occur Amador and Sacramento counties, California. Please refer to information above for the environmental baseline.

Literature Cited

NatureServe. 2015. NatureServe Explorer, An online encyclopedia of life [web application]. Available online at: http://explorer.natureserve.org/.

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San Diego Ambrosia (*Ambrosia pumila*)

Listing Status

San Diego ambrosia was federally listed as endangered on July 2, 2002, due to present or threatened destruction, fragmentation, and degradation of habitat primarily by construction and maintenance of highways, maintenance of utility easements, development of recreational facilities, and residential and commercial development; inadequate regulatory mechanisms; potential competition, encroachment, and other negative impacts from non-native plants; mowing and discing for fuel modification; and trampling, as well as soil compaction by horses, humans, and vehicles (67 FR 44372). Critical habitat was designated on November 30, 2010 (75 FR 74546).

Life History and Habitat

San Diego ambrosia is a clonal herbaceous perennial plant occurring in southern California. It is historically known from western Riverside County, south through western San Diego County, to central Baja California, Mexico. The species is found primarily on upper terraces of rivers and drainages. However, several patches occur within the watershed of a large vernal pool at the Barry Jones (Skunk Hollow) Wetland Mitigation Bank in Riverside County and near dry lake beds in Baja California, Mexico (Service 2021).

Population Status

At listing, 15 native occurrences of San Diego ambrosia were considered extant in the United States: three in Riverside County and 12 in San Diego County. There are currently 37 occurrences in the United States that are presumed extant, including 11 from translocations. In addition, 31 occurrences are known from three geographic areas in northern Baja California, Mexico and two records from southern Baja California, Mexico (Service 2021).

The 2010 5-year review identified habitat fragmentation and climate change as additional threats to the species and that grazing was no longer a threat. Inadequate regulatory mechanism was previously considered a threat but is no longer considered to be a threat. At the 2010 5-year review, some degree of conservation was afforded to 11 of 16 occurrences (Service 2010). Of the 26 extant, natural occurrences of San Diego ambrosia in the United States documented in the 2021 5-year review, only 6 are completely conserved and 9 are partially conserved. The remaining 11 occurrences are not conserved and are more vulnerable to habitat loss from urban development. Protections afforded under the approved, regional habitat conservation plans have decreased but not eliminated major habitat loss and alteration. Overall, 41 percent (78.4 of 191.8 acres) of occupied habitat (natural, extant records) is considered conserved, typically with some degree of management including 15.1 of 54.4 acres in Riverside County and 63.4 of 137.4 acres in San Diego County (Service 2021). None of the San Diego ambrosia in Baja California, Mexico is conserved or provided regulatory protection.

Critical Habitat

Designated critical habitat occurs in seven units in Riverside and San Diego counties for a total of approximately 783 acres. The physical and biological features of designated critical habitat include:

1. Sandy loam or clay soils (regardless of disturbance status), including (but not limited to) the Placentia (sandy loam), Diablo (clay), and Ramona (sandy loam) soil series that occur near (up to several hundred meters from but not directly adjacent to) a river, creek, or other drainage, or within the watershed of a vernal pool, and that occur on an upper terrace (flat or gently sloping areas of 0 to 42 percent slopes are typical for terraces on which San Diego ambrosia occurrences are found).

2. Grassland or ruderal habitat types, or openings within coastal sage scrub, on the soil types and topography described in physical and biological feature 1, that provide adequate sunlight, and airflow for wind pollination.

Environmental Baseline

The status description above also serves as the environmental baseline, except for the 31 occurrences in Mexico for which there is limited information.

Literature Cited

Service (U.S. Fish and Wildlife Service). 2010. *Ambrosia pumila* (San Diego ambrosia) 5-year review: summary and evaluation. 39 pp.

Service (U.S. Fish and Wildlife Service). 2021. Five-year review: *Ambrosia pumila* (San Diego ambrosia) 19 pp.

San Diego Button-celery (*Eryngium aristulatum* var. *parishii*)

Listing Status

San Diego button-celery was federally listed as endangered on August 3, 1993, due to habitat loss and degradation from urban and agricultural development, livestock grazing, off-road vehicle use, trampling, invasion from weedy non-native plants, and other factors (58 FR 41384).

Life History and Habitat

San Diego button-celery is a biennial or longer-lived perennial gray-green herb that has a storage tap root. It has a spreading shape and reaches a height of 16 inches. The stems and lanceolate leaves give the plant a prickly appearance. It is a clay soil, surface and non-surface hard pan, vernal pool obligate and relies on ephemerally wet conditions to reproduce, blooming from April to June. It is an outcrossing taxon that reproduces exclusively by seeds (Service 2010).

Population Status

San Diego button-celery currently occurs in 14 geographic areas in Riverside and San Diego counties. Collection records document occurrences in six areas of Riverside County at listing; however, there are now only four sites, all on the Santa Rosa Plateau (Service 2010). Most of the occupied range of the taxon in the United States occurs in ten regional locations in San Diego County including Marine Corps Base Camp Pendleton, Carlsbad, San Marcos, Ramona, Del Mar Mesa, Carmel Mountain, Mira Mesa, Marine Corps Air Station Miramar, Otay Lakes, and Otay Mesa. Current status of the species in Mexico is unknown (Service 2010).

San Diego button-celery can be locally abundant in remnant vernal pools; however, the distribution of this variety has been dramatically reduced due to loss of most (95 to 97 percent) of the vernal pool habitat in San Diego County. In 2003, the City of San Diego conducted a survey of vernal pools within their jurisdiction; these surveys revealed that of the 69 sites surveyed, 28 contained San Diego button-celery and it was found on 20 of 36 acres of basin habitat. Based on survey data at Marine Corps Air Station Miramar that incorporates survey efforts since 1993, San Diego button-celery was found in 20 of 45 vernal pool complexes located on the installation (Service 2010).

At the time of listing, all sites occupied by San Diego button-celery were under threat of development or other impacts. Overall, San Diego button-celery has maintained its population and distribution since the time of listing. Though threats remain, impacts from trampling associated with immigrant travel, road development and construction activities, and mowing and plowing of extant habitat have been minimized as threats. Outside of continued urbanization, climate change and fire may have the longest lasting impact for degrading the species long term retention, setting back potential recovery. The dense concentrations of vernal pools on military bases will be protected from most development but may be subject to off-highway vehicle activity, trampling impacts, and potential habitat impacts if Marine Corps Base Camp Pendleton or Marine Corps Air Station Miramar requires a change in the military mission (Service 2010).

Much progress has been made to conserve vernal pool habitat where San Diego button-celery occurs. Land acquisition and conservation under the Western Riverside County Multiple Species Habitat Conservation Plan and San Diego Multiple Species Conservation Plan, as well as management efforts under the Marine Corps Air Station Miramar and Marine Corps Base Camp Pendleton Integrated Natural Resource Management Plans, have reduced or ameliorated many of the original threats. Regardless, though San Diego button-celery is found to be locally abundant at sites where habitat has been conserved or where management of anthropogenic activities has protected the vernal pool site, impacts from current threats remain (Service 2010).

Recovery Plan Information

A recovery plan for San Diego button-celery and other vernal pool species was released on September 3, 1998 (Service 1998) and a clarification to this plan was released on October 1, 2019 (Service 2019). The delisting criteria include the following:

1) All 74 geographic areas and associated vernal pool complexes as identified in Appendices F and G of the 1998 Recovery Plan under each of the specific management areas are protected and managed to ensure long-term viability.
2) The Service must determine that the following factors are no longer present, or continue to adversely affect, San Diego fairy shrimp: (a) the present or threatened destruction, modification, or curtailment of their habitat range; (b) over utilization for commercial, recreational, scientific, or educational purposes; (c) disease or predation; (d) the inadequacy of existing regulatory mechanisms; and (e) other natural and manmade factors affecting their continued existence.
3) Population trends continue to be stable or increasing for 10 consecutive years after threats have been sufficiently ameliorated or managed completion of delisting criterion 2 prior to consideration for delisting.

Environmental Baseline

Since the known occurrences of San Diego button-celery occur entirely within California, the status description above also serves as the environmental baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 1998. Vernal pools of southern California recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon. 113+pp.

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San Joaquin (= San Joaquin Valley) Orcutt Grass (*Orcuttia inaequalis*) and its Critical Habitat

Listing Status

The San Joaquin Orcutt grass (*Orcuttia inaequalis*) was listed as threatened on March 26, 1997 (62 FR 14338). Critical habitat was designated for the San Joaquin Orcutt grass on February 10, 2006 (71 FR 7118).

Life History and Habitat

Typical landforms upon which *Orcuttia inaequalis* occurs include remnant alluvial fans and stream terraces as well as tabletop lava flows. *O. inaequalis* is known to occur in acidic soils with textures ranging from clay to sandy loam. It has been documented on the Hideaway soil series on Fresno and Madera County tabletops, and Amador, Cometa, Corning, Greenfield, Los Robles, Madera Peters, Pollasky-Montpellier complex, Raynor, Redding and San Joaquin soil series throughout its range (Recovery Plan). Vollmar (2002) reported that *O. inaequalis* populations occur on Riverbank, North Merced Gravels, and Mehrten geologic surfaces, which could relate to the tendency of these surfaces to support larger pools, noting that soil characteristics may also play a role (USFWS 2013).

O. inaequalis is a highly specialized C4 plant (an evolutionary adaptation that facilitates photosynthetic productivity in arid and semi-arid climates) that is dependent on deep vernal pools for survival (USFWS 2013). Species inhabits mall, seasonal pools (NatureServe, 2015). High ecological integrity of the population and site fidelity as well as low tolerance ranges are inferred based on the specific habitat needs of this species and its relatively small geographic range.

Spikelets break apart and scatter their seeds when autumn rains arrive (USFWS 2005).

One reproductive quality observed in Orcuttia species that promotes high genetic variation among successive generations is the flowering pattern. *O. inaequalis* is wind-pollinated, and generally flowers from April to September. The first two flowers on plants of these species open simultaneously and do not produce pollen until the ovaries are no longer receptive. Thus, fertilization for these flowers is solely a result of outcrossing from different plants (USFWS 2013).

Population Status

Rangewide Status of the Species

The historical range of the San Joaquin Orcutt grass is believed to be in the Southern Sierra Foothills Vernal Pool Region, which includes parts of Stanislaus, Merced, Madera, Fresno and Tulare counties, California (USFWS 2013).

The current range of the San Joaquin Orcutt grass includes portions of: Solano, Merced, Madera, Fresno, and Tulare counties, California (USFWS 2013).

Population Summary

At least 16 populations of *O. inaequalis* have been extirpated; 23 populations remain, all within a 79 km-long range (NatureServe 2015).

Across the contemporary range, 14 of 31 (45%) extant O. *inaequalis* localities are currently protected or proposed for protection. Direct impacts from the threat of land conversion or urbanization are currently, or have potential to be, excluded from these localities. Conversely, 17 extant occurrences have no known protection at this time, and therefore continue to be vulnerable to threats. Moreover, the potential effects of climate change could threaten the stability of all localities for this highly specialized species that is dependent upon a specific set of environmental conditions (USFWS 2013).

Threats

Threats to this species include:

- The vast majority of land on the Central Valley floor has potential for urbanization and agricultural conversion due to flat topography and its vicinity to existing infrastructure (USFWS 2013).
- Hydrologic modifications from human activities have both benefited and impacted *O. inaequalis* populations (USFWS 2013).
- While improperly timed grazing can negatively impact the plant and its habitat, research by Marty (2004 and 2005) indicates that livestock grazing plays an important role in maintaining species diversity in vernal pool grasslands through control of invasive species. Direct consumption of *O. inaequalis* by grazers in the winter and early spring may be limited, due to the fact that the majority of the plants have not emerged or are in the aquatic growth stage of the lifecycle. Nonetheless, impacts to *O. inaequalis* plants, as a result of improper grazing regimes, are still recognized as a threat to extant populations (USFWS 2013).
- The Recovery Plan included foraging during grasshopper outbreaks as a potential reason for decline of the species in certain areas. Although grasshoppers have been observed on *O. inaequalis* plants at two localities, this species appears to be only slightly susceptible to grasshopper predation. This characteristic has been attributed to the viscidaromatic (sticky, fragrant) exudate produced by *Orcuttia* species, which may act as an effective deterrent to grasshoppers (USFWS 2013).
- Soil disturbance from overgrazing by cattle may adversely affect *O. inaequalis* indirectly by facilitating invasive plant species (USFWS 2013).
- *O. inaequalis* occurrences on private lands may be threatened by off-road vehicle use (USFWS 2013).
- Vulnerability of *O. inaequalis* from small populations. annual precipitation affects both seed production and seed germination. Therefore the number of individuals that make up a given population of *O. inaequalis* can vary widely from year to year. In fact, some extant localities do not appear during dry years and appear the next year, under more favorable rainfall conditions, with plants numbering in the thousands (USFWS 2013).
- Climate change is also a threat to this species (USFWS 2013).

Five-Year Status Review

On August 7, 2013, the USFWS issued a five-year status review of the San Joaquin Orcutt grass, which resulted in no change in listing status (USFWS 2013). On June 27, 2023, the U.S. Fish and Wildlife Service completed another five-year status review of the San Joaquin Valley Orcutt grass, and concluded that this species' threatened status would remain unchanged (USFWS 2023).

Critical Habitat

Critical habitat was designated for the San Joaquin Orcutt grass on February 10, 2006 (71 FR 7118). The critical habitat designation for *Orcuttia inaequalis* includes six units in Fresno, Madera, Mariposa, Merced, and Tulare counties, California. This species critical habitat encompasses approximately 136, 312 acres (ac) (55,164 hectares (ha)) (71 FR 7118).

- Unit 1: Merced and Mariposa counties, California. From USGS 1:24,000 topographic quadrangles Snelling, Merced Falls, Winton, Yosemite Lake, Haystack Mountain, Indian Gulch, Merced, and Owens Reservoir.

- Unit 2: Merced, Madera, and Mariposa counties, California. From USGS 1:24, 000 topographic quadrangles Owens Reservoir, Plainsburg, Le Grand, and Raynor Creek.
- Unit 3: Madera County, California. (i) Unit 3A: Madera County, California. From USGS 1:24,000 topographic quadrangle Kismet. (ii) Unit 3B: Madera County, California. From USGS 1:24,000 topographic quadrangles Daulton, Little Table Mountain, Gregg, and Lanes Bridge. (iii) Unit 3C: Madera County, California. From USGS 1:24,000 topographic quadrangle Lanes Bridge.
- Unit 4: Fresno County, California. From USGS 1:24,000 topographic quadrangle Friant.
- Unit 5: Madera County, California. (i) Unit 5A: Madera County, California. From USGS 1:24,000 topographic quadrangles North Fork and Millerton Lake East. (ii) Unit 5B: Fresno County, California. From USGS 1:24,000 topographic quadrangles Millerton Lake East and Academy.
- Unit 6: Tulare County, California. (i) Unit 6A: Tulare County, California. From USGS 1:24,000 topographic quadrangle Monson. (ii) Unit 6B: Tulare County, California. From USGS 1:24,000 topographic quadrangle Monson. Unit 6C: Tulare County, California. From USGS 1:24,000 topographic quadrangle Ivanhoe. Unit 6D: Tulare County, California. From USGS 1:24,000 topographic quadrangle Woodlake.

Primary constituent elements (PCEs) are the physical and biological features of critical habitat essential to a species' conservation. The PCEs of *Orcuttia inaequalis* critical habitat consists of two components (71 FR 7118):

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described in paragraph (ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools; and

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the San Joaquin Orcutt grass (USFWS 2005).

Recovery Actions

- The amount of existing suitable habitat across the range has not been determined and the Service does not currently have sufficient information to quantify either the acreage of suitable habitat within each core area or the acreage of protected suitable habitat for *O. inaequalis* (USFWS 2013).

Environmental Baseline

The San Joaquin Orcutt grass and its designated critical habitat occur in the Southern Sierra Foothills Vernal Pool Region, California. Please refer to information above for the environmental baseline.

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Slender Orcutt Grass (*Orcuttia tenuis*)

Listing Status

Slender Orcutt grass was listed as threatened on March 26, 1997 (62 FR 14338). Critical habitat was designated for this species on February 10, 2006 (71 FR 7118).

Life History and Habitat

Slender Orcutt grass is a member of a small tribe (three genera and nine species) of semi-aquatic grasses that are unique among grasses in exhibiting single-cell C4 photosynthesis, which occurs in only 0.003% of known species of C4 flowering plants (Boykin et al. in review). Plants with C4 photosynthesis utilize a more complex biochemical process than most plants (with C3 photosynthesis) in converting CO_2 to energy, which increases photosynthetic efficiency at low CO_2 concentrations (Boykin et al. unpublished manuscript). The species is endemic to California vernal pools. Slender Orcutt grass occurs across a wide range of elevations (27-1,856 m, or 90-5,761 ft), but is associated primarily with vernal pool habitat on Northern Volcanic Ashflow and Northern Volcanic Mudflow substrates. The species is typically associated with larger and/or deeper vernal pools (typically ≥ 30 cm, or 11.8 in. deep) that have relatively long periods of inundation. The plant is also restricted to the deepest portion of the pools (Service 2005). The main habitat requirement for the plant appears to be inundation of sufficient duration and quantity to eliminate most competition and to meet the plant's physiological requirements for prolonged inundation, followed by gradual desiccation (Griggs and Jain 1983, Corbin and Schoolcraft 1990). However, pools that normally retain moisture until the end of summer allow out-competition of slender Orcutt grass by marsh vegetation (*Scirpus* spp., *Typha* spp.) (Griggs and Jain 1983).

Population Status

Disjunct occurrences of the species occur in vernal pools on remnant alluvial fans, high stream terraces, and recent basalt flows from the Modoc Plateau in northeastern California, west to Lake County, and south through the Central Valley to Sacramento County. The plant has also been reported from other natural and artificial seasonal wetlands such as creek terraces, stock ponds, and borrow pits; however, occurrence records suggest that most such locations are altered vernal pool habitats (CNDDB 2006).

Populations of slender Orcutt grass can vary greatly in size from year to year; fluctuations in population size of up to four orders of magnitude have been recorded. The grass germinates even in dry years, but the proportion surviving to maturity varies (Service 2005). Population trends for this species on managed or protected lands appear to be stable over time, although quantitative monitoring has apparently been discontinued at many sites. Ongoing monitoring of these occurrences does show large, inter-annual fluctuations in the number of living plants at many sites, with some years producing no living plants in some locations (C. Lentz in litt. 2006, L. Serpa pers. comm. 2006).

Recent surveys on the Modoc National Forest have located additional occurrences, thereby increasing the number of occurrences within the Modoc Plateau Vernal Pool Region (C. Beyer in litt. 2006a). Few additional occurrences have been discovered in other regions: one new occurrence has been found in the Southeastern Sacramento Valley Region, within Sacramento's urban development boundary. Its size and status are unknown (Sacramento County undated). Most occurrences on private lands were last evaluated in the late 1980s. At this time, the population trends for 61 occurrences are listed as unknown (CNDDB 2006).

Threats

The reduction and fragmentation of habitat due to urban development, flood control projects, landfill projects, highway development, and agricultural land conversion are listed as the primary threats to this species in the 1997 listing rule. Habitat degradation from agricultural and human-related changes to vernal pool hydrology is listed as an additional threat. Consistent with the 1997 rule, the largest continuing threat to this species is land type conversion and urban development along the periphery of urban areas, especially in the Redding and Sacramento areas (Service 2005, C. Martz in litt. 2006). For example, the new occurrence found within Sacramento's urban development boundary is currently threatened by surrounding development (Sacramento County undated). The population of California is expected to increase to 58 million, almost double the 1990 State population, by 2040 (Field et al. 1999). Between 1994 and 2005, the Sacramento FWS office engaged in Section 7 consultations for projects with impacts to approximately 20,250 ha (50,000 ac) of vernal pool habitat, including loss of 10,125 ha (25,000 ac) to residential, commercial, and industrial development (Service 2005). This loss is expected to continue as urban boundaries expand further through high and low terrace formations on the eastern side of the valley.

More subtle threats have the ability to change habitat suitability in natural lands remaining within the developed landscape. For example, loss of vernal pool habitat to residential, commercial, and industrial development can also lead to modification of remaining suitable habitat. Development can result in the loss of hydrological connections that sustain the remnant vernal pools. Vernal pool plants are sensitive to variations in the period of vernal pool inundation (Bauder 2000); populations of slender Orcutt grass could be impacted by such changes. On private lands, numerous pools with slender Orcutt grass occurrences have either been partially filled, or remain on relatively small parcels of lands adjacent to development (CNDDB 2006). Some pools have been partially drained, while others are inundated during longer periods of time due to nearby irrigation or runoff from development (CNDDB 2006).

Changes to vernal pool habitat associated with residential development include facilitation of the introduction of non-native plants to vernal pool habitats (Zedler and Black 2004). Non-native grasses occur commonly in vernal pool complexes and have become a threat to native vernal pool plants through their capacity to change pool hydrology. Exotic grasses maintain dominance at pool edges, sequestering light and soil moisture, promoting thatch build-up, and shortening inundation periods. Although the mechanism responsible for the change in inundation is not documented, reduction in inundation period is thought to be due to increased evapotranspiration at the vernal pools (Marty 2005). In areas near the urban boundary, cattle-grazing is often discontinued in anticipation of land use changes (C. Martz pers. comm.). Cessation of cattle grazing has been found to exacerbate the negative effects of invasive non-native plants on vernal pool inundation period. The change in vernal pool inundation due to loss of grazing is an emerging threat for this species, especially in the Sacramento Valley (C. Lentz in litt. 2006, C. Martz pers. comm.). Vernal pool inundation was reduced by 50-80% in the Southeastern Sacramento Valley when grazing was discontinued (Marty 2005).

The vernal pools of the Modoc Plateau are not threatened by development, but habitat suitability for some populations may be modified by OHV use and the alteration of pools by damming and excavating to provide cattle watering holes (and maintenance of alterations). These activities pose continued threats to individual populations. Numerous pools harboring slender Orcutt grass occurrences in this region have been fenced to exclude grazing and protect occurrences; however, cessation of grazing may have less effect on pool inundation in the Modoc Plateau region (Marty 2005, A. Sanger in litt. 2006, C. Beyer in litt. 2006b).

Suitable habitat for this species may also be modified through changes to vernal pool hydrology at a relatively large scale. Recent research by Rains et al. (2006) has illustrated the manner in which many, if not most, vernal pools located on duripan or claypan in the Central Valley appear to be supported by perched aquifers. In these hydrological features, seasonal surface water and perched groundwater hydrologically connect uplands, vernal pools, and streams at the catchment scale. Perched groundwater discharges from uplands to vernal pools thereby stabilizing the pools, and causing them to remain inundated for longer periods than would be the case if they were recharged only by precipitation. Accordingly, small changes in local land use, such as development of irrigated agriculture or parkland may have considerable impacts on vernal pools, although the degree to which such changes affect pools is poorly understood (Rains et al. 2006).

Loss of suitable habitat has been offset to some extent by the development of conservation banks. Stillwater Plains Conservation Bank within the Northeastern Sacramento Valley Region has created suitable habitat for slender Orcutt grass. However, in the last several years the inflated price of land along the urban front in the Redding area has provided an unexpected threat to preservation of suitable slender Orcutt grass habitat by reducing the land-purchasing capability of conservation and governmental organizations (C. Martz pers. comm.).

Slender Orcutt grass occurrences on conservation banks and small preserves are often subject to the same threats as occurrences on unprotected, fragmented habitat. Disruption of perched aquifers underlying small, protected parcels may impact populations within preserves. In addition, development of offsite banks may not adequately protect the rare landform types associated with specific plant species or meet the functional equivalence of the original wetlands ecosystems (see discussion in Wacker and Kelly 2004). In the Southeastern Sacramento Valley Region, Wacker and Kelly (2004) illustrated that the majority of project site characteristics were replicated at the corresponding mitigation sites. However, when compared at the landscape scale across all development projects, they found that relatively rare pool types, such as Northern Volcanic Mudflow pools, are decreasing while Drainageway pools (pools formed in recent alluvial deposits over other formations, which typically support lower species richness) are becoming more common. The four occurrences of slender Orcutt grass in Sacramento County are found on the high terrace Laguna Formation (Sacramento County undated). High terrace formations generally support larger and deeper (longer lasting) pools (Wacker and Kelly 2004). Although projects have occurred fairly equally on high and low terrace sites in the study area, compensation sites were established disproportionately on low terrace formations (Wacker and Kelly 2004). Such shifts in availability of landform types could have negative consequences for persistence of the grass, although the degree of risk is unknown.

In summary, habitat for slender Orcutt grass continues to be highly fragmented throughout most of its range due to conversion of natural habitat for urban and agricultural uses. This fragmentation results in small, isolated populations of this species in all areas but the Modoc Plateau. Highly fragmented, small populations may be highly susceptible to extirpation due to stochastic events, inbreeding depression, or additional environmental disturbance (Gilpin and Soule 1986; Goodman 1987). If an extirpation event occurs in a population that has been fragmented, the opportunities for natural re-colonization will be greatly reduced due to physical isolation from other source populations. In addition, both protected and unprotected populations in the Central Valley may be increasingly subject to decreased suitability of habitat due to competitive exclusion by either native *Eleocharis* spp. (as grazing is discontinued near urban expansion), invasive non-native plant species such as waxy manna grass (C. Witham pers. comm., C. Martz, CDFG, in litt. 2006), or changes in hydrology of vernal pools (Service 2005, Rains et al. 2006, C. Witham pers. comm.).

Five-Year Status Review

In November 2009, the Service issued a five-year status review of the slender Orcutt grass, which resulted in no change in listing status (Service 2009). On April 2, 2024, the U.S. Fish and Wildlife Service completed another five-year status review of the slender Orcutt grass, and concluded that this species' threatened status would remain unchanged (USFWS 2024).

Critical Habitat

Critical habitat was designated for the slender Orcutt grass on February 10, 2006 (71 FR 7118).

Primary Constituent Elements

The primary constituent elements of critical habitat for *Orcuttia tenuis* (slender Orcutt grass) are the habitat components that provide:

(i) Topographic features characterized by isolated mound and intermound complex within a matrix of surrounding uplands that result in continuously, or intermittently, flowing surface water in the depressional features including swales connecting the pools described in paragraph (ii) of this section, providing for dispersal and promoting hydroperiods of adequate length in the pools; and

(ii) Depressional features including isolated vernal pools with underlying restrictive soil layers that become inundated during winter rains and that continuously hold water or whose soils are saturated for a period long enough to promote germination, flowering, and seed production of predominantly annual native wetland species and typically exclude both native and nonnative upland plant species in all but the driest years. As these features are inundated on a seasonal basis, they do not promote the development of obligate wetland vegetation habitats typical of permanently flooded emergent wetlands.

Recovery Plan Information

On December 15, 2005, the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon was issued, which includes the slender Orcutt grass (Service 2005).

According to the 5-Year Review for this species (Service 2009), Core Recovery Areas include:

- Lake-Napa Vernal Pool Region
- Modoc Plateau Vernal Pool Region
- Northeastern Sacramento Valley Vernal Pool Region
- Northwestern Sacramento Valley Vernal Pool Region
- Southeastern Sacramento Valley Vernal Pool Region

Delisting Criteria

In addition, general delisting criteria and recovery actions (Service 2009) for this species include:

1. Habitat Protection: Accomplish habitat protection that promotes vernal pool ecosystem function sufficient to contribute to population viability of the covered species.

   1A. Suitable vernal pool habitat within each prioritized core area for the species is protected.

   1B. Species localities distributed across the species geographic range and genetic range are protected. Protection of extreme edges of populations protects the genetic differences that occur there.

   1C. Reintroduction and introductions must be carried out and meet success criteria.

   1D. Additional occurrences identified through future site assessments, GIS and other analyses, and status surveys that are determined essential to recovery are protected. Any newly found occurrences may count towards recovery goals if the occurrences are permanently protected, as described in the recovery plan.

1E. Habitat protection results in protection of hydrology essential to vernal pool ecosystem function, and monitoring indicates that hydrology that contributes to population viability has been maintained through at least one multi-year period that includes above average, average, and below average local rainfall as defined above, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

2. Adaptive Habitat Management and Monitoring.

2A. Habitat management and monitoring plans that facilitate maintenance of vernal pool ecosystem function and population viability have been developed and implemented for all habitat protected, as previously discussed in Sections 1 (A-E).

2B. Mechanisms are in place to provide for management in perpetuity and long-term monitoring of habitat protected in Sections 1 (A-E) (e.g., funding, personnel, etc.).

2C. Monitoring indicates that ecosystem function has been maintained in the areas protected under Sections 1 (A-D) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

2D. Seed banking actions have been completed for species that would require it as insurance against risk of stochastic extirpations or that will require reintroductions or introductions to contribute to meeting recovery criteria.

3. Status Surveys.

3A. Status surveys, 5-year status reviews, and population monitoring show populations within each vernal pool region where the species occur are viable (e.g., evidence of reproduction and recruitment) and have been maintained (stable or increasing) for at least one multi-year period that includes above average, average, and below average local rainfall, a multi-year drought, and a minimum of 5 years of post-drought monitoring.

3B. Status surveys, status reviews, and habitat monitoring show that threats identified during and since the listing process have been ameliorated or eliminated. Site-specific threats identified through standardized site assessments and habitat management planning also must be ameliorated or eliminated.

4. Research.

4A. Research actions necessary for recovery and conservation of the covered species have been identified (these are research actions that have not been specifically identified in the recovery actions but for which a process to develop them has been identified). Research actions (both specifically identified in the recovery actions and determined through the process) on species biology and ecology, habitat management and restoration, and methods to eliminate or ameliorate threats have been completed and incorporated into habitat protection, habitat management and monitoring, and species monitoring plans, and refinement of recovery criteria and actions.

4B. Research on genetic structure has been completed (for species where necessary – for reintroduction and introduction, seed banking) and results incorporated into habitat protection plans to ensure that within and among population genetic variation is fully representative by populations protected in the Habitat Protection section of this document, described previously in Sections 1 (A-E).

4C. Research necessary to determine appropriate parameters to measure population viability for each species have been completed.

5. Participation and Outreach.

5A. Recovery Implementation Team is established and functioning to oversee rangewide recovery efforts.

5B. Vernal Pool Regional working groups are established and functioning to oversee regional recovery efforts.

5C. Participation plans for each vernal pool region have been completed and implemented.

5D. Vernal Pool Regional working groups have developed and implemented outreach and incentive programs that develop partnerships contributing to achieving recovery criteria 1-4.

Environmental Baseline

Because the known occurrences of slender Orcutt grass occur entirely within California, the status description above also serves as the baseline for this consultation.

Literature Cited

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Spreading Navarretia (*Navarretia fossalis*)

Listing Status

Spreading navarretia was federally listed at threatened on October 13, 1998, primarily due to habitat destruction and fragmentation (63 FR 54975). Critical habitat was designated on October 18, 2005 (70 FR 60658).

Life History and Habitat

Spreading navarretia, a member of the Polemoniaceae (phlox family), is a low, mostly spreading or ascending annual plant, 4 to 6 inches tall. The leaves are 0.4 to 2 inches long and finely divided into slender spine-tipped lobes. Spreading navarretia depends on the inundation and drying cycles of its habitat for survival. This regime allows for germination and other life history phases of the plant. This annual species germinates from seeds left in the seed bank. Spreading navarretia abundance also varies from year to year depending on precipitation and the inundation/drying time of the vernal pool. This annual variation makes it impossible to obtain an accurate count of the number of individuals in the population because the proportion of standing plants to remaining seeds in the seed bank that makes up the population cannot be measured. Additionally, the occurrences can vary spatially in alkali playa habitat where pools are not in the same place from year to year. After germination, the plant usually flowers in May and June as the vernal pool is devoid of water. The plant then produces fruit, dries out, and senesces in the hot, dry summer months (Service 2009).

Population Status

Spreading navarretia extends from northwestern Los Angeles County to western Riverside County, and coastal San Diego County in California, to San Quintin in northwestern Baja California, Mexico. At the time of listing, 34 populations were known to be extant in the United States, including populations contained in the listing rule and in the recovery plan. Nearly 60 percent of these populations were concentrated at three locations: Otay Mesa in southern San Diego County, alongside the San Jacinto River in western Riverside County, and near Hemet in western Riverside County. At the time of listing, spreading navarretia was documented in less than 300 acres of habitat in the United States (Service 2009). However, since listing, new occurrences of spreading navarretia have been identified, bringing the number of occurrences to 48 (Service 2009).

The listing rule characterizes the size of spreading navarretia populations as highly variable, identifying two locations in Riverside County with 300,000 and 100,000 individuals (Stowe Pool and San Jacinto River, respectively), while most populations contain fewer than 1,000 individuals. At the time of listing, seven sites in Stowe Pool and Salt Creek occurrences contained an estimated 375,500 plants, including 300,000 in Stowe Pool. The highest report for Upper Salt Creek since listing is 10,500. Additional occurrences along the San Jacinto River have been detected since listing. Occurrences along three of the sections of the river were observed to support approximately 63,500 individuals. In 2005, those same three sections were recorded as supporting 361,000 individuals. The changes in abundance of spreading navarretia along the San Jacinto River and at Stowe Pool illustrate the dynamic nature of the seasonally flooded alkali playa habitat, impacts from agriculture, the results of different methodologies for measuring abundance, and recent climatic variation. As such, abundance of standing plants is not a good measure of health for occurrences (Service 2009).

Through conservation, 31 occurrences (63 percent) are considered protected from development, while 14 occurrences have been impacted by development, extirpated, or proposed for development since listing. Further, the largest populations along the San Jacinto River and at the Stowe Road Pool are not conserved (Service 2009).

At listing, spreading navarretia was threatened by development and degradation of vernal pool habitat due to agricultural practices, invasive nonnative plants, and drought conditions and these are still considered threats. Agricultural activities, such as manure dumping (not identified in the listing rule) and discing, are currently affecting some occurrences in Riverside County. The degree to which drier conditions (considered a threat in the listing rule) have caused a rangewide decrease in the abundance of spreading navarretia is unknown. As development surrounds and fragments the remaining habitat, associated effects of human access and disturbance (including off-highway vehicle use, trash and debris dumping, and trespassing) will continue to impact many of the occurrences. These threats continue to affect the existence of spreading navarretia and compromise its potential for recovery (Service 2009).

Critical Habitat

Designated critical habitat occurs in six units in Los Angeles, Riverside, and San Diego counties, California, for a total of approximately 6,720 acres (75 FR 62192). The physical and biological features of designated critical habitat include:

1) Vernal pools (up to 10 acres) and seasonally flooded alkali vernal plains that become inundated by winter rains and hold water or have saturated soils for 2 weeks to 6 months during a year with average rainfall (i.e., years where average rainfall amounts for a particular area are reached during the rainy season (between October and May)). This period of inundation is long enough to promote germination, flowering, and seed production for spreading navarretia and other native species typical of vernal pool and seasonally flooded alkali vernal plain habitat, but not so long that true wetland species inhabit the areas.
2) Areas characterized by mounds, swales, and depressions within a matrix of upland habitat that result in intermittently flowing surface and subsurface water in swales, drainages, and pools described in physical and biological feature 1.
3) Soils found in areas characterized in physical and biological features 1 and 2 that have a clay component or other property that creates an impermeable surface or subsurface layer. These soil types include but are not limited to: CienebaPismo-Caperton soils in Los Angeles County; Domino, Traver, Waukena, Chino, and Willows soils in Riverside County; and Huerhuero, Placentia, Olivenhain, Stockpen, and Redding soils in San Diego County.

Recovery Plan Information

A recovery plan for spreading navarretia and other vernal pool species was released on September 3, 1998 (Service 1998). The delisting criteria include the following:

1) All the existing vernal pools and their watersheds identified in Appendix F and G of the recovery plan should be secured from further loss and degradation in a configuration that maintains habitat function and viability (as determined by prescribed research tasks).
2) Secured vernal pools must be enhanced or restored such that population levels of existing species are stabilized or increased.
3) Population trends must be shown to be stable or increasing for a minimum of 10 consecutive years prior to consideration for reclassification.

Environmental Baseline

Since spreading navarretia and its designated critical habitat occur mostly within California, except for potential locations in Mexico for which we have limited information, the status description above also serves as the baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 1998. Vernal pools of southern California recovery plan. U.S. Fish and Wildlife Service, Portland, Oregon. 113+pp.

Service (U.S. Fish and Wildlife Service). 2009. *Navarretia fossalis* (Spreading navarretia) 5-year review: summary and evaluation. 59 pp.

Thread-leaved Brodiaea (*Brodiaea filifolia*)

Listing Status

Thread-leaved brodiaea was federally listed as threatened on October 13, 1998, due to habitat destruction and modification (63 FR 54975). Critical habitat was designated on February 8, 2011 (76 FR 6848).

Life History and Habitat

Thread-leaved brodiaea is a perennial herb with dark-brown, fibrous-coated corms (underground bulblike storage stem). The flower stalks (scapes) are 8 to 16 inches tall. The flowering period extends from March to June (Service 2009). This species is usually found in herbaceous plant communities such as valley needlegrass grassland, valley sacaton grassland, nonnative grassland, alkali playa, southern interior basalt vernal pools, San Diego mesa hardpan vernal pools, and San Diego mesa claypan vernal pools. It grows in interstitial areas (often narrow bands of habitat surrounded by other vegetation) in association with coastal sage scrub in some locations. These herbaceous communities occur in open areas on clay soils, soil with clay subsurface, or clay lenses within loamy, silty loam, loamy sand, silty deposits with cobbles or alkaline soils; they may range in elevation from 100 feet to 2,500 feet, depending on soil series (Service 2009).

Population Status

The historical range of thread-leaved brodiaea extends from the foothills of the San Gabriel Mountains at Glendora (Los Angeles County), east to Arrowhead Hot Springs in the western foothills of the San Bernardino Mountains (San Bernardino County), and south through eastern Orange and western Riverside counties to Rancho Santa Fe in central coastal San Diego County, California. Currently, there are 68 occurrences, with 23 that are newly identified or confirmed since listing. Two new occurrences are in Riverside County; four are in Orange County; and seven in San Diego County. Additionally, 10 more occurrences have been found on Marine Corps Base Camp Pendleton (Service 2009).

Currently, the largest natural occurrences of thread-leaved brodiaea are on the Santa Rosa Plateau in Riverside County, the San Dimas/Gordon Highlands occurrence in Los Angeles County, the Cristianitos Canyon/Lower Gabino Canyon occurrence in Orange County, and the Rancho Carrillo and Upham occurrences in San Diego County. Although each occurrence on Marine Corps Base Camp Pendleton generally supports fewer than 2,000 plants, the occurrences on the base comprises a significant portion of all the known occurrences of the plant. No accurate estimate of the overall abundance of thread-leaved brodiaea is available currently. There is no comprehensive survey data of all known occurrences and different survey techniques have been used (Service 2009).

The current threats to this species are essentially the same as they were at listing and include urbanization, alteration of hydrological conditions and channelization, discing, unauthorized off-highway vehicle activity, grazing, and nonnative plants. Additional threats since listing include manure dumping and mowing. Development remains the most prominent rangewide threat to thread-leaved brodiaea, though the protective provisions of the Act have had a significant impact relative to addressing this threat through the development of regional habitat conservation plans and section 7 consultations. As habitat continues to be placed into permanent conservation with adaptive management, the threats to thread-leaved brodiaea will be further reduced rangewide; current conservation efforts address approximately 75 percent of occurrences. The second most significant rangewide threat to thread-leaved brodiaea is competition from nonnative plants, which impact at least 15 of the known occurrences. Other threats from unauthorized off-highway vehicle use, grazing, and manure dumping threaten specific occurrences of thread-leaved brodiaea, and while they are not rangewide threats to the species, these threats hinder recovery (Service 2009).

Critical Habitat

Designated critical habitat occurs in 10 units in Los Angeles, San Bernardino, Riverside, Orange, and San Diego counties, California, for a total of approximately 2,947 acres (76 FR 6848). The physical and biological features of designated critical habitat include:

1) Appropriate soil series at a range of elevations and in a variety of plant communities, specifically: (A) Clay soil series of various origins (such as Alo, Altamont, Auld, or Diablo), clay lenses found as unmapped inclusions in other soils series, or loamy soils series underlain by a clay subsoil (such as Fallbrook, Huerhuero, or Las Flores) occurring between the elevations of 100 and 2,500 feet. (B) Soils (such as Cieneba-rock outcrop complex and Ramona family Typic Xerothents soils) altered by hydrothermal activity occurring between the elevations of 1,000 and 2,500 feet. (C) Silty loam soil series underlain by a clay subsoil or caliche that are generally poorly drained, moderately to strongly alkaline, granitic in origin (such as Domino, Grangeville, Traver, Waukena, or Willows) occurring between the elevations of 600 and 1,800 feet. (D) Clay loam soil series (such as Murrieta) underlain by heavy clay loams or clays derived from olivine basalt lava flows occurring between the elevations of 1,700 and 2,500 feet. (E) Sandy loam soils derived from basalt and granodiorite parent materials; deposits of gravel, cobble, and boulders; or hydrologically fractured, weathered granite in intermittent streams and seeps occurring between 1,800 and 2,500 feet.
2) Areas with a natural, generally intact surface and subsurface soil structure, not permanently altered by anthropogenic land use activities (such as deep, repetitive discing, or grading), extending out up to 820 feet from mapped occurrences of thread-leaved brodiaea to provide for space for individual population growth, and space for pollinators.

Environmental Baseline

Since the known occurrences of thread-leaved brodiaea and its designated critical habitat occur entirely within California, the status description above also serves as the baseline for this consultation.

Literature Cited

Service (U.S. Fish and Wildlife Service). 2009. *Brodiaea filifolia* (thread-leaved brodiaea) 5-year review: summary and evaluation. 47 pp.

Other Plant Species (Non-Vernal Pool Plants)

Ben Lomond Spineflower (*Chorizanthe pungens* var. *hartwegiana*)

Listing Status

The Ben Lomond spineflower was federally listed as endangered on February 4, 1994 (Service 1994).

Life History and Habitat

The known populations of Ben Lomond spineflower are restricted in distribution to the Zayante sandhills in Santa Cruz County and found between 295 and 2,000 feet in elevation (California Native Plant Society 2011). This taxon is a short-lived annual species that undergoes large variations in abundance from year to year depending on climatic conditions and other factors.

Population Status

Ben Lomond spineflower is not restricted to sandy soils due to any chemical, physical, or biological requirement, but is intolerant of shade and unable to compete for light with other species that commonly occur on the non-sandy soils (Service 1998). We cannot draw any conclusions about population trends for this species because there is very little historical or recent survey data that contains a record of the number of individuals. Currently, monitoring is only taking place at Quail Hollow Quarry (Service 2012). The primary threats described for this species are habitat destruction and habitat conversion. Habitat conversion due to fire exclusion and human disturbance continues to be a major concern when examining suitability of habitat and ecosystem dynamics for continued survival of this species.

Critical Habitat

Critical habitat has not been designated.

Recovery Plan Information

The Recovery Plan for Insect and Plant Taxa from the Santa Cruz Mountains in California (recovery plan) (Service 1998) outlines downlisting and delisting criteria for the Mount Hermon June beetle, Zayante band-winged grasshopper, Ben Lomond wallflower, and Ben Lomond spineflower. Definitive delisting criteria will be developed for each species as more information becomes available on biology, range, and distribution through research and surveys. When the downlisting criteria have been met for a species, the species can be considered for delisting if threats are reduced or eliminated so that populations are capable of persisting without significant human intervention or perpetual endowments are secured for management necessary to maintain the continued existence of the species.

<u>*Recovery Actions*</u>

- Protect habitat for Santa Cruz Mountains species on private land through Habitat Conservation Plans and landowner agreements;
- Manage habitat for Santa Cruz Mountains species;
- Conduct research on the life history, ecology, and population dynamics of these species that will contribute to appropriate management strategies;
- Locate additional habitat/populations within the historic range of the species;
- Develop and implement a public outreach program; and
- Evaluate progress of recovery effectiveness of management and recovery actions and revise management plans.

Environmental Baseline

The species only occurs within the State of California, please refer to the above information regarding the species environmental baseline.

Literature Cited

[Service] U.S. Fish and Wildlife Service. 1994. Endangered and threatened wildlife and plants; endangered status for three plants and threatened status for one plant from sandy and sedimentary soils of Central Coastal California. Federal Register 59:5499-5510.

[Service] U.S. Fish and Wildlife Service. 1998. Recovery plan for insect and plant taxa from the Santa Cruz Mountains in California. Portland, Oregon. 83 pp.

California Seablite (*Suaeda californica*)

Listing Status

Suaeda californica was designated as federally endangered on December 15, 1994 (59 FR 64613). It occurred historically in high tidal marsh in portions of San Francisco Bay, where it became nearly extinct because of habitat loss (Service 2013).

Life History and Habitat

Suaeda californica occupies a narrow zone at the upper edge of tidal marsh, and prefers coarse marsh sediments or sheltered estuarine beaches. It requires well-drained marsh substrates, primarily sandy wave-built berms or ridges along marsh banks, and estuarine beaches. Because its habitat is naturally prone to destruction by wave erosion, it requires widespread populations in diverse environments over large areas to enable it to recolonize by seed after populations are destroyed by storms.

Population Status

Due to several reintroductions between 1999 and 2008, it is currently known from three sites in the San Francisco Bay and scattered locations along the shoreline of Morro Bay, San Luis Obispo County. It is threatened in Morro Bay by shoreline development, storm erosion, and interference with seedling regeneration caused by invasive nonnative vegetation (mostly *Carpobrotus edulis* [iceplant]). Artificial stabilization of sandy shores, or other static modification of suitable estuarine shorelines, threatens the resilience of its population in Morro Bay, and could constrain its recovery in San Francisco Bay. In both locations, it is threatened with the long-term but severe threat of sea level rise in the face of limited opportunities for landward migration of habitat (Service 2013).

Critical Habitat

Critical habitat has not been designated for this species.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The species only occurs within the State of California, please refer to the above information regarding the species environmental baseline.

Literature Cited

[Service] U.S. Fish and Wildlife Service. 2013. Recovery Plan for Tidal Marsh Ecosystems of Northern and Central California. Region 8, U.S. Fish and Wildlife Service, Sacramento, California.

La Graciosa Thistle (*Cirsium scariosum var. loncholepis*) and its Critical Habitat

Listing Status

La Graciosa thistle was listed as endangered on March 20, 2000 (65 Federal Register 14888). The Service designated critical habitat for La Graciosa thistle on March 17, 2004 (69 FR 12553) and published a revised critical habitat designation on November 3, 2009 (74 FR 56978).

Life History and Habitat

Dune swales develop behind the foredunes in areas where wind moves sand to such an extent that a depression forms and intersects the water table (creating small wetlands and back dune lakes). The largest coastal dune system in California, the Guadalupe dune complex covers approximately 18 square miles (47 square kilometers) extending about 2 miles (3.2 kilometers) inland from the coast. The species needs intact wetland habitats with water on or near the surface across the landscape. La Graciosa thistle exists as groups of individuals in wetland habitats in an arid and semi-arid landscape. The plants inhabit the margins of wetlands (swales, lakes, ponds, freshwater marshes, streams, rivers, seeps). Many of the wetlands in the sand dune complexes occur where the groundwater table is at or near the surface and the water levels rise and fall naturally with rainfall.

Population Status

La Graciosa thistle is currently restricted to back dune and coastal wetlands of southern San Luis Obispo County and northern Santa Barbara County. The majority of the extant populations of La Graciosa thistle occur in wetlands associated with the Guadalupe dune complex; these include the freshwater wetlands of the Santa Maria River mouth and wetlands found in dune swales and dune lakes north of the river. There are currently 23 known occurrences of La Graciosa thistle. Of these, eight occurrences are currently known to be extant, (which includes a new occurrence established by outplanting), 15 occurrences are likely extirpated (USFWS 2020, entire). The primary threats to La Graciosa thistle are the following: (1) reduced water/lack of water, with groundwater decline as the likely major cause, along with hydrological alteration and climate change, including severe drought and increased temperatures, and (2) flooding resulting from hydrological alteration (USFWS 2020, p. 12). The groundwater decline appears to result primarily from extraction for urban, agricultural and industrial uses, and it is exacerbated by drought and climate change.

Critical Habitat

A total of 24,103 acres (as 6 units) were designated as critical habitat for the La Graciosa thistle in 2 California counties (San Luis Obispo and Santa Barbara). A detailed discussion of the methods used in designating critical habitat can be found in the final rule. All of the areas of critical habitat for the La Graciosa thistle are within the species' historical geographic range and contain PCEs to support at least one of the species' essential life history functions. Based on the current knowledge of the life history, biology, and ecology of the La Graciosa thistle, the Service determined that the PCEs of La Graciosa thistle critical habitat consist of:

1. Mesic areas associated with margins of dune swales, dune lakes, marshes, and estuaries that are associated with dynamic (changing) dune systems including the Santa Maria Valley Dune Complex and Santa Ynez Valley Dune Complex, and margins of dynamic riparian systems including the Santa Maria and Santa Ynez Rivers and Orcutt/Solomon and San Antonio Creeks, and freshwater seeps;
2. Associated plant communities that include Central dune scrub, coastal dune, coastal scrub, freshwater seep, coastal and valley freshwater marsh and fen, riparian scrub (e.g., mule fat scrub, willow scrub), oak woodland, intermittent streams, and other wetland communities;
3. Soils with a sandy component including but not limited to dune sands; and

4. Features that allow dispersal and connectivity between populations.

The balance of the species' critical habitat has been, and continues to be, disturbed by off-road vehicle activity, recreation, oil exploration, livestock grazing, agriculture, and installation and maintenance of roads and other transportation corridors.

Recovery Plan Information

A recovery plan for the species was published in 2021 (Service 2021). The primary strategy for recovery of La Graciosa thistle is to first implement a series of actions to prevent extinction of the species. These near-term actions focus efforts at the remaining extant occurrences to prevent local extirpations by restoring habitat and minimizing the threats at each of these sites. Then a series of longer-term actions will be implemented for La Graciosa thistle recovery that are intended to fill knowledge gaps, streamline management and monitoring techniques, and systematically re-establish the species at several extirpated occurrences and potentially introduce the species to new sites across the historical range.

Environmental Baseline

The species only occurs within the State of California, please refer to the above information regarding the species environmental baseline.

Literature Cited

[Service] U.S. Fish and Wildlife Service. 2020. Species Status Assessment for La Graciosa thistle (*Cirsium scariosum var. loncholepis* [*Cirsium loncholepis*], Asteraceae). Ventura Fish and Wildlife Office, Ventura, California.

[Service] U.S. Fish and Wildlife Service. 2021. La Graciosa thistle (*Cirsium scariosum var. loncholepis*) Recovery Plan. U.S. Fish and Wildlife Service, Ventura, California.

Marsh Sandwort (Arenaria paludicola)

Listing Status

Marsh sandwort was listed as endangered on August 3, 1993 (58 FR 41378). At the time of listing, *Arenaria paludicola* was known from a single natural occurrence within Black Lake Canyon, in southwestern San Luis Obispo County. Its historic range is thought to extend along the Pacific Coast from Washington state south throughout Southern California.

Life History and Habitat

Arenaria paludicola is an herbaceous perennial in the Caryophyllaceae (pink family). This species typically blooms from May through August.

Population Status

A 5-Year Review for the species was conducted in 2008 and *Arenaria paludicola* was still known only from a single wild occurrence. However, this 2008 occurrence was different than the location known at the time of listing, which had become extirpated to spite several unsuccessful three outplanting attempts. The newly discovered occurrence was found at Oso Flaco Lake, but was also in a state of decline. In addition to plants at this site, another successful outplanting was established at the Sweet Springs Nature Preserve, managed and owned by the Morro Coast Audubon Society. Since that time, several other outplanting efforts have taken place and occurrences have been established at sites in Marin and Santa Cruz counties. The main threats to the species include habitat modification from invasive species, climate change and resultant sea level rise and stochastic (random and unpredictable) extirpation and extinction. (Service 2019).

Critical Habitat

Critical habitat has not been designated for this species.

Recovery Plan Information

A recovery plan was published for the species in 1998 (Service 1998), with an amendment to the recovery plan published in 2019 (Service 2019). The main objective for the long-term management and recovery of *Arenaria paludicola* is to secure viable, self-sustaining populations of the species in its natural habitat. The objective is to reclassify it from endangered to threatened status, and ultimately to delist completely. Preliminary criteria for downlisting are: 1) new plants are established so that there are at least 5 populations of at least 500 individuals each, 2) some of these populations occur in permanently protected habitats in Black Lake Canyon and the Dune Lakes area, 3) some of the populations must be in other areas of suitable habitat within the species historical range in the United States, and 4) the populations remain viable for at least 5 years. Delisting may be warranted when the downlisting criteria have been met and the species exhibits sufficient resiliency, redundancy, and representation to support long-term viability. For this species, the historical distribution of colonies within four geographically separated areas (Puget Sound in Washington State, San Francisco Bay to Santa Cruz, central coastal region (Santa Barbara County to Los Angeles County), and San Bernardino County) is important for its resiliency, redundancy, and representation.

Environmental Baseline

The species only occurs within the State of California, please refer to the above information regarding the species environmental baseline.

Literature Cited

[Service] U.S. Fish and Wildlife Service. 1998. Recovery Plan for Marsh Sandwort (*Arenaria paludicola*) and Gambel's Watercress (*Rorippa gambelii*). U.S. Fish and Wildlife Service, Portland, Oregon. 50 pp. + appendices.

[Service] U.S. Fish and Wildlife Service. 2019. Amendment 1 to Recovery Plan for Marsh Sandwort (*Arenaria paludicola*) and Gambel's Watercress (*Rorippa gambelii*). U.S. Fish and Wildlife Service, Pacific Southwest Region, Ventura, California.

Salt Marsh Bird's Beak (*Chloropyron maritimum* subsp. *maritimum*)

Listing Status

Salt marsh bird's beak was federally listed as endangered on September 28, 1978, primarily due to habitat modification of coastal salt marshes (43 FR 44810).

Life History and Habitat

Salt marsh bird's-beak is a hemiparasitic annual plant found in disjunct coastal salt marshes of southern and central California and adjacent northern Baja California, Mexico. Specimens are branched and may be up to 16 inches tall with numerous flowers arranged on flower stalks termed spikes. The flowering period is between May and October. Each flower may produce 10-40 seeds. Seeds germinate generally over a three-to-five-week period in March or April and may be followed by a high mortality rate after 4 to 6 weeks. Individual plants senesce in late July after flowering and setting seed. The flowers are self-compatible and are pollinated by various bees including *Bombus pennsylvanicus sonorous*, *Anthidium edwardsii*, and *Melissodes tepida timberlakei* (Service 2009).

Population Status

Salt marsh bird's beak is currently extant at nine coastal marsh complexes across the species' range, including seven marsh complexes in the United States [Morro Bay, Carpinteria Salt Marsh, Ormond Beach/Mugu Lagoon, Upper Newport Bay, San Diego River Mouth, San Diego Bay (including Sweetwater Marsh) and Tijuana Estuary], and two marsh complexes in Baja California, Mexico (Estero Punta Banda and Bahía de San Quintín). One new population has been established since the 2008 5-year review, at the San Diego River Mouth. Conservation efforts have occurred and are ongoing throughout the subspecies' range, including work to introduce salt marsh bird's beak at Magnolia Marsh, within the Huntington Beach Wetlands (Service 2020).

Historically, habitat loss due to development and urbanization was a substantial threat to salt marsh bird's beak. While urbanization is not currently a direct threat, development surrounding coastal wetlands interacts with other threats, including altered hydrology and climate change, to reduce the amount of space available for marsh transgression (Service 2020).

Despite signs of larval moth granivory, Parsons and Zedler (1997) reported that granivory did not significantly affect the number of salt marsh bird's beak seeds produced in two years of study. However, in San Diego County, biologists have noted high levels of seed predation at salt marsh bird's beak occurrences, especially at drier locations. Overall, the magnitude of this threat is unknown (Service 2020).

Nonnative *Limonium* has emerged as a moderate threat to salt marsh bird's beak and occurs at five of the extant marshes. In addition, models of wetland accretion and sea level rise project considerable losses of high marsh habitat in the 21st century (Service 2020).

Recovery Plan Information

The Service completed a recovery plan for salt marsh bird's beak on December 6, 1985 (Service 1985). The 1985 Recovery Plan didn't include threats-based criteria, and in the 2009 5-year review, we recommended that a recovery plan revision include assessments of sea-level rise. In addition, since completion of the recovery plan, nonnative *Limonium* has emerged as a threat. Regardless, the downlisting criteria include the following, which have been partially met:

1) 15 acres of secured and protected high marsh habitat at appropriate elevations is required at a minimum of eight marshes for a period of at least 5 consecutive years.

2) 20 acres of secured, protected, and managed high marsh habitat at appropriate elevations is required at each of the 12 major marshes within the historical range of the plant for a period of 10 consecutive years.

As mentioned above, salt marsh bird's beak is present at nine coastal marsh complexes across its range (seven in the United States, and two in Mexico), not counting a reestablishment effort at Huntington Beach Wetlands. At least 15 acres of high marsh habitat is conserved within seven of the nine marshes (all except Estero Punta Banda and Bahía de San Quintín, where the amount of conserved habitat is unknown). At seven of nine marshes (all except Estero Punta Banda and Bahía de San Quintín), salt marsh bird's beak has been continuously present for at least 5 years, although plant abundance fluctuates annually. The new population at the San Diego River Mouth has been continuously present since at least 2014. However, only seven of nine occupied marshes contain at least 15 acres of high marsh habitat, and we don't have marsh acreage estimates for marshes in Mexico (Service 2020).

Environmental Baseline

Salt marsh bird's beak occurs primarily in California, but also occurs in Mexico. However, we have limited information regarding this species in Mexico, as described above. Thus, the status description above also serves as the baseline for this consultation.

Literature Cited

Parsons, L.S. and J.B. Zedler. 1997. Factors affecting the reestablishment of an endangered annual plant at a California salt marsh. Ecological Applications 7(1):253-267.

Service (U.S. Fish and Wildlife Service). 1985. Salt marsh bird's-beak (*Cordylanthus maritimus* subsp. *maritimus*) recovery plan. 100 pp.

Service (U.S. Fish and Wildlife Service). 2009. *Chloropyron maritimmum* subsp. *maritimum* (*Cordylanthus maritimum* subsp. *maritimus*) (salt marsh bird's-beak). 5-year review: summary and evaluation. Carlsbad Fish and Wildlife Office. 38 pp.

Service (U.S. Fish and Wildlife Service). 2020. *Chloropyron maritimmum* subsp. *maritimum* (*Cordylanthus maritimum* subsp. *maritimus*) (salt marsh bird's-beak). 5-year review: summary and evaluation. Carlsbad Fish and Wildlife Office. 20 pp.

Ventura Marsh Milk-vetch (*Astragalus pycnostachyus* var. *lanosissimus*) and its Critical Habitat

Listing Status

The final rule listing the plant as endangered was published on May 21, 2001 (66 *FR* 27901). Critical habitat for the species was designated on May 20, 2004 (69 *FR* 29081).

Life History and Habitat

The best description we have of the habitat of Ventura marsh milk-vetch is from Wilken and Wardlaw (2001) who concluded that the species occurs in low elevation coastal dune-swale areas, where freshwater levels (in the form of saturated soils or groundwater) are high enough to reach the roots of the plants. Sometimes, high groundwater is shown by the presence of water in sloughs or coastal creeks, but more typically evidence for freshwater availability is seen in the presence of native, freshwater-dependent plants, such as willows (*Salix* spp.), cattails (*Typha* spp.), mulefat, and others. The soils associated with Ventura marsh milk-vetch are well-drained, yet contain a mix of sand and clay. Because of the freshwater influence, the soils do not exhibit a white crust that would indicate saline or alkaline conditions.

Population Status

Four populations (three introduced and the rediscovered population) currently contain reproductive individuals of *Astragalus pycnostachyus var. lanosissimus*. The rediscovered population and an associated introduced population are actively managed through regulatory requirement. A third population contains a single individual after several years of no individuals being observed and a fourth population was introduced in 2019 at a newly developed restoration site. Two additional populations have no reproductive adults, but house a viable seedbank and suitable habitat that could support reproductive adults. Three other populations have no reproductive adults, and habitat conditions that are not likely to support seed germination and seedling survival to reproductive age. Those three populations are considered to be functionally extirpated, meaning that conditions do not currently exist, and are not expected to exist in the future, that would support the species. Between the six extant populations, two have low resiliency, two have moderate resiliency, and two have high resiliency. Populations with low resiliency have poor habitat conditions with less than 10 individuals and are very susceptible to stochastic events. Populations with moderate resiliency have moderate quality habitat and greater than 10 individuals with an assumed adequate seed bank. Populations with high resiliency have high quality habitat, greater than 100 individuals, and an assumed seed bank. Populations with high quality habitat are generally supported by active management. The reliance on active management suggests that these populations are conservation-reliant. Representation, adaptive capacity, was found to be low because all introduced and existing populations are derived from a single source population. Redundancy, the ability to withstand catastrophic events, was also found to be low because of the low number of populations across a small geographic extent (Service 2020).

Critical Habitat

Approximately 420 acres (170 hectares) of land fall within the boundaries of the critical habitat designation. The designated critical habitat is located in Santa Barbara and Ventura counties, California. Based on the best available information from the only extant site of the species, the primary constituent elements of critical habitat for *Astragalus pycnostachyus var. lanosissimus*. consist of, but are not limited to: (1) Vegetation cover of at least 50 percent but not exceeding 75 percent, consisting primarily of known associated native species, including but *not limited to, Baccharis salicifolia, Baccharis pilularis, Salix lasiolepis, Lotus scoparius (deerweed), and Ericameria ericoides* (coast goldenbush); (2) Low densities of nonnative annual plants and shrubs; (3) The presence of a high water table, either fresh or brackish, as evidenced by the presence of channels, sloughs, or depressions that may support stands of *Salix* *lasiolepis*, *Typha spp*., and *Scirpus spp.* (cattail); (4) Soils that are fine-grained, composed primarily of sand with some clay and silt, yet are well-drained; and (5) Soils that do not exhibit a white crystalline crust that would indicate saline or alkaline conditions.

Determining what constitutes habitat for Astragalus *pycnostachyus var. lanosissimus* is difficult because there is only one extant population, and the site has been altered by soil dumping and oil waste disposal. Also, the historical collections did not fully document the habitat where the plants were found.

Recovery Plan Information

A recovery plan has not been developed for this species.

Environmental Baseline

The species only occurs within the State of California, please refer to the information above regarding the species environmental baseline.

Literature Cited

[Service] U.S. Fish and Wildlife Service. 2020. Ventura Marsh Milk-Vetch (*Astragalus pycnostachyus var. lanosissimus*) Special Status Assessment. U.S. Fish and Wildlife Service, Ventura, California.

Wilken, D. and T. Wardlaw. 2001. Ecological and Life History Characteristics of Ventura Marsh Milkvetch (*Astragalus pycnostachyus* var. *lanosissimus*) and their Implications for Recovery. Prepared for California Department of Fish and Game, South Coast Region, San Diego, California. Funded by U.S. Fish and Wildlife Service Section 6 Program, Contract No. P995002. 55pp.