Appendix A
Covered Species Accounts

A.1 Palmate-Bracted Bird's-Beak (Chloropyron palmatum)

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A.1.1 Listing Status

Federal: Endangered.

State: Endangered.

California Native Plant Society (CNPS) California Rare Plant Rank: 1B.1; 1B: Rare, threatened, or endangered in California and elsewhere. 0.1: Seriously endangered in California.

Recovery Plan: Palmate-bracted bird's-beak is included in the *Recovery Plan for Upland Species of the San Joaquin Valley, California* (U.S. Fish and Wildlife Service [USFWS] 1998).

A.1.2 Species Description and Life History

Palmate-bracted bird's-beak is a winter germinating, highly branched, herbaceous annual plant in the snapdragon family (Scrophulariaceae) that grows from 10 to 30 centimeters (cm) (4 to 12 inches) tall (Calflora 2008; Chuang and Heckard 1973; Hickman 1993). Formerly, it was classified as the morphologically and ecologically distinct subgenus *Hemistegia* of the genus *Cordylanthus* (Chuang and Heckard 1973) but has recently been assigned to a new family and genus as Orobanchaceae: *Chloropyron palmatum* (Tank et al. 2009). All members of *Chloropyron* develop mucilage containing cells in their leaf tissue, are covered with glandular salt-excreting hairs, and grow in saline soils (Chuang and Heckard 1986). Furthermore, all members of the genus are hemiparasitic and can obtain water and nutrients from the roots of other plants (Chuang and Heckard 1971).

Adult plants begin flowering in late May and continue flowering as late as October (L. C. Lee and Associates, and Center for Conservation Biology 2002). While palmate-bracted bird's-beak has a mixed mating system, it requires an insect pollinator to transfer pollen between its male and female reproductive structures in order to set viable seed. The primary pollinators at the Springtown population, near Livermore, Alameda County, are bumblebees (*Bombus vosnesenskii* and *B. californicus*) early in the season and small native bees (*Halictus tripartitus*, *Lasioglossum* [*Dialectus*], and *Lasioglossum* [*Evylaeus*]) later in the season (L. C. Lee and Associates, and Center for Conservation Biology 2002; Saul-Gershenz 2004). No pollinator studies have been conducted for any other populations. Pollinators are critically important for seed set. A study at the Springtown site found that *Lasioglossum* native bee species, which nest in bare soil areas adjacent to palmate-bracted bird's-beak plants, were particularly important, as 96 percent of the bees visiting palmate-bracted bird's-beak from June through July were of this genus (L. C. Lee and Associates, and Center for Conservation Biology 2002; Saul-Gershenz 2004). Those same species of small native bees also utilized nectar and pollen from common spikeweed (L. C. Lee and Associates, and Center for Conservation Biology 2002). Studies of the important pollinators of crop plants in Yolo County have found that populations of these same species of bees require bare ground and rodent burrows for nest sites and that the intensification of agriculture is eliminating their nesting habitat (Kremen 2001; Kremen et al. 2002a, 2002b; Kremen et al. 2004). Additionally, a shortage of pollinators has been reported in California as a result, at least partly, from the infestation of honeybees with the parasitic mite, *Varroa destructor* (Sousa 2005).

The timing of palmate-bracted bird's-beak seed germination has not been studied, but Fleishman et al. (1994) stated that the seed germinates in January and February. Observations that the seed can float for up to three weeks (Showers 1990) and that individuals are less densely aggregated during years of overland flows than during years of no overland flows (Showers 1988) also suggest that germination occurs during the winter months. Germination of previously buried seed may also be an important factor in the distribution and density of individuals in a population. While no studies have been conducted to determine the germination characteristics of seed under field conditions, seeds can remain viable for at least three years under laboratory conditions (Center for Conservation Biology 1994).

A.1.3 Habitat Requirements and Ecology

This species is restricted to seasonally flooded, saline-alkali soils in lowland plains and basins at elevations of less than 155 meters (500 feet) (USFWS 1998). Small differences in soil topography are critical for seedling establishment, as seedlings establish on banks and sides of raised irrigation ditches and on small berms in areas subject to overland flows (Showers 1988). Extensive soil tests across mound and swale topography at the Springtown population have shown that soil salt concentrations are generally highest in the bottoms of swales and lowest on the tops of mounds (Coats et al. 1988, 1989, 1993). At Springtown, palmate-bracted bird's-beak was found to occur primarily on soils with intermediate salt content along the sides of the swales. The authors concluded that it was generally excluded from the scalds in the swales due to high soil salt content, and it was excluded from the tops of the mounds due to competition from exotic annual grasses (Coats et al. 1988, 1989, 1993). The descriptions of the Woodland population suggest that it also occurs on the sides of small topographic features and that the plants are shaded by dense populations of exotic annual grasses (Foothill Associates 2002; Showers 1988).

The extant population in the Plan Area is located southeast of the City of Woodland in a heavily human-impacted area of what historically was alkaline sink adapted vegetation occurring along both sides of Willow Slough and above the Yolo Basin (U.S. Bureau of Soils 1909a, 1909b; Mann et al. 1911). The hydrology, salts, and clay soils that created and maintained the alkaline sink vegetation were deposited when floodwaters from Putah Creek flowed northward from the area near the city of Davis and emptied into Willow Slough. That flow was supplemented when the combined floodwaters of Putah Creek, Cache Creek, and all of the drainages of the Blue Ridge filled the Cache/Putah Basin, drained eastward through a gap in the Plainfield Ridge, and flowed into the Yolo Basin through Willow Slough (Graymer et al. 2002).

Laguna de Santos Callé, as Willow Slough was previously known, was a unique perennial stream (Eliason 1850; Anonymous 1870) that during the dry season originated from a series of pond-like springs approximately 9 miles southwest of Woodland on the eastern edge of the Plainfield Ridge. As the slough approached the area of Merritt, south of Woodland, it transformed into a 2.5-mile-long, gravel bottomed, linear lake, with an average width of 150 feet and a maximum depth of 75 feet. Approximately 1 mile east of County Road 103, the stream flowing from the lake branched as it dropped over the edge of the alluvial deposits into the Yolo Basin, where it flowed another 2.5 miles northeastward until it emptied into a tule marsh. This perennial stream would have created a very shallow saline water table along Willow Slough that is comparable to the water table along Altamont Creek, which created and maintained the alkaline sink at Springtown. Recent studies show a localized trough in the underlying Tehama formation under this section of Willow Slough and a localized area of shallow groundwater (Wood Rodgers 2004; Lundorff and Scalmanini 2004). Large floods from Cache Creek and Putah Creek have flowed through Willow Slough as recently as 1942, but gravel mining in Cache Creek, dam building on both Cache and Putah Creeks, and the construction of the Willow Slough Bypass have drastically altered the hydrology, salt budgets, and clay deposition patterns in the area of the alkali sink vegetation. Aerial photographs show that all of the alkaline sink vegetation was either converted into rice fields or ditched for drainage, except for a single pool-meadow complex immediately along Willow Slough (U.S. Department of Agriculture [USDA] 1952). That pool has been disked multiple times (Showers 1990, 1996) but the southeastern upper margin of that pool still supports the largest number of plants in the area (Center for Natural Lands Management 2012). Given the intensity and extent of the agricultural impacts to the entire alkali sink area and the irreversible changes in hydrology, the area where palmate-bracted bird's-beak does not currently support alkali sink vegetation, and it would be very difficult to replicate the natural hydrological regimes that would allow that type of vegetation to be successfully restored in the area. However, the historical aerial photographs show that the disked pool-meadow complex did receive extensive amounts of supplemental summer water through ditches draining the upstream rice fields, so it may be possible to restore the appropriate hydrology artificially.

Monitoring studies have documented that populations of palmate-bracted bird's-beak experience significant mortality between early spring and early summer, and then low mortality rates through September (Center for Conservation Biology 1992; Fleishman et al. 1994; Cypher 1998). A positive correlation between high mortality rates and high seedling densities has been demonstrated at some research locations. However, because these data were obtained from field surveys where seedling density was not manipulated, density-independent causes of seedling mortality cannot be ruled out. Alternative explanations for high mortality rates include lack of appropriate hosts, drought stress, and competition with introduced annual grasses. Finally, there are no data describing the soil moisture requirements of palmate-bracted bird's-beak during the period of maximum mortality in spring, but studies have found that plants grow where they have access to adequate levels of soil moisture during the summer rainless period.

According to current data on the species, only perennial plants, such as saltgrass (*Distichlis spicata*), Mojave red sage (*Kochia californica*), and Torrey seepweed (*Suada moquinii*), are assumed to function as appropriate host plants for palmate-bracted bird's-beak (Coats et al. 1988; Cypher 1998; EIP Associates 1998). However, in a greenhouse host-preference experiment, Chuang and Heckard (1971) observed that palmate-bracted bird's-beak was vigorous and produced many flowers when grown with common sunflower (*Helianthus annuus*), which is a summer-flowering annual. This finding suggests that common spikeweed, a summer- and fall-flowering annual plant in the same plant family as common sunflower, and which is closely associated with palmate-bracted bird's-beak in its natural habitat, may be a suitable host. Recent research indicates that alkali heath (*Frankenia salina*) is the most important host plant for this species (Cypher 2015). Because the roots of older perennials become increasingly lignified (woody) and resistant to parasitism, age and spatial distribution of the roots may also contribute to the suitability of a potential host plant for palmate-bracted bird's-beak parasitism (see Marvier and Smith 1997).

A.1.4 Species Distribution and Population Trends

A.1.4.1 Distribution

Palmate-bracted bird's-beak is endemic to the west side of the Sacramento Valley, the north side of the Sacramento National Wildlife Refuge (NWR) Complex, the San Joaquin Valley, and the Springtown area of the Livermore Valley. This species is currently known to exist at six locations outside of the Plan Area: Delevan NWR, Sacramento NWR (established from seed collected at the Delevan NWR), Colusa NWR, the Springtown area, western Madera County, and the combined Alkali Sink Ecological Reserve and Mendota Wildlife Management Area (USFWS 1998).

Very little information exists concerning the historical distribution of palmate-bracted bird's-beak in the Plan Area prior to extensive habitat conversion. The documented locations in the Plan Area consist of an extirpated population that was located northeast of the city of Woodland near the Cache Creek Settling Basin and an extant population located southeast of Woodland (California Natural Diversity Database [CNDDB] 2012; Center for Natural Lands Management 2012; Crampton 1979; Dean 2009). Within the last 25 years, the species has been observed in areas adjacent to the Woodland population in an alkali playa/meadow (Crampton 1979) and on Pescadero silty clay, saline-alkali, and Willows clay soil types (Showers 1988, 1996; EIP Associates 1998; Foothill Associates 2002).

Individuals in the existing Woodland population are generally found on small topographic features such as old irrigation checks, banks of shallow ditches, along the shoreline of a pond, and along the upper margin of a vernal pool. The entire population is limited to Pescadero silty clay, saline-alkali, and Willows clay soil types (Andrews 1970; Showers 1988, 1996; EIP Associates 1998).

A.1.4.2 Population Trends

Little is known about regional population trends of palmate-bracted bird's-beak. The conversion of land to farming and development is resulting in declines because of the destruction of extensive areas of potential habitat in the Sacramento and San Joaquin Valleys (USFWS 1998). However, populations are known to fluctuate. For instance, populations of palmate-bracted bird's-beak in the central San Joaquin Valley, in areas such as Mendota, have fluctuated between 0 and 800 flowering individuals from 1987 to 1993 (Fleishman et al. 2001).

The Colusa, Delevan, and Springtown populations appear to be robust with large populations of between 10,000 and 100,000 flowering individuals in 1991 and 1992, while the Mendota population is small and has fluctuated between 0 and 800 flowering individuals from 1987 to 1993 (Fleishman et al. 2001). Between 1983 and 1990, the Woodland population was restricted to a single property that is known as the City of Woodland Preserve. The size of this population ranged from 200 to 1,400 flowering individuals (EIP Associates 1990). In 1996 and 1998, special-status species surveys of the area discovered additional individuals on the adjoining Woodland Regional Park, Brauner, and Maupin properties (Showers 1996; EIP Associates 1998, Center for Natural Lands Management 2012, Dean 2009).

A.1.5 Threats to the Species

Natural threats to palmate-bracted bird's-beak populations include potential lack of appropriate hosts and pollinators, and competition with introduced annual grasses such as annual ryegrass

(Dawson et al. 2007). A number of specific threats to the species were identified in the 1998 recovery plan but only urban expansion, altered hydrology, and limited genetic variation were identified as threats to the Woodland population (USFWS 1998). More recently, the Woodland site has been extensively invaded by annual ryegrass, which poses a severe threat to the species at this site (M. Showers pers. comm.)

Finally, as previously mentioned, studies of the important pollinators of crop plants in Yolo County have found that intensification of agriculture is eliminating the nesting habitat of native bees, upon which the palmate-bracted bird's-beak depends for pollination (Kremen et al. 2001, 2002a, 2002b, 2004). Additionally, a shortage of pollinators has been reported in California as a result, at least partly, from the infestation of honeybees with the parasitic mite, *Varroa destructor* (Sousa 2005).

A.1.6 Species Habitat Model and Location Data

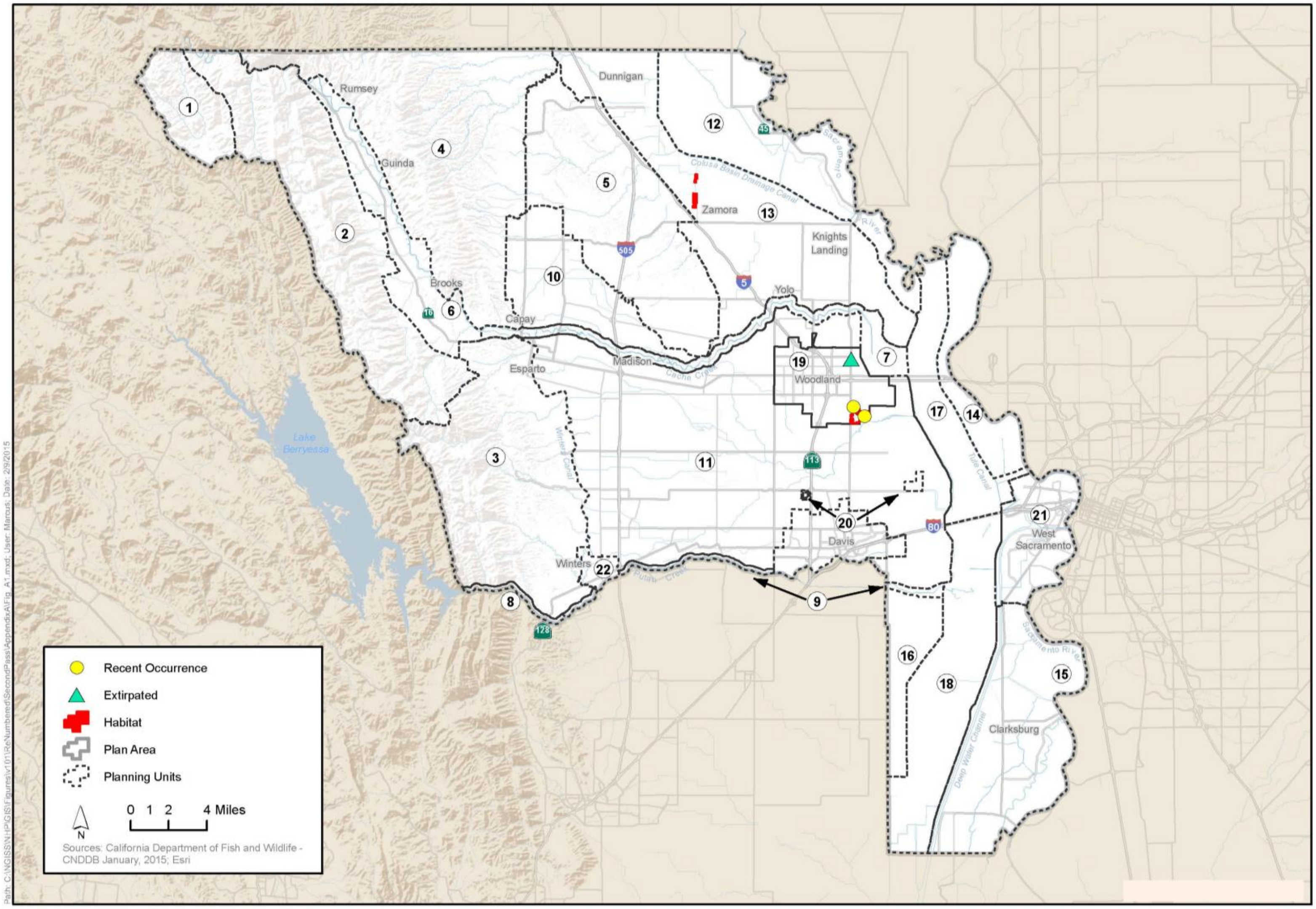
A.1.6.1 Geographic Information System (GIS) Map Data Sources

The palmate-bracted bird's-beak habitat is map based and uses the Yolo NHP vegetation dataset, which is based on a heads-up GIS digitization of the alkali sink habitat in the NHP Plan Area (Figure A-4). A habitat map of the distribution of palmate-bracted bird's-beak habitat in the Plan Area was then created. The habitat type was based on the species requirements as described in Section A.4.3, *Habitat Requirements and Ecology* above and the assumptions described below. Occurrences were mapped as the point at the center of any California Natural Diversity Database (CNDDB) polygons that fall within the Plan Area.

Mapped palmate-bracted bird's-beak habitat is comprised of the following vegetation type.

- **Alkali Sink:** This habitat was mapped based on current and historical soils maps, aerial imagery from 1933 and 1952, and current Google Earth imagery to determine existing land use. Additional habitat was mapped in Planning Unit 13 using polygons supplied by the California Department of Fish and Wildlife (DFW).
- **Assumptions.** Historical and current records of this species in the Plan Area indicate that it was present in the alkaline soil area between Willow Slough and Cache Creek, but that its known current distribution is limited to the mapped alkali sink habitat with some individuals present on adjacent severely disturbed sites.

Figure A-1. Palmate-Bracted Bird's Beak Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

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A.2 Valley Elderberry Longhorn Beetle (*Desmocerus californicus dimorphus*)

A.2.1 Listing Status

Federal: Threatened.

State: None.

Recovery Plan: None.

A.2.2 Species Description and Life History

The valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*) is an atypical lepturine; the Lepturinae is a subfamily of the Cerambycidae (longhorn beetle family). Elderberry beetles are separated from all other lepturines by the form of the mandibles, which are broad and short, without internal pubescence (Linsley and Chemsak 1972). Originally described by Horn (1881), valley elderberry longhorn beetle is black in color, with red to orange margins on the elytra (wing covers), which fades to yellow after death. The pronotum (plate behind the head) is smooth, with confluent punctuations. The elytra are densely punctate or rugose. Adult beetles range from 14 to 25 millimeters (mm) (0.55 to 0.98 inch) in length (Linsley and Chemsak 1972).

The valley elderberry longhorn beetle was described as a separate species by Fisher (1921) and was reduced to subspecific status by Doane et al.(1936). The majority of male valley elderberry longhorn beetles can be separated from other subspecies by the short, suberect, pale setae (bristle or hair-like structures) on the antennae (as opposed to dark setae) and the black markings on each forewing (Linsley and Chemsak 1972). The female valley elderberry longhorn beetle cannot be separated morphologically from other subspecies.

Female valley elderberry longhorn beetles lay between eight and 20 eggs in bark crevices on the host plant and produce only one generation per year (Burke 1921; Barr 1991). The host plant is the elderberry (*Sambucus mexicana*, *S. caerulea, S. racemosa, S. glauca*) (Burke 1921; Linsley and Chemsak 1972, 1997; Barr 1991). The eggs, which are white initially then darken to a reddish brown, are 3.5 to 1.25 mm (0.14 to 0.05 inch) in diameter; oblong with a small knob at each end; and have wavy, longitudinal ridges (Burke 1921; Barr 1991). The egg is attached to the shrub by a thin secretion, and the larva encloses within 30 to 40 days (Burke 1921).

The newly emerged larvae bore into the wood of the host plant (Linsley and Chemsak 1972; Barr 1991). Burke (1921) and Eya (1976) reported that the larvae take two years to mature; however, Halstead (1991) believes that one year is the norm. The larva typically bores into the central pith of stems and feeds there; however, on large trunks, the larvae feed on the wood (Burke 1921). The larvae create an elongated, longitudinal gallery through the heart of the stems, filling it with debris and shredded wood (Barr 1991). When the larva is ready to pupate, it chews a circular to slightly oval exit hole (7 to 10 mm [0.28 to 0.39 inch] in diameter) to the outside, which it plugs with frass. Then the larva backs up into the gallery and constructs a pupal chamber out of shredded wood and frass (Barr 1991). Jones & Stokes (1985, 1986, 1987a, 1987b) and Halstead (1991) reported that 70 percent of exit holes are within 1.2 meters (3.9 feet) of the ground in stems greater than 13 mm (0.51 inch) in diameter; however, holes may be as high as 3 meters (10 feet) above the ground (Barr 1991). Pupae can be found between January and April, and the pupal stage lasts about one month (Burke 1921).

After pupation, the adult remains in the pupal cell for several weeks prior to emergence (Burke 1921). The adult eventually emerges from the pupal chamber through the exit hole (Barr 1991). The adults readily fly from shrub to shrub. Valley elderberry longhorn beetle is most often seen on, in, or immediately under the host plant's flowers. However, copulation occurs on the lower parts of the stems (Barr 1991). The adults feed on the leaves (Linsley and Chemsak 1972; Barr 1991; Talley et al. 2006) and are active from March to early June.

A.2.3 Habitat Requirements and Ecology

The valley elderberry longhorn beetle is completely dependent on its host plant, the elderberry (Linsley and Chemsak 1972, 1997; Eng 1984; Barr 1991; Collinge et al. 2001). This shrub is a component of riparian forests throughout the Central Valley. Although this shrub occasionally occurs outside riparian areas, shrubs supporting the greatest beetle densities are located in areas where the shrubs are abundant and interspersed among dense riparian forest, including Fremont cottonwood (*Populus fremontii*), box elder (*Acer negundo*), California sycamore (*Platanus racemosa*), California walnut (*Juglans californica*), white alder (*Alnus rhombifolia*), willow (*Salix* spp.), button willow (*Cephalanthus occidentalis*), Oregon ash (*Fraxinus latifolia*), wild grape (*Vitis californica*), California hibiscus (*Hibiscus californica*), and poison oak (*Toxicodendron diversilobum*) (Barr 1991; USFWS 1999; Collinge et al. 2001). There is also a strong association between blue elderberries and valley oaks which historically extended beyond riparian zones. Isolated elderberry shrubs separated from contiguous habitat by extensive development are not typically considered to provide viable habitat for valley elderberry longhorn beetle (USFWS 1998; Collinge et al. 2001).

Elderberry savannah was a habitat type that was previously more extensive in the California Central Valley but now is limited to the confluence area of the American River, which is outside the Plan Area (Jones & Stokes 1985, 1986, 1987a, 1987b; Barr 1991; USFWS 1984, 1999), and the valley elderberry longhorn beetle was probably a component of this habitat. Therefore, potential valley elderberry longhorn beetle habitat is defined as stands of elderberry shrubs that are adjacent to, or contiguous with, riparian forest, floodplains, or relict elderberry savannah.

There are no known diseases that are considered a source of mortality for valley elderberry longhorn beetle. Numerous species of Cleridae (checkered beetles), Cucujidae (flat bark beetles), Ostomatidae (bark-gnawing beetles), Elateridae (click beetles), Asilidae (robber flies), Phymatidae (ambush bugs), Reduviidae (assassin bugs), and some Thysanoptera (thrips) are known predators of Cerambycid beetles (Linsley 1961). All are common in the Central Valley, but none have been reported feeding on valley elderberry longhorn beetle.

Birds that hunt insect larvae in wood, such as woodpeckers, creepers, and nuthatches, may also predate upon valley elderberry longhorn beetle but no observations of this have been reported. Due to the valley elderberry longhorn beetle's warning colors, birds may not take adult beetles. Whether these warning colors are genuine or represent Batesian mimicry is unknown.

A.2.4 Species Distribution and Population Trends

A.2.4.1 Distribution

Desmocerus californicus is one of three species of *Desmocerus* in North America. Valley elderberry longhorn beetle is one of two subspecies of *D. californicus*. One subspecies is widespread in coastal California, ranging from Mendocino County southward to western Riverside and northern San Diego Counties, and into the southern Sierra Nevada range (Kern and Tulare Counties).

The valley elderberry longhorn beetle subspecies is a narrowly defined, endemic taxon, limited to portions of the Central Valley (USFWS 1999; USFWS 2006). Studies to assess the distribution and extent of the valley subspecies began in the late 1970s (Eya 1976), and the USFWS proposed the species for listing in 1978. Since valley elderberry longhorn beetle was listed in 1980 (45 FR 52803), numerous distributional studies have been conducted (summarized in Talley et al*.* 2006). This subspecies is endemic to California, occurring below 900 meters (2,953 feet) elevation (USFWS 1999).

In the Central Valley of California, valley elderberry longhorn beetle was first collected from “Sacramento, CA,” the precise location unknown (Fisher 1921). Additional material was identified from Putah Creek in Solano and Yolo Counties and from along the Lower American River in Sacramento County (Linsley and Chemsak 1972). Linsley and Chemsak (1972) also reported a single female from the Merced River; however, since the females cannot be separated to subspecific level, the identification is unverified.

Subsequent to various surveys throughout the California Central Valley, the USFWS (1999) prepared a map of the presumed range of valley elderberry longhorn beetle. This map encompasses the entire California Central Valley and the Sacramento River Delta below 900 meters (2,953 feet) elevation.

In Yolo County, numerous records of occupied and potential valley elderberry longhorn beetle habitat occur throughout the Sacramento River corridor (Eya 1976; Jones & Stokes 1985, 1986, 1987a, 1987b; USFWS 1984; Barr 1991; Collinge et al. 2001; California Natural Diversity Database [CNDDB] 2000), as well as along Putah Creek from Monticello Dam east to Davis (Eya 1976; USFWS 1984; Barr 1991; Collinge et al. 2001; CNDDB 2005) and along Cache Creek (Barr 1991; CNDDB 2005). However, because comprehensive surveys for valley elderberry longhorn beetle in Yolo County have not been conducted and because known occurrences throughout the species’ range are based mostly on incidental observations (e.g., CNDDB), the population size and locations of this species in the Yolo Natural Community Conservation Plan (NCCP) study area are not fully known. Few surveys focused on valley elderberry longhorn beetle have been conducted within and adjacent to Yolo County, and the total extent of potential habitat is unknown. Within and adjacent to Yolo County exist several preserves, parks, and mitigation banks that support valley elderberry longhorn beetle occurrences, including the Lake Solano Park and the American River Parkway.

A.2.4.2 Population Trends

Habitat occupied by valley elderberry longhorn beetle tends to form and exist in riparian corridors and on the level, open ground of periodically flooded river and stream terraces and floodplains. This geomorphic setting historically has been desirable for agricultural, urban, or industrial development. As a result, much of this habitat type has been converted through dams and levees for use as developable land. Although it has been estimated that 90 percent of California riparian habitat has been lost over the last century and a half (Smith 1980; Barr 1991; Naiman et al. 1993; Naiman and Décamps 1997), these losses are difficult to accurately quantify in terms of direct valley elderberry longhorn beetle habitat losses (Talley et al. 2006). Therefore, an unknown amount of riparian forest and elderberry savannah habitat has been lost and an unknown number of valley elderberry longhorn beetle populations as well (Collinge et al. 2001). Due to current pressures from increasing human populations in California, more valley elderberry longhorn beetle habitat is being encroached on and affected throughout the species' range.

A.2.5 Threats to the Species

The greatest historical threat to valley elderberry longhorn beetle has been the elimination, loss, or modification of its habitat by urban, agricultural, or industrial development and other activities that reduce or eliminate its host plants (Talley et al. 2006). While mitigation and restoration actions do not come close to restoring the enormous amount of habitat lost in the more remote past they appear to be adequate for current levels of impact (Talley et al. 2006). However, Talley et al. (2006) observed that the quality and persistence of mitigation and restoration efforts are uncertain and that there have been declines in the total number of valley elderberry longhorn beetle–occupied sites and in the number of riparian sites. Talley et al. (2006) also noted that the information included in reports is often unusable, making assessments of mitigation and restoration success difficult.

The greatest current threat to valley elderberry longhorn beetle is from the invasive nonnative Argentine ant (*Linepithema humile*) and European earwig (*Forficula auricularia*) (Talley et al*.* 2006). The nonnative invasive Argentine ant has been observed attacking and killing valley elderberry longhorn beetle larvae. The ants enter the exit hole that the beetle makes prior to pupation and remove the larva (Huxel 2000; Huxel et al. 2003). Given that the invasion of riparian systems by Argentine ant in the Central Valley is continuing to spread, it is unclear how the invasion will impact valley elderberry longhorn beetle, but it appears that the Argentine ant may have caused the disappearance of some populations (Talley et al. 2006). Field bait and trapping experiments have determined that Argentine ant has been introduced widely through mitigation plantings and irrigation (Klasson et al. 2005). Irrigation plays a major role in Argentine ant's rate and distance of dispersal in other ecosystems (Menke and Holway 2006). Those data also suggest that there may be a threshold of Argentine ant density above which valley elderberry longhorn beetle is extirpated from a site (Klasson et al. 2005). If confirmed, this would be a serious threat to valley elderberry longhorn beetle's recovery because once valley elderberry longhorn beetle is extirpated from a site, recolonization is unlikely (Talley et al. 2006). The nonnative invasive European earwig is also considered to be a threat to valley elderberry longhorn beetle through direct predation or by supporting higher populations of predators of insects (Talley et al. 2006), and earwig populations are also significantly larger in mitigation plantings and irrigated areas (Klasson et al. 2005).

Nonnative invasive plant species such as black locust (*Robinia pseudoacacia*), giant reed (*Arundo donax*), red sesbania (*Sesbania punicea*), Himalaya blackberry (*Rubus armeniacus*), tree of heaven (*Ailanthus altissima*), Spanish broom (*Spartium junceum*), Russian olive (*Eleagnus angustifolia*), edible fig (*Ficus carica*), and Chinese tallowtree (*Sapium sebiferum*), may have significant indirect impacts on valley elderberry longhorn beetle by impacting elderberry shrub vigor and recruitment (Talley et al. 2006). It is also predicted that ripgut brome (*Bromus diandrus*), foxtail barley (*Hordeum murinum*), *Lolium multiflorum*, and yellow starthistle (*Centaurea solstitialis*) may increase seedling mortality through competition for light and water or through increased fire return intervals (Talley et al. 2006).

The taxonomic status of valley elderberry longhorn beetle was questioned by Halstead (1991) and Halstead and Oldham (2000). However, in a reanalysis of that data in support of the five-year status review, Talley et al. (2006) found that it supported a distinct biomodal distribution separation between California elderberry longhorn beetle and valley elderberry longhorn beetle. That analysis also found that there appeared to be some interbreeding where there is contact between the two subspecies, and molecular genetic study would be required to completely describe their distributions (Talley et al. 2006).

Long-term data regarding site persistence, population size and dynamics, extirpation, and recolonization are also lacking, as are estimates regarding the minimum self-sustaining population size, riparian forest corridor size, or habitat complex size for valley elderberry longhorn beetle or other riparian forest organisms.

A.2.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.14.3, *Habitat Requirements and Ecology* (Figure A-14).

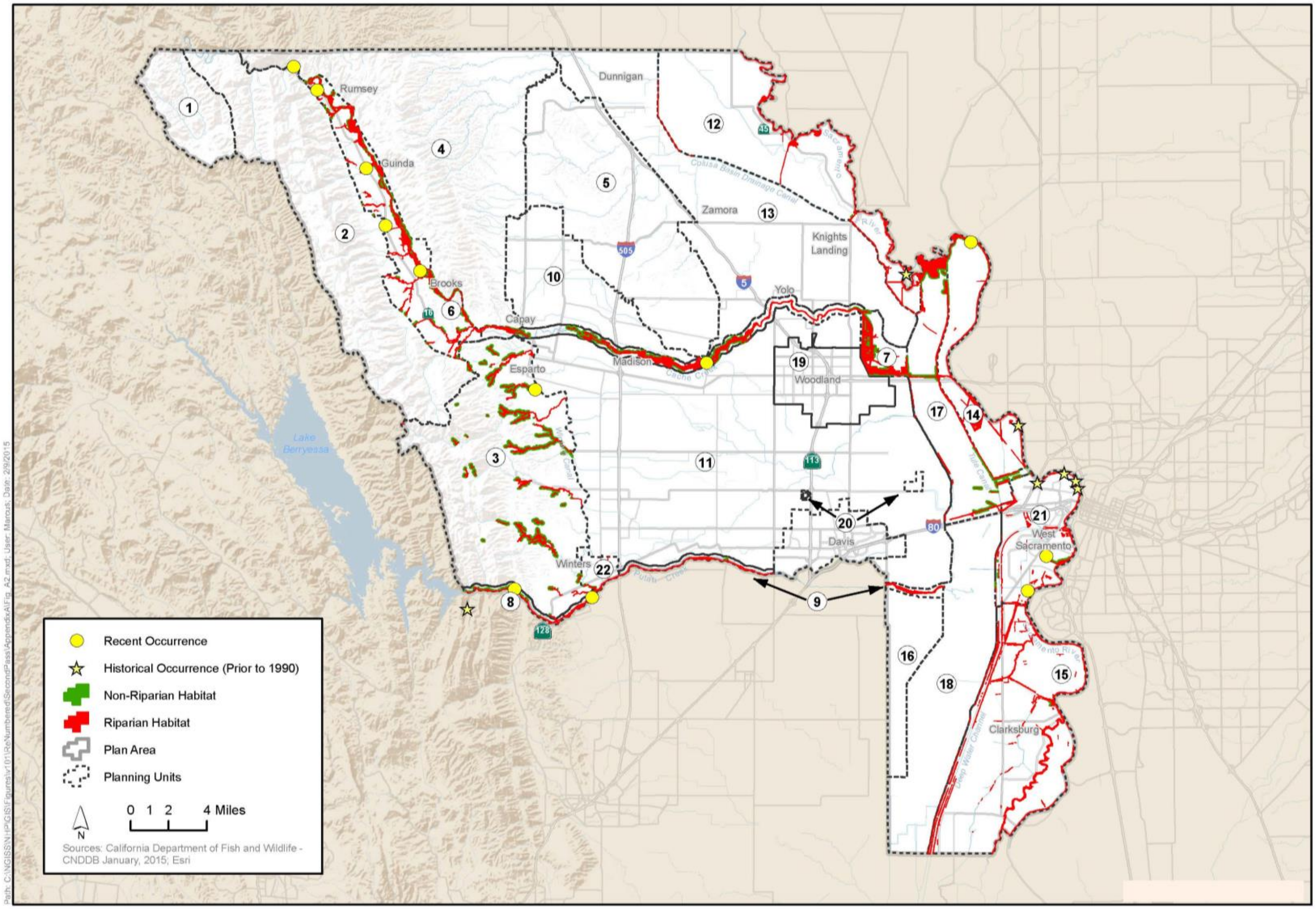
The model parameters include the following:

- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: This is the location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., CNDDB, University of California, Davis).
- Riparian Habitat: This habitat includes all potentially suitable riparian habitat where elderberry shrubs (the species host plant) are most likely to occur. This habitat was modeled by selecting all mapped Valley Foothill Riparian vegetation types.
- Nonriparian Habitat: This habitat includes all potentially suitable areas adjacent to the riparian zone that are likely to also include elderberry shrubs. This habitat was modeled by creating a buffer zone of 250 feet from modeled riparian habitat and selecting the vegetation types listed below.
- Limited modeling to the following Planning Units: 3, 6, 7, 8, 9, 12, 14, 15, 17, 20, 21, 22

A.2.6.1 Nonriparian Habitat–Vegetation Types

- All Annual Grassland
- All Barren
- Carex spp. – *Juncus* spp. – Wet Meadow Grasses Not Formally Defined (NFD) Super Alliance
- *Crypsis* spp. – Wetland Grasses – Wetland Forbs NFD Super Alliance

Figure A-2. Valley Elderberry Longhorn Beetle Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

A.3 California Tiger Salamander (*Ambystoma californiense*)

A.3.1 Listing Status

Federal: Threatened range-wide (69 *Federal Register* [FR] 47212); Endangered Sonoma County (65 FR 57242); Endangered Santa Barbara County (68 FR 13498); critical habitat designated (70 FR 49380).

State: Candidate Endangered; Species of Special Concern.

Critical Habitat: Endangered and Threatened Wildlife and Plants: Designation of Critical Habitat for California Tiger Salamander; Central Population: Final Rule (70 FR 49380–49458).

The Dunnigan Creek Unit (Central Valley Region Unit 1) of designated critical habitat, comprising 1,105 hectares (2,730 acres), located just west of Interstate 5 and the town of Dunnigan in north-central Yolo County, is the only unit within the Plan Area. Critical habitat has also been designated in Santa Barbara County (69 FR 68568) and within 20 counties in central California, including Yolo County (70 FR 49380).

Recovery Plan: Under development.

A.3.2 Species Description and Life History

The California tiger salamander (*Ambystoma californiense*) is an amphibian in the family Ambystomatidae. These terrestrial salamanders are large and thickset, with a wide, rounded snout (69 FR 47212). Adults range in size from 7.5 to 12.5 centimeters (cm) (2.95 to 4.92 inches) snout-to-vent length (SVL) (Jennings and Hayes 1994). Average SVL for both adult males and females is approximately 9 cm (3.58 inches), although the average total length for males and females is 20.3 and 17.3 cm (7.99 and 6.81 inches), respectively (69 FR 47212). Dorsal (back) coloration consists of a black background on the back and sides, interspersed with white or pale yellow spots or bars (69 FR 47212). Ventral (belly) coloration ranges from almost uniform white or pale yellow to a variegated pattern of white, pale yellow, and black (Jennings and Hayes 1994). The salamander's small eyes have black irises and protrude from their heads (Jennings and Hayes 1994). During the breeding season, the cloacal region of males becomes enlarged (Petranka 1998) and is a useful means of distinguishing sexes. The cloaca is a body cavity that receives the collective discharges from the intestinal, urinary, and reproductive canals. Males also have larger tails with more developed fins.

The California tiger salamander is restricted to grasslands, oak savannah, and coastal scrub communities of lowlands and foothill regions where aquatic sites are available for breeding. California tiger salamanders are typically found at elevations below 460 meters (1,509 feet) (68 FR 13498), although the known elevational range extends up to 1,053 meters (3,458 feet) (Jennings and Hayes 1994). Breeding sites generally consist of natural ephemeral pools (Barry and Shaffer 1994) or artificial ponds that mimic them (e.g., stock ponds that are allowed to dry). Bobzien and DiDonato (2007) report that in the East Bay Regional Park District (Contra Costa and Alameda

Counties) California tiger salamanders breed almost exclusively in seasonal and perennial stock ponds. Breeding sites may also include perennial features with open water refugia that do not support populations of bullfrog (*Rana catesbeiana*) or predatory fishes (Holomuzki 1986; Fitzpatrick and Shaffer 2004). Pools characterized by deep water may also support larvae through metamorphosis in relatively dry years (Trenham et al. 2000), whereas shallow pools may not (Semlitsch et al. 1996). Populations associated with shallow, natural vernal pools may be more dependent on suitable hydroperiod (Trenham et al. 2000). As illustrated by the 114-year-old reservoir at Lagunita (Stanford University, Santa Clara County), constructed ponds may also serve as habitat for California tiger salamander as long as they are drained annually, thus preventing exotic fish and amphibian predators (i.e., bullfrogs) from establishing (Barry and Shaffer 1994). Barry and Shaffer (1994) attribute the persistence of the salamander population at Lagunita to (1) large size of both aquatic and terrestrial habitats, and (2) the continuous filling and draining of the reservoir every year, which provides larvae a head start over fish predators each year.

Larvae require a minimum of approximately 10 weeks to complete metamorphic transformation (P. Anderson 1968; Feaver 1971), significantly longer than other amphibians such as the Pacific tree frog (*Pseudacris regilla*) and western spadefoot (*Spea hammondii*). Hydroperiod, or the timing and duration of waters in potential breeding sites, can be critical for reproductive success. Shaffer et al. (2008) indicate that California tiger salamanders can breed successfully in stock ponds, and in natural or constructed vernal pools remaining wet until mid-May. Larvae in coastal regions may not metamorphose until late July, and pools holding water into June, July, or later generally have higher success (Barry and Shaffer 1994). Larvae have been documented overwintering in perennial ponds in the higher elevations of the Ohlone Regional Wilderness in Alameda County (Bobzein and DiDonato 2007). Compared to the western toad (*Bufo boreas*) or western spadefoot, California tiger salamanders are poor burrowers and require subterranean refuges constructed by ground squirrels and other burrowing mammals (Jennings and Hayes 1994). Salamanders spend the dry season, which comprises most of a year, within these burrows (69 FR 47212). Although California tiger salamanders are often considered to be in a state of dormancy, called aestivation, during the period in which in they occupy these burrows, evidence suggests that salamanders may remain active while within their burrows (S. Sweet in litt. in 69 FR 47212).

Males usually migrate to the breeding ponds before females (Twitty 1941; Shaffer et al. 1993, Loredo and Van Vuren 1996; Trenham 1998b) and remain in the ponds for an average of six to eight weeks, while females stay for approximately one to two weeks (USFWS 2004b). Salamanders typically return to the same pond to breed in subsequent breeding seasons (Trenham 1998b). However, interpond dispersal does occur and is dependent on the distance between ponds and the quality of intervening upland habitat (Trenham 1998a). It appears that breeding takes place in pulses, with time between breeding events and the proportion of breeding adults per event associated with rainfall patterns and wetland inundation (J. Alvarez pers. comm.; S. Bobzein pers. comm.; D. Cook pers. comm.; M. Ryan pers. comm.). In Sonoma County there is a main breeding event in mid-December, which corresponds to the first large winter rain event that is sufficient to fill vernal pools, followed by one to two smaller breeding events after the next rainfalls (D. Cook pers. comm.). In drought years, insufficient water in the breeding pools may prevent breeding (Barry and Shaffer 1994). Trenham et al. (2000) found that within a population in Monterey County, female California tiger salamanders skipped breeding opportunities at a higher rate than males in years with later rainfall, a bias attributed to the date of pond filling, but not to total annual rainfall. Barry and Shaffer (1994) suggest that while local California tiger salamander populations may not breed during drought years when ephemeral pools do not fill, the longevity of adults is probably sufficient to ensure population persistence through all but the longest of droughts.

After mating, females lay their eggs in the water of the breeding habitat (Twitty 1941; Shaffer et al. 1993; Petranka 1998). Females usually attach their eggs to twigs, grass stems, vegetation, or debris (Storer 1925; Twitty 1941; Jennings and Hayes 1994). After breeding, adults leave the pool and return to the upland habitat, taking shelter during the day in small mammal burrows and emerging at night to feed during the breeding season (Shaffer et al. 1993; Loredo et al. 1996; Trenham 1998a). In two to four weeks, eggs hatch into aquatic larvae (Petranka 1998). Larvae feed on zooplankton, small crustaceans, and aquatic insects for about six weeks and then begin consuming larger prey such as small tadpoles (J. Anderson 1968). The larval stage usually lasts three to six months (Petranka 1998), but individuals may remain in their breeding sites over the summer if breeding pools remain inundated (Shaffer and Trenham 2005). The longer the inundation period, the larger the larvae and metamorphosed juveniles are able to grow, and the more likely they are to survive and reproduce (Semlitsch et al. 1988; Pechmann et al. 1989; Morey 1998; Trenham 1998b).

Lifetime reproductive success for California tiger salamanders is generally low, with many individuals breeding only once in their lifetime (Trenham 1998b; Trenham et al. 2000). Over the lifetime of a female, only a small number of metamorphic offspring are produced; and only a small percentage of a cohort survive to become breeding adults (Trenham 1998b; Trenham et al. 2000). Trenham et al. (2000) found that reproduction at Hastings Reserve in Monterey County was lower than replacement in all of six years studied. According to this study, the average female California tiger salamander breeds 1.4 times over a lifetime, producing 8.5 young surviving to metamorphosis per event and 12 lifetime metamorphic offspring per female (Trenham et al. 2000). To achieve 1:1 replacement by this reasoning would require 18.2 percent survival from metamorphosis to breeding; survival at Hastings during this time was only 5 percent, leading the authors to suggest that isolated breeding ponds may be insufficient for maintaining viable populations over the long term.

Juvenile California tiger salamanders have been observed to disperse up to 2.59 kilometers (km) (1.6 mile) from breeding pools to upland areas (Austin and Shaffer 1992). Adults have been observed up to 2 km (1.3 miles) from breeding ponds. Trenham et al. (2001) observed California tiger salamanders moving up to 670 meters (2,198 feet) between breeding ponds in Monterey County. Similarly, Shaffer and Trenham (2005) found that 95 percent of California tiger salamanders resided within 640 meters (2,100 feet) of their breeding pond at Jepson Prairie in Solano County.

Adults emerge from upland sites on rainy nights during fall and winter rains to feed and migrate to breeding ponds (Stebbins 1989, 2003; Shaffer et al. 1993). Adults use the same migratory routes between breeding pools and upland burrows year after year (Petranka 1998; Loredo et al. 1996). Metamorphosed juveniles leave the breeding sites in late spring or early summer and migrate to small mammal burrows (Zeiner et al. 1988; Shaffer et al. 1993; Loredo et al. 1996). Like adults, juveniles may emerge from burrows to feed during nights of high relative humidity (Storer 1925; Shaffer et al. 1993) before settling in their selected upland sites for the summer months. While most California tiger salamanders rely on rodent burrows for shelter, some individuals may utilize soil crevices as temporary shelter during upland migrations (Loredo et al. 1996).

The distance between occupied upland habitat and breeding sites depends on local topography and vegetation, and the distribution of California ground squirrel (*Spermophilus beecheyi*) or other rodent burrows (Stebbins 1989). California tiger salamanders seem to follow the pattern of a broadly defined metapopulation structure, in which a population is divided into a set of subpopulations, some of which become extinct and are later recolonized by migrants from other subpopulations (69 FR 47212). Semlitsch et al. (1996) points out that because many vernal pools and ponds used by salamanders are temporary over geological and ecological time, local extinction must be counterbalanced by colonization of new sites; thus, conservation plans must incorporate terrestrial habitats providing corridors for movement to new sites. In the case of California tiger salamanders, Trenham (1998b) indicates that the spatial arrangement of ponds and the migratory behavior of salamanders substantially affect pond utilization and sustainability of local populations. Interpond distances directly affect the probability of recolonization and subsequent opportunities for population rescue, which is important because physiology limits the distance that amphibians are able to disperse (Semlitsch 2000). While Marsh and Trenham (2001) reviewed the fit between theoretical metapopulations and pond-breeding amphibians and found that random extinctions of local populations were uncommon as long as terrestrial habitats were intact, Trenham and Shaffer (2005) found that local extinctions were likely where the probability of reproductive failure exceeded 0.5, and that reproductive failure was common in both permanent and highly ephemeral pools, underscoring the importance of interconnected breeding sites.

A.3.3 Habitat Requirements and Ecology

A diverse array of flora and fauna have adapted to the seasonal hydric cycle of vernal pools (69 FR 47212). Vernal pools and other seasonal rain pools are the primary breeding habitat of California tiger salamanders (68 FR 13498). Within the species range, there are numerous other sensitive vernal pool species, comprising 24 plants, four crustaceans, and one insect (Keeler-Wolf et al. 1998). Listed vernal pool crustaceans are able to complete their life cycle within a relatively short period of inundation (59 FR 48136). Therefore, many pools that support vernal pool crustaceans may not retain water for the 10 weeks or more required to complete metamorphosis of California tiger salamander larvae (P. Anderson 1968; Feaver 1971). Laabs et al. (2001) reported that, in eastern Merced County, California tiger salamander larvae were observed only in the largest vernal pools. California tiger salamanders, unlike vernal pool crustaceans, are known to successfully reproduce in perennial ponds (69 FR 47212).

Outside of the breeding season, post-metamorphic California tiger salamanders spend most time in burrows of small mammals, such as California ground squirrels and Botta's pocket gopher (*Thomomys bottae*) (Storer 1925; Loredo and Van Vuren 1996; Petranka 1998; Trenham 1998a). Active rodent burrow systems are considered an important component of California tiger salamander upland habitat (Seymour and Westphal 1994; Loredo et al. 1996). Utilization of burrow habitat created by burrowing mammals such as ground squirrels suggests a commensal relationship (a relationship between two species in which one obtains food or other benefits without detriment or benefit to the other) between the two species (Loredo et al. 1996). Loredo et al. (1996) indicate that active ground-burrowing rodent populations are probably necessary to sustain California tiger salamander populations because inactive burrow systems begin to deteriorate and collapse over time. In a two-year radiotelemetry project in Monterey County (Hastings), Trenham (2001) found that salamanders preferentially used open grassland and isolated oaks; salamanders present in continuous woody vegetation were never more than 3 meters from open grassland, potentially because ground squirrels prefer to construct burrows in open habitats (Jameson and Peeters 1988 in Trenham 2001).

A.3.4 Species Distribution and Population Trends

A.3.4.1 Distribution

The California tiger salamander is endemic to California. Within the coastal range, the species occurs from southern San Mateo County south to San Luis Obispo County, with isolated populations in Sonoma and northwestern Santa Barbara Counties (CNDDB 2005). In the Central Valley and surrounding Sierra Nevada foothills, the species occurs from northern Yolo County southward to northwestern Kern County and northern Tulare and Kings Counties (CNDDB 2005). Throughout its range, occurrences of California tiger salamander are strongly associated with uplifted and dissected undeformed to moderately deformed Plio-Pleistocene sediments (Jennings and Hayes 1994, Wahrhaftig and Birman 1965).

Recorded occurrences of California tiger salamanders in Yolo County include an occurrence of several larvae in a stock pond on the west slope of the Capay Hills east of Rumsey Rancheria (Downs 2005), and five occurrences in the northern end of the Solano-Colusa vernal pool region, west and northwest of Dunnigan (CNDDB 2007) (Figure A-15). Four recorded occurrences were located within an area bounded by Interstate 5 to the east, Bird Creek to the south, and Buckeye Creek to the north and west. These four occurrences are from within an area that now comprises the Dunnigan Creek Unit (Central Valley Region Unit 1) of designated critical habitat Land ownership within this unit is entirely private (70 FR 49380) and therefore restricted (another historical, but extirpated occurrence, is recorded from a site adjacent to the designated critical habitat). A fifth recorded occurrence, from 1993, represents an individual found in the Willows apartment complex in Davis, adjacent to a stormwater detention basin managed by the City of Davis (CNDDB 2007). Queries of the online databases of the California Academy of Sciences (2008) and Museum of Vertebrate Zoology (2008) yielded no additional occurrence records.

A.3.4.2 Population Trends

California tiger salamanders still occur throughout much of their historical range (Trenham et al. 2000) and can be common at localities where it still occurs. Total adult population size is unknown, but certainly exceeds 10,000. Populations are thought to be declining due to habitat loss. Approximately 75 percent of the species' historical natural habitat has been lost. The species has been eliminated from 55 to 58 percent of historical breeding sites. Holland (1998) indicated that about 75 percent of the historical vernal pool breeding habitat has been lost, although some question the reliability of this estimate. Barry and Shaffer (1994) stated that this salamander soon will be in danger of extinction throughout its range and noted that it already is gravely threatened in the San Francisco Bay Area and in the San Joaquin Valley. In Santa Barbara County, half of the 14 documented breeding sites have been destroyed or have suffered severe degradation since mid-1999 (65 FR 57242).

Little is known of the population trends of California tiger salamanders in Yolo County. Four of the five recorded occurrences of the species in the county are from within an area that now comprises the Dunnigan Creek Unit (Central Valley Region Unit 1) of designated critical habitat. Land ownership within this unit is entirely private (70 FR 49380) and therefore restricted. The fifth recorded occurrence, in the City of Davis, consists of a solitary individual; lack of supporting habitat suggests this observation is the result of a translocated individual or a released pet (M. Ryan pers. comm.).

A.3.5 Threats to the Species

Conversion of land to residential, commercial, and agricultural activities is considered the most significant threat to California tiger salamanders. These activities result in destruction and fragmentation of upland and/or aquatic breeding habitat, and killing of individual California tiger salamanders (Twitty 1941; Hansen and Tremper 1993; Shaffer et al. 1993; Jennings and Hayes 1994; Fisher and Shaffer 1996; Launer and Fee 1996; Loredo et al. 1996; Davidson et al. 2002).

Fisher and Shaffer (1996) found an inverse relationship between introduced exotics and native amphibians. Exotic species, such as bullfrogs (*Ranacates beiana)*, mosquitofish (*Gambusia affinis*), sunfish species (e.g., largemouth bass [*Micropterus salmoides*] and bluegill [*Lepomis macrochirus*]), catfish (*Ictalurus* spp.), and fathead minnows (*Pimephales promelas*), that live in perennial ponds such as stock ponds are considered to have negatively affected California tiger salamander populations by preying on larval salamanders (Morey and Guinn 1992; Graf and Allen-Diaz 1993; Shaffer et al. 1993; Seymour and Westphal 1994; Fisher and Shaffer 1996; Lawler et al. 1999; Laabs et al. 2001; Leyse 2005). Shaffer et al. (2008) found that for other ambystomatids, introduction of larger fish can result in the loss of salamander life stages within one year while introduction of mosquitofish (*Gambusia affinis*) can eliminate salamanders in three to four years. Native fish, including salmonids, are known to prey on amphibian larvae that are palatable (Hencar and M'Closkey 1996). In a thorough review of available data, Fisher and Shaffer (1996) found that historical California tiger salamander localities are lower in elevation than current ones, implying extirpation in many areas occurring below 200 meters. In general, introduced exotics now occupy lower elevations, and suggest that habitat modification and low levels of topographic relief may facilitate invasion by increasing opportunities for dispersal through interconnected watersheds or suitable terrestrial habitats, or through deposition by floodwaters (Fisher and Shaffer 1996). Bobzein and DiDonato (2007) found pond co-occurrence to be negatively correlated for California tiger salamander and California newt, with sympatry only occurring in xeric regions of oak savannas and open woodland habitats. California newts are generally associated with mesic habitats such as redwood forests, deciduous hardwood forests, and oak bay woodlands, suggesting that California tiger salamanders and California newts segregate out along elevation lines (Bobzein and DiDonato 2007).

Pond size may bear on the ability of California tiger salamander to avoid invertebrate predators. In large fishless ponds, *A. Tigrinum nebulosum* larvae avoided predation by aquatic invertebrates by moving from the shallow, vegetated margins to deeper waters while predators were active (Holomuzki 1986), underscoring the importance of pond size and open water refuge for larval success.

Riley et al. (2003) examined hybridization between California tiger salamanders and an introduced congener, the tiger salamander (*Ambystoma tigrinum*). The tiger salamander has been deliberately introduced as fish bait in California and is contaminating the genome of California tiger salamanders through interbreeding (Riley et al. 2003). In the Salinas Valley, Riley et al. (2003) sampled salamanders from four artificial ponds and two natural vernal pools. Based on mitochondrial DNA and two nuclear loci, Riley et al. (2003) found that hybrids were present in all six ponds, and that these hybrids were viable and fertile. Hybridization with the barred tiger salamander (*Ambystoma tigrinum mavortium*) has been occurring since fishermen and bait shop owners began introducing the species 50 to 60 years ago, resulting 15–30 generations of genetic mixing (Fitzpatrick and Shaffer 2004). Fitzpatrick and Shaffer (2004) report more nonnative alleles in large perennial ponds despite the proximity of ephemeral ponds, perhaps attributable to the presence of open water refugia providing an extended breeding season or facilitating a paedomorphic life history strategy in which adult salamander retain larval characteristics. Fitzpatrick and Shaffer (2007) report evidence of hybrid vigor or increased fitness of hybrids based on early-larval survival. This finding raises questions regarding the relative values of genetic purity verses fitness and viability that are central to developing conservation strategies for California tiger salamander.

Pesticides, hydrocarbons, and other pollutants are all thought to negatively affect breeding habitat, while rodenticides and gases used in burrowing mammal control (e.g., chlorophacinone, diphacinone, strychnine, aluminum phosphide, carbon monoxide, and methyl bromide) are considered toxic to adult salamanders (Salmon and Schmidt 1984). California ground squirrel and pocket gopher control operations may have the indirect effect of reducing the availability of upland burrows for use by California tiger salamanders (Loredo-Prendeville et al. 1994).

Roads can fragment breeding and dispersal migratory routes in areas where they traverse occupied habitat. Features of road construction, such as solid road dividers, can further impede migration, as can other potential barriers such as berms, pipelines, and fences.

In the 70 FR 49380 critical habitat designation for the California tiger salamander, the concept of critical habitat was described as follows: “Critical habitat identifies specific areas, both occupied and unoccupied by a listed species, which are essential to the conservation of the species and that may require special management considerations or protection.” 70 FR 49380 further stated that “primary constituent elements for the California tiger salamander are aquatic and upland areas, including vernal pool complexes, where suitable breeding and nonbreeding habitats are interspersed throughout the landscape, and are interconnected by continuous dispersal habitat,” and that one or more of the primary constituent elements are present in all areas proposed for designation as critical habitat for the central population.

A recovery plan has not yet been prepared for the California tiger salamander, although the 69 FR 47212 has stated the intention to do so. In the interim, efforts toward conservation and recovery of the species should emphasize habitat preservation. Specifically, efforts should be directed toward protecting sites with vernal pool and other suitable rain pool habitat—in the largest blocks possible—from loss, fragmentation, degradation, and incompatible uses. Surrounding upland habitats will require similar protections that conserve burrowing mammals. Managed grazing programs may be a necessary component at many or all preserve sites in order to maintain the open, low-height grasslands required to sustain populations of California ground squirrels.

Physical disturbances to the underlying soils of seasonal rain pools should be avoided, as such disturbances could reduce their water-retaining capacity (Jennings and Hayes 1994). Such disturbances to vernal pool substrates also could destroy eggs of listed crustacean species.

In locations where roads traverse potential migratory routes, tunnels should be incorporated into the road design (Barry and Shaffer 1994). Barriers to migration, in the form of solid road dividers, should also be avoided on roads traversing potential migratory routes (Shaffer et al. 1989 in Jennings and Hayes 1994). Other potential barriers, such as berms and certain types of pipelines or fences, that can inhibit or prevent migration, should be avoided (Jennings and Hayes 1994).

Pesticides, hydrocarbons, and other pollutants should not be used or applied in a manner that runoff of these substances is transported into potential California tiger salamander breeding habitat. Rodenticides and gases used in burrowing mammal control may be toxic to resident adult and juvenile salamanders. Operations to control California ground squirrel and pocket gopher populations should be avoided in areas where California tiger salamanders may be present due to direct effects on the species and the potential indirect effects of reducing the availability of upland burrows.

Efforts should be undertaken to control the spread and introduction of exotic predatory species such as bullfrogs, mosquitofish, sunfish, catfish, and fathead minnows that live in perennial ponds—especially in areas where California tiger salamanders are known to occur. Although the sale of nonnative tiger salamanders for use as fish bait has been banned in California, efforts should continue to prevent the introduction and spread of this species, which has been shown to interbreed with native California tiger salamanders.

Based on a Monterey County study and a limited understanding of essential terrestrial habitats and buffer requirements of the species, Trenham et al. (2001) recommended that plans to maintain local populations of California tiger salamanders should include pond(s) surrounded by buffers of terrestrial habitat occupied by burrowing mammals, but noted that single isolated ponds might not support populations indefinitely even if surrounded by optimal uplands (Pechman and Wilbur 1994; Semlitsch and Bodie 1998 in Trenham et al. 2001). Based on individual dispersal of juveniles up to 1000 meters from their pool of origin, Searcey and Shaffer (2008) estimated that 95 percent of the reproductive value from a single large pond falls within approximately 2.4 km. Based on these findings, Shaffer et al. (2008) recommend a minimum buffer of 1 mile around breeding pools, relating to a preserve size of approximately 800 hectares (1,977 acres), greatly exceeding the 290-meter upper bound described by Semlitsch and Bodie (2003). This recommendation provides a useful and reasonable guideline for establishing salamander preserves of minimal functional size. Due to the potential for extirpation at single ponds due to random, stochastic events, sites with multiple complexes of vernal pools surrounded by much larger areas of suitable upland habitat should be considered for preserve sites, if feasible. Furthermore, sites with potential linkage corridors to other subpopulations should be considered. Sites chosen for preserves should also be occupied by burrowing mammals, especially California ground squirrels, in order to provide terrestrial habitat. Because contiguous blocks of land this size are not always available (e.g., Sonoma County), an experimental metapopulation approach may be required.

In their final report to USFWS titled "Guidelines for the relocation of California tiger salamanders (*Ambystoma californiense*)," Shaffer et al. (2008) make the following principal management recommendations: (1) eliminate fish and bullfrogs, (2) provide a means for draining all permanent ponds or eliminate them in favor of ephemeral ponds, (3) pools ponds should have sufficient watershed to provide an adequate hydroperiod for metamorphosis (three to six months), and (4) graze or burn to manage upland and wetland vegetation. Maret et al. (2006) found that disturbance or disruption of natural disturbance regimes can increase invisibility by exotic predators, but that disturbance-intolerant fish and bullfrogs can be eliminated by pond drying. Bullfrogs, which prefer permanent or semi-permanent water (Stebbins 1951), may be less likely to establish in ephemeral waters (Barry and Shaffer 1994). Increased drying regimes can limit predators, but can also reduce viability of salamander populations by limiting salamander breeding. However, Maret et al. (2006) found that the negative effects of drying on Sonoran tiger salamanders were generally minor relative to the negative effects of less frequent drying, and recommend ponds of varying depth to maintain a suitable hydroperiod for successful salamander reproduction while keeping exotic predators in check. At appropriate densities, cattle grazing can extend hydroperiod in ephemeral wetlands (Marty 2005) and may be an important factor in counteracting the hydrologic changes associated with climate change (Pyke and Marty 2005). Livestock grazing may also assist in maintaining open grassland and oak savanna communities that support rodents such as California ground squirrel and valley pocket gophers that provide retreats for California tiger salamanders (Bobzein and DiDonato 2007).

The most significant data gaps regarding California tiger salamanders are a lack of knowledge of its distribution and population trends within the Plan Area. California tiger salamanders may be more abundant in the Plan Area than available occurrence records indicate; however, surveys have not been conducted within the Dunnigan Unit of proposed critical habitat area and other areas where the species potentially occurs, and no information indicates recent or ongoing surveys at any Yolo County sites from which occurrences have been recorded.

A.3.6 Species Habitat Model and Location Data

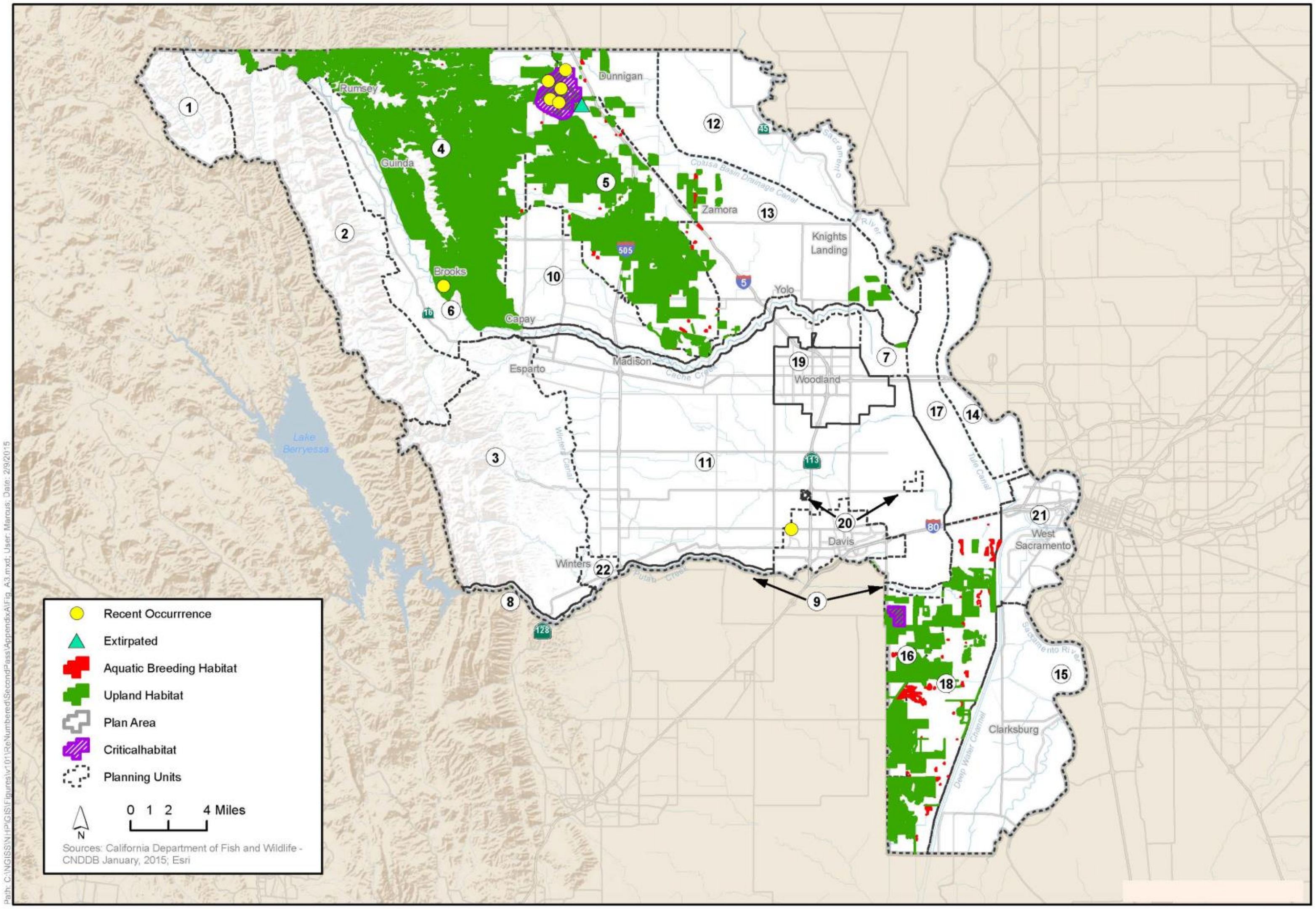
The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.15.3, *Habitat Requirements and Ecology* (Figure A-15). The model parameters include the following.

- Aquatic Breeding Habitat: This habitat includes all potentially suitable aquatic breeding areas and was modeled by selecting all mapped vernal pools, alkali sinks, and ponds (except those that are known to be perennial) as listed below that occur below an elevation 1,509 feet. Habitat located within planning units 1 – 3, 6 – 12, 14, 15, 17, and 19 - 22 is excluded from the model because these Planning Units are not known to be currently occupied and are isolated from occupied habitat areas and are unlikely to be occupied in the future (e.g., presence of levees and highways that create barriers to movement).
- Upland Habitat: This habitat includes all potentially suitable upland nonbreeding habitat (including aestivation and dispersal areas). This habitat was modeled by selecting all mapped vegetation types as listed below that occur within 1.3 miles of modeled breeding habitat and below an elevation 1,509 feet. Studies indicate that 95 percent of California tiger salamanders reside within 2,100 feet of breeding habitat (Shaffer and Trenham 2005). Habitat located within planning units 1 – 3, 6 – 12, 14, 15, 17, and 19 - 22 is excluded from the model for the reasons described above. Upland habitat in the Yolo Bypass is suitable as dispersal habitat but is considered to generally be unsuitable as aestivation habitat because of frequent winter flooding of the Bypass.

A.3.6.1 Upland Habitat – Vegetation Types

- All Annual Grassland
- Blue Oak Woodland
- All Blue Oak – Foothill Pine
- Valley Oak Alliance
- Pastures

Figure A-3. California Tiger Salamander Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

Path: C:\NGIS\SNHP\GIS\Figures\v101\ReNumbered\SecondPass\AppendixA\Fig_A3.mxd; User: Marcus; Date: 2/9/2015

A.4 Western Pond Turtle (*Actinemys marmorata*)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A.4.1 Listing Status

Federal: None.

State: Species of Special Concern.

Recovery Plan: None.

Other Common Names: Northern Pacific Pond Turtle

Other Related Names: *Clemmys marmorata marmorata* (Baird and Girard 1852); *Emys (=Clemmys) marmorata marmorata* (Baird and Girard 1852); *Emys marmorata marmorata* (Baird and Girard 1852).

A.4.2 Species Description and Life History

The western pond turtle (*Actinemys marmorata marmorata*) (Holman and Fritz 2001; McCord and Joseph-Ouni 2006; Obst 2003) is a medium-sized aquatic turtle. Previously assigned to the genus *Clemmys*, Feldman and Parham (2002) have also proposed taxonomic realignments that would place *A. marmorata* within the genus *Emys*; current literature may refer to this taxon under either generic name. The carapace (upper portion of shell) color ranges from brown to black (Holland 1994). The carapace may be unmarked or covered with small, fine dark spots or lines (Holland 1994; Stebbins 2003). Adult size ranges from 8.9 to 21.6 centimeters (3.5 to 8.5 inches) straight-line carapace length (Stebbins 2003). The plastron (lower portion of shell) contains six pairs of yellowish shields, usually with dark blotches (Stebbins 2003). The head usually contains spots or a network of black coloring (Stebbins 2003). Adult females have a more domed, taller carapace, as compared to males, which have a more flattened, lower profile carapace (Holland 1994). Males also have larger, thicker tails than females (Holland 1994). Juveniles have a uniformly brown or olive carapace, with yellow markings along the edge of the marginals (the ring of shields encircling the carapace) and a tail nearly as long as the carapace (Stebbins 2003).

Field observations have reported copulation in May, June, and late August (Holland 1988). Oviposition (egg-laying) may occur as early as late April in central California (Rathbun et al. 1993) to late July, with most occurring in June and July (Holland 1994). A gravid (pregnant) female approaches the nesting site, empties the contents of her bladder onto the soil, excavates a nest chamber 90 to 125 millimeters (3.5 to 4.9 inches) deep and deposits one to 13 hard-shelled eggs (Holland 1994, Jennings and Hayes 1994). Incubation time ranges from 80 to more than 100 days in California (Holland 1994). In Northern California, hatchling western pond turtles (which are about the size of a quarter) overwinter inside the nest chamber and emerge the following spring (Holland 1994). The terrestrial movements of post-emergent hatchlings are poorly understood (Holland 1994), although it is known that at least some move quickly to aquatic habitats.

Adults sometimes engage in extended overland movements, which may be in response to drought or normal movements to aquatic habitats within a home range (Holland 1994). In one study, a turtle was observed making an overland movement of 5 kilometers (km) (3.1 miles), although in all other cases, overland movements were less than 3 km (1.9 miles) (Holland 1994). Such overland movements may be responses to an environmental stress such as drought or may be part of an individual's normal movements within a home range, which may consist of a series of ponds (Holland 1994). In lotic (stream) habitats, individuals move along the watercourse from pool to pool. During the course of one summer, Bury (1972) found average male, female, and juvenile linear movements were 354, 169, and 142 meters (1,161, 554, and 466 feet), respectively. In that study, adult males had the largest home ranges (0.98 hectare [2.42 acres]), followed by juveniles (0.36 hectare [0.89 acre]) and adult females (0.25 hectare [0.62 acre]).

A.4.3 Habitat Requirements and Ecology

The western pond turtle, although primarily found in natural aquatic habitats, also inhabits impoundments, irrigation ditches, and other artificial and natural water bodies (Ernst et al. 1994) and is found at elevations ranging from sea level to 2,041 meters (6,696 feet) (Stebbins 2003). The species is usually found in fresh water, but brackish habitats are also utilized (Ernst et al. 1994; D. Holland pers. comm.). The aquatic habitat may be comprised of either mud or rocky substrates and usually contains some vegetation (Ernst et al. 1994). Habitat quality often seems to be positively correlated with the number of available basking sites (Jennings and Hayes 1994). Turtles seem to avoid areas lacking in significant refugia (Holland 1994). Basking sites may be rocks, logs, vegetation, terrestrial islands within the aquatic habitat, and human-made debris (Holland 1994). Hatchlings use shallow, slow-moving waters with emergent vegetation, such as that found alongside channels of stream or pond margins, while juveniles one year old or older tend to utilize the same aquatic habitats as adults (D. Holland pers. comm.). Western pond turtles may overwinter in aquatic or upland habitats (Holland 1994). Like the giant garter snake (*Thamnophis gigas*), western pond turtles inhabit the irrigation ditches servicing rice agriculture in the Central Valley (E. Hansen, unpublished notes). While rice fields probably confer little advantage for adult western pond turtles, mature rice probably provides valuable cover and foraging habitat for hatchlings.

When overwintering in aquatic habitats, turtles enter a state of torpor and rest quietly on the pond or stream bottom, often in mud or under some type of refugium such as a log or undercut bank (Holland 1994). Overwintering western pond turtles may move between several sites during winter and have been observed swimming under ice in water temperatures as low as 1 degree Celsius (°C) (34 degrees Fahrenheit [F]) (Holland 1994). Individuals may occasionally emerge to bask on warm, sunny days during winter, even in northern Oregon (D. Holland pers. comm.).

Western pond turtles are generalist feeders, with most food being obtained by opportunistic foraging or scavenging (Ernst et al. 1994). Known food items include algae, various plants, crustaceans, various types of insects, spiders, fish, frogs, tadpoles, and birds (Pope 1939 in Ernst et al. 1994; Evenden 1948 in Ernst et al.1994; Carr 1952; Holland 1985; Bury 1986). Scavenging carrion of various vertebrate species may be a locally and/or seasonally important part of the diet (Holland 1994). Neustophagia, (a form of filter feeding) may be utilized to obtain abundant small invertebrate prey such as *Daphnia* (Ernst et al. 1994; Holland 1994).

Upland habitats are also important to western pond turtles for nesting, overwintering, and overland dispersal (Holland 1994). Nesting sites may be as far as 400 meters (1,312 feet) or more from the aquatic habitat, although usually the distance is much less and generally around 100 meters (328 feet) (Jennings and Hayes 1994; D. Holland pers. comm.; Slavens 1995). Nesting sites typically have a southern or western aspect, with slopes of 0 to 46 percent and compact, dry soils (Holland 1994;

Bury et al 2001). When turtles choose to overwinter in upland habitats, individuals typically leave the aquatic habitat in late fall, moving as much as 500 meters (1,640 feet) from the aquatic habitat (Holland 1994). Turtles typically burrow into duff (leaf litter) and/or soil, where they remain during the winter months (Holland 1994). For reasons not entirely clear, western pond turtles may move into upland habitats for variable intervals at other times of the year, during which times they may be found burrowed into duff or under shrubs (Rathbun et al. 1993).

Raccoons (*Procyon lotor*), bullfrogs (*Rana catesbeiana*), largemouth bass (*Micropterus salmoides*), gray fox (*Urocyon cineroargenteus*), coyote (*Canis latrans*), and feral and domestic dogs (*Canis familiaris*) are known to be major predators of western pond turtles (Holland 1994). Holland (1994) indicates that other known predators include Osprey (*Pandion haliaetus*), Bald Eagle (*Haliaetus leucocephalus*), black bear (*Euarctos americanus*), river otter (*Lutra canadensis*) (Manning 1990), and mink (*Mustela vison*).Numerous other fish, amphibian, bird, and mammal species are suspected to prey on the species (Holland 1994). Raccoons, in particular, are known to depredate nests, sometimes destroying all nests in an entire communal nesting area (D. Holland pers. comm.).

Western pond turtles spend considerable time basking in order to thermoregulate, preferring body temperatures between 24°C and 32°C (75°F and 90°F). Turtles seem to avoid body temperatures above 34°C (93°F) and usually cease basking at body temperatures well below their critical thermal maximum of 40°C (104°F). Individuals often bask above the water level on emergent logs, rocks, rocks, vegetation, or other objects. Turtles may sometimes bask at the surface, however, and sometimes within vegetation, where water temperatures may be 10°C to 15°C (18°F to 27°F) warmer than the water immediately below (Holland 1994). This type of basking may be utilized when air temperatures become too high for aerial basking (D. Holland pers. comm.). Western pond turtles also spend considerable time foraging (Holland 1994). Foraging may occur during the day or night (D. Holland pers. comm.; N. Sisk pers.obs.). Intraspecific (within-species) aggressive interactions, in the form of open-mouth gestures and shoving or bumping to secure positions on basking sites, are also common among western pond turtles (Holland 1994).

Nonnative invasive species are a threat to western pond turtles. Bullfrogs and exotic large predatory fish (e.g., largemouth bass) compete for invertebrate prey with western pond turtles and are known to eat hatchlings and small juveniles. Carp alter or eliminate emergent vegetation required as microhabitat by hatchlings (Holland 1994). Exotic turtles, including painted turtles, snapping turtles, and sliders, may compete with pond turtles for food and basking sites (D. Holland pers. comm.). These exotic turtles also may harbor and transmit diseases, such as upper respiratory diseases, to pond turtles (Holland 1994). Cattle trample and eat aquatic vegetation that serves as habitat for hatchlings and may crush nests. Domestic dogs sometimes kill or injure turtles (D. Holland pers. comm.).

A.4.4 Species Distribution and Population Trends

A.4.4.1 Distribution

The range of the western pond turtle in North America extends primarily from Pacific slopes of western Washington State (where it may now be extinct) south to the San Francisco Bay area, where it intergrades with the southwestern pond turtle (*C. m. pallida*) (Stebbins 2003). The range of the southwestern pond turtle (which does not occur in the Plan Area) extends from the zone of intergradation with the western pond turtle in central California, south to Baja California Norte, Mexico. Outside California, occurrences east of the Pacific crest include the Truckee, Carson, and

East Walker Rivers in Nevada; Drews Creek in Lake County, Oregon; the Canyon Creek area in Lake County, Oregon; and introduced occurrences along the Deschutes River at Bend in Deschutes County, Oregon (Jennings and Hayes 1994; Stebbins 2003). In California, the western pond turtle ranges primarily from Pacific slopes along the Oregon-California state boundary south to the San Francisco Bay area (Stebbins 2003). Occurrences east of the crest of the Sierra Nevada Mountain Range include Susanville in Lassen County (Stebbins 2003). Molecular analyses place western pond turtles into four distinct groups, or clades, which include (1) a Northern clade extending from Washington south to San Luis Obispo County, California, west of the Coast Ranges; (2) a San Joaquin Valley clade from California's Great Central Valley; (3) a Santa Barbara clade from California's Santa Barbara and Ventura counties; and (4) a Southern clade occurring south of the Tehachapi Mountains and west of the Transverse Range south to Baja California, Mexico (Spinks and Shaffer 2005).

Queries conducted in January 2008 of the collection database of the California Academy of Sciences (2008) yielded seven Yolo County records of western pond turtles, all from 1997. Two of those records were from Davis Creek, near Davis Creek Reservoir in western Yolo County. The remaining five records were from the University of California (UC) Davis Arboretum (n = 1) and Arboretum Waterway (n = 4). Spinks et al. (2003) estimate a naturally occurring population of 53 individuals (95 percent CI = 48, 66) within the Arboretum Waterway. A similar query of records of the Museum of Vertebrate Zoology (2008) in Berkeley yielded no record of the western pond turtle in Yolo County. The California Natural Diversity Database (CNDDB) (2007) lists one record from 1990 of multiple western pond turtle individuals along Putah Creek and an unnamed tributary. This site is located less than 1.6 kilometers (1 mile) south-southeast of Winters, along the southern boundary of Yolo County. The CNDDB reports another occurrence from 2005 within Cache Creek, extending for 5.3 miles between Camp Haswell to an upper regional park, northwest of Capay Valley. A healthy population is also present at the Cache Creek Nature Preserve just west of Woodland (Spinks pers. comm.) Jennings and Hayes' (1994) distribution map shows one other extant occurrence from near the northeast corner of Yolo County and three extant occurrences from the Sacramento River Basin, along the southeastern boundary of Yolo County. At least three western pond turtles were observed within the Willow Slough Bypass between County Road 104 and County Road 105 during 2007 (E. Hansen unpublished notes). No other records from Yolo County, either extant or extirpated, were discovered.

More recent observations of western pond turtle have been made by Whisler (pers. comm., 2015). These include the following:

- Sacramento River at Gray's Bend (planning unit 12). Western pond turtle observe at Gray's Bend in1983, and were repeatedly observed through 2012.
- Putah Creek Riparian Reserve at UC Davis (between the University Airport and the Old Davis Road Bridge: planning unit 9). Western pond turtles observed throughout this area in 2014.
- Putah Creek Sinks (2010 and 2011) in the Yolo Bypass Wildlife Area: planning unit 18). Western pond turtles observed in the Putah Creek Sinks along with red-eared sliders and American bullfrogs.
- Lower Willow Slough area (planning unit 11): One adult western pond turtle observed sunning in the Conaway Ranch Water Delivery Canal at Yolo CRs 104 and 27 on March 27, 2010. The area is dominated by rice.
- Sacramento River Delta (planning unit15): Western pond turtles observed in Babel Slough and Winchester Lake during 2015. They probably occur in Elk Slough as well.

- West Sacramento (planning unit 21). Several western pond turtles in the borrow sloughs near the Water Treatment Plant south of Burrows Road in 2009.
- City Davis (planning unit 20). Several western pond turtles observed at the storm water detention basins and other ponds in Davis (West Davis Pond) and North Davis Ponds (Northstar Park Pond and Julie Partansky Pond). Red-eared sliders and American bullfrogs have also been observed at these ponds and are breeding successfully.

A.4.4.2 Population Trends

Populations in Washington State, where the species may be extinct (Stebbins 2003), have likely suffered the most. Stable populations remain in southern Oregon (D. Holland pers. comm.); however, northern Oregon populations have suffered severe declines (Hays et al. 1999), and most populations throughout the range have exhibited some declines (Holland and Bury 1998; D. Holland pers. comm.).

In California, Jennings and Hayes (1994) consider the western pond turtle as endangered from the Mokelumne River south and threatened elsewhere within the state. Loss of habitat is the most significant factor in western pond turtle declines. Over 90 percent of the historical wetlands in California have been drained, filled, or diked to support agricultural and urban development (Frayer et al. 1989). Many populations throughout California are heavily adult-biased (D. Holland pers. comm.), an indication that little recruitment is occurring within those populations. In the Central Valley, pond turtles were exploited for food from the 1890s to the 1920s, which is believed to have played an important role in the declines in the San Francisco area and Central Valley (Storer 1930; Hays et al. 1999).

It is likely that the western pond turtle once occurred in a relatively continuous distribution within suitable habitat in Yolo County, although there is no known site in the county where extirpation of a population has occurred. The population at the UC Davis Arboretum is characterized by a demographic profile characteristic of senescing populations, but has been supplemented by at least 33 captive-hatched individuals since 1996 (Spinks et al. 2003). Because the oldest record obtained from the County is from 1990, status changes that may have occurred prior to 1990 would not be evident from an examination of existing records. Moreover, although no extirpations have been recorded at any known occupied sites in Yolo County, recent survey data could not be located, and data on population trends at those sites are lacking. Therefore, with the exception of the UC Davis Arboretum, current status and population trends of the western pond turtle within the Plan Area are unknown.

A.4.5 Threats to the Species

The most significant threats to the western pond turtle are the continuing loss, degradation, and fragmentation of occupied habitats (D. Holland pers. comm.). Agricultural-related disturbances to wetlands and streams such as changes in the hydrological regime (e.g., water diversions) and removal of aquatic vegetation can render such wetlands unsuitable for pond turtles (D. Holland pers. comm.). The destruction of upland habitats comprising communal nesting areas for agricultural or urban development can result in significant adverse consequences on recruitment for many individuals or an entire population (D. Holland pers. comm.). Water releases from reservoirs, which alter the natural hydrologic regime, may adversely affect downstream habitat by eliminating or altering basking sites, refugia, foraging areas, and hatchling microhabitat (Holland 1991; Hays et al. 1999; U.S. Fish and Wildlife Service [USFWS] 1999). The potential transmission of parasites and diseases from exotic turtle species is a serious concern (Holland 1994; Jennings and Hayes 1994; Hays et al. 1999). Exotic turtles released into the wild typically originate from pet stores, where they are often kept in common containers under unsanitary conditions. When reared under such conditions, the potential for harboring and transmitting exotic pathogens and parasites is greatly increased when these diseased or parasite-ridden turtles are released into habitats occupied by pond turtles. Other threats include collection of individuals for the pet trade and shooting or other means of indiscriminate killing by humans (Holland 1994). Extended drought and associated fire can also result in significant mortality of western pond turtles (Holland 1991). Holland (1994) indicated that mortality caused by automobile strikes probably matches or exceeds mortality from most other anthropogenic sources.

Jennings and Hayes (1994) consider the variation in nesting location in response to variation in habitat, movement responses to habitat change, patterns of movement in the absence of change, and recolonization ability in structurally different habitats to be the most significant data gaps for the species. The lack of data on these parameters led Rathbun et al. (1992) to recommend protecting at least 500 meters (1,640 feet) from known occupied aquatic habitat to avoid impacts to nesting habitat. No recovery plan has been prepared for California populations of western pond turtles because the species is not listed, but the species is included among the recovery goals and objectives contained in the USFWS's (1999) *Draft Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*)*, a species that shares the same wetland habitat types as the western pond turtle. The Plan does not propose any conservation measures designed to benefit the western pond turtle exclusively; however, recovery actions (e.g., habitat protection and restoration) undertaken in the Plan are expected to provide secondary benefits to the species.

Several conservation measures should be implemented in areas where the western pond turtle is known to occur. Populations of exotic predators or competitors, such as bullfrogs, large fish (e.g., largemouth bass), and turtles, should be controlled in habitats occupied by western pond turtles; and efforts to prevent their spread or introduction should be undertaken throughout the Plan Area. Controlling population size and spread of exotic wildlife within Yolo County could also reduce the transmission of infectious diseases to pond turtle populations. Protecting suitable nesting habitat, especially known historical nesting sites, is crucial. Jennings and Hayes (1994) recommended fencing off corridors between aquatic habitats and nesting habitat, and around nesting habitat, in a manner that allows turtle movement to and from nesting areas and prevents trampling of nests during incubation. To reduce the incidence of mortality caused by automobile strikes, the construction of new roads near occupied western pond turtle habitat should be avoided when possible. Maintaining a natural flow regime within lotic habitats occupied by western pond turtles is also of considerable importance in maintaining and improving existing habitat conditions. Considering the abundance of suitable aquatic habitat, western pond turtles may be more widely distributed within the Plan Area than indicated by existing occurrence records.

A.4.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.4.3, *Habitat Requirements and Ecology* (Figure A-4). The model parameters include the following.

- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., CNDDB, California Academy of Sciences Herpetology Department Collection Catalog).
- Other Unmapped Incidental Sightings Where Species is Known to Occur:

Unmapped Incidental Sighting	Source
Willow Slough Bypass between County Road 104 and County Road 105	Hansen pers. comm.
Cache Creek Nature Preserve just west of Woodland	Spinks pers. comm.

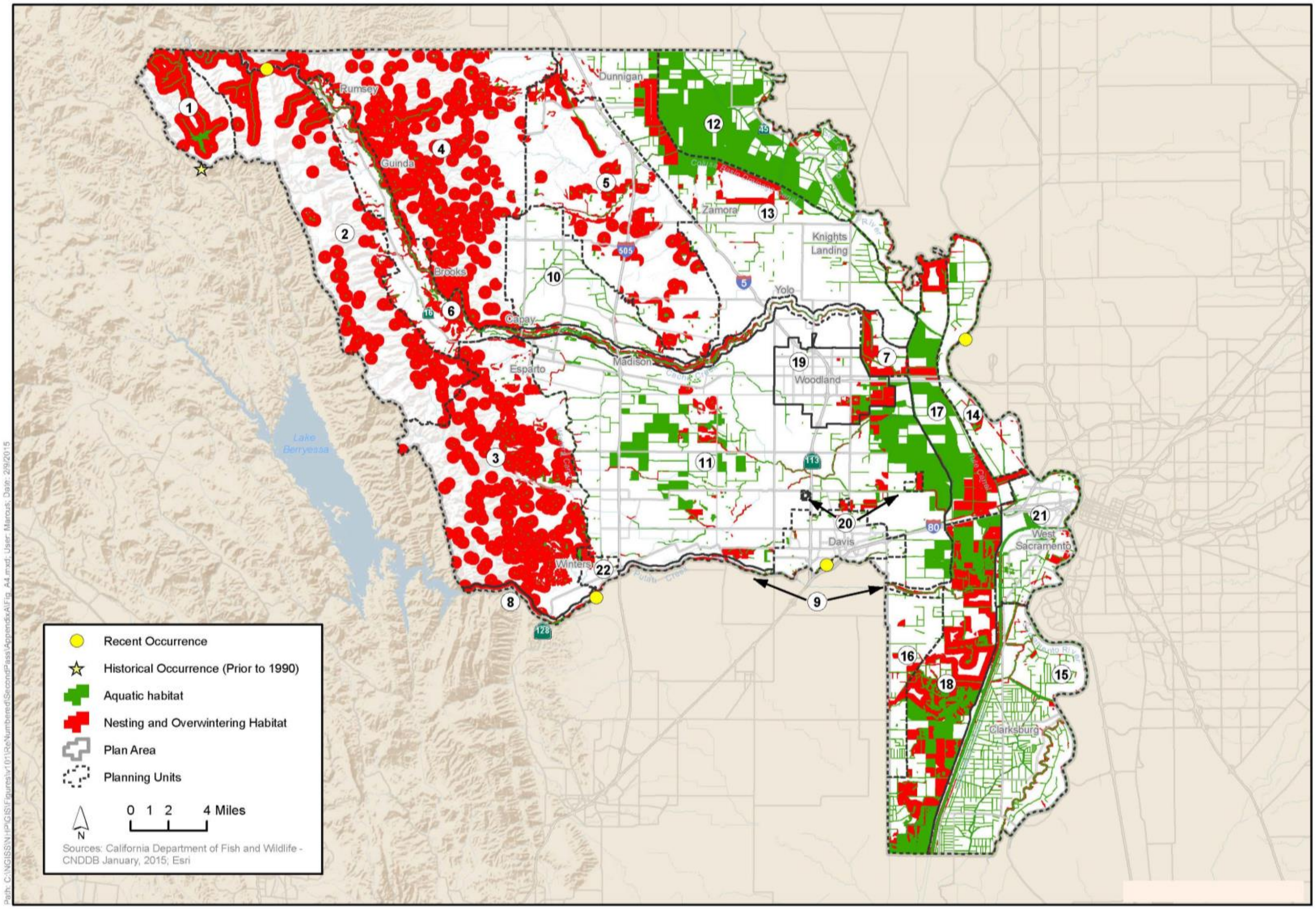
- Aquatic Habitat: This habitat includes all potentially suitable aquatic habitat and was modeled by selecting all mapped land cover types as listed below and by selecting and buffering 10 feet all perennial streams from the National Hydrography Dataset (Ernst et al. 1994) and perennial ponds in the Yolo NHP geographic information system (GIS) database set. Because the water land cover type includes water in small agricultural water conveyance channels that does not support habitat, the model overestimates the extent of this habitat type within the Valley Landscape Unit.

A.4.6.1 Aquatic Habitat – Vegetation Types

- Water
- Bulrush – Cattail Wetland Alliance
- Bulrush – Cattail Fresh Water Marsh Not Formally Defined (NFD) Super Alliance
- Alkali Bulrush – Bulrush Brackish Marsh NFD Super Alliance
- Rice

Nesting and Overwintering Habitat: This habitat includes all potentially suitable nesting habitat. This habitat was modeled by selecting all natural vegetation types that occur within 1,312 feet of aquatic habitat (maximum distance nest can be from aquatic habitat) (Jennings and Hayes 1994; D. Holland pers. comm.; Slavens 1995; Bury et al. 2001). This habitat also includes all potentially suitable overwintering habitat outside of the nesting habitat. This habitat was modeled by selecting all natural vegetation types that occur between 1,312 feet and 1,640 feet from aquatic habitat (maximum distance of overwintering from aquatic habitat) (Holland 1994). Note that nesting habitat may also be used as overwintering habitat. Both modeled nesting and overwintering habitat exclude urban and agriculture vegetation types.

Figure A-4. Western Pond Turtle Modeled Habitat and Occurrences



A.5 Giant Garter Snake (*Thamnophis gigas*)

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A.5.1 Listing Status

Federal: Threatened.

State: Threatened.

Recovery Plan: Draft Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*) (USFWS 1999).

Revised Draft Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*) (USFWS 2015)

A.5.2 Species Description and Life History

The giant garter snake (*Thamnophis gigas*) is an aquatic snake endemic to the Central Valley of California. Described as among California's most aquatic garter snakes (Fitch 1940), giant garter snakes are associated with low-gradient streams, and valley floor wetlands and marshes; they have adapted successfully to regions of rice agriculture. Giant garter snakes are one of the largest snakes in the genus *Thamnophis*. A sexually dimorphic species, females can reach sizes in excess of 1 meter (3.3 feet) and 850 grams (1.87 pounds), while proportionally smaller males seldom exceed 250 grams (0.55 pound). Giant garter snakes possess a dark brown or olive background color separated by light-colored longitudinal stripes. For this species, coloration is geographically and individually variable. Snakes from the San Joaquin Valley region may exhibit a black-checkered pattern along the back and sides, and often lack a distinct dorsal stripe; while snakes from the Sacramento Valley region are typically darker, with a complete dorsal stripe that varies from bright yellow to orange or dull brown. Originally considered a subspecies of *Thamnophis ordinoides* (Fitch 1940), the giant garter snake has undergone a lengthy series of taxonomic revisions, finally being accorded full species status based on morphological and distribution data in the late 1980s (Rossman and Stewart 1987), a classification later confirmed through genetic analyses (Paquin 2001; Paquin et al. 2006).

Upon emerging from overwintering sites, male giant garter snakes immediately disperse in search of mates and will continue breeding from March into early May. Female giant garter snakes brood young internally, giving birth to live young from late July through early September (Hansen and Hansen 1990). Young immediately disperse and seek shelter to absorb their yolk sacs, after which they molt and begin feeding on their own. Brood size ranges from 10 to 46 young, with a mean of 23.1 (n=19) (Hansen and Hansen 1990). Averaging 3 to 5 grams (0.11 to 0.18 ounce) with a snout-to-vent length of approximately 20.6 centimeters (8.1 inches), young giant garter snakes will double their size within their first year (Hansen and Hansen 1990; U.S. Fish and Wildlife Service [USFWS] 1999). Sexual maturity probably averages three years in males and five years in females (G. Hansen pers. comm.; USFWS 1999).

Giant garter snakes are strongly associated with aquatic habitats, typically overwintering in burrows and crevices near active season foraging habitat (Hansen 2004a; Hansen 2004b). Individuals have been noted using burrows as far as 50 meters (164 feet) from marsh edges during the active season, and retreating as far as 250 meters (820 feet) from the edge of wetland habitats while overwintering, presumably to reach hibernacula above the annual high water mark (Hansen 1986; Wylie et al. 1997; USFWS 1999).

Changing agricultural regimes, development, and other shifts in land use create an ever-changing mosaic of available habitat. Giant garter snakes disperse in response to these changes in order to find suitable sources of food, cover, and prey. Connectivity between regions is therefore extremely important for providing access to available habitat and for genetic interchange. In an agricultural setting, giant garter snakes rely largely upon the interconnected network of canals and ditches that provide irrigation and drainage to provide this connectivity. The canals and ditches within the Plan Area likely serve an important role in giant garter snake movement.

Data based on radiotelemetry studies show that home range varies by location, with median home range estimates varying between 9.2 hectares (23 acres) (range 4.2 to 82 hectares [10.3 to 203 acres], n=8) in a semi-native perennial marsh system and 53.2 hectares (131 acres) (range 1.3 to 1,330 hectares [3.2 to 2,792 acres], n=29) in a managed refuge (USFWS 1999).

A.5.3 Habitat Requirements and Ecology

Habitats occupied by giant garter snakes typically contain permanent or seasonal water, mud bottoms, and vegetated dirt banks (Fitch 1940; Hansen and Brode 1980). Abundances and densities of giant garter snakes vary with context of habitat; they are lowest in seasonal/managed marshes (dry in summer, flooded in winter for waterfowl habitat), greatest in natural marshes, and intermediate in rice fields (Wylie et al. 2012). Prior to reclamation, these wetlands consisted of freshwater marshes and low-gradient streams. In some rice-growing areas, giant garter snakes have adapted to vegetated, artificial waterways and associated rice fields (Hansen and Brode 1993) where velocities fall within tolerable limits (E. Hansen in litt. 2009).

This species appears to be mostly absent from permanent waters that support established populations of predatory game fishes; from streams and wetlands with sand, gravel, or rock substrates; and from riparian woodlands lacking suitable basking sites, prey populations, and cover vegetation (Hansen and Brode 1980; Rossman and Stewart 1987; Brode 1988; USFWS 1999). The species may also avoid natural or artificial waterways that undergo routine dredging, mechanical or chemical weed control, or compaction of bank soils (Hansen 1988; Hansen and Brode 1993). Giant garter snakes are associated with aquatic habitats characterized by the following features: (1) sufficient water during the snake's active season (typically early spring through mid-fall) to supply cover and food such as small fish and amphibians; (2) emergent, herbaceous wetland vegetation, such as cattails (*Typha* spp.) and bulrushes (*Scirpus* spp.), accompanied by vegetated banks to provide basking and foraging habitat and escape cover during the active season; (3) upland habitat (e.g., bankside burrows, holes, and crevices) to provide short-term refuge areas during the active season; and (4) high ground or upland habitat above the annual high water mark to provide cover and refuge from flood waters during the dormant winter period (Hansen and Brode 1980; Hansen 1998).

Survivorship and longevity of giant garter snakes are largely unknown, with few quantitative studies of survivorship available for the genus as a whole. One proxy comes from data on individual survival rates for a population of valley garter snakes (*Thamnophis sirtalis fitchi*) at a mountain lake in Northern California. Snakes from this population exhibited first-year survivorship among neonates ranging from 28.7 to 43.0 percent, with a second-year neonate survivorship of 16.4 percent. Survival of yearling snakes was greater than that of juveniles, at 50.8 percent, while survival of snakes two years and older decreased to 32.7 percent (Jayne and Bennett 1990). In a different study, Lind et al. (2005) found that survival estimates for female Pacific coast aquatic garter snakes (*Thamnophis atratus*) in northwestern California was higher than that of males, which is consistent with trends reported for giant garter snakes in the Natomas Basin (Jones & Stokes 2007).

Spending cool winter months in dormancy or periods of reduced activity, giant garter snakes typically emerge from late March to early April and remain active through October; the timing of annual activity is subject to varying seasonal weather conditions. Daily activity consists of emerging from burrows after sunrise, basking to warm bodies to active temperatures, and foraging or courting for the remainder of the day (Hansen and Brode 1993). Like others in their genera, giant garter snakes likely rely on chemical cues to determine reproductive status and to locate mates (Shine et al. 2003; O'Donnell et al. 2004; E. Hansen, pers. obs.). Activity generally peaks during spring emergence and courtship from April into June, whereupon observations of giant garter snakes diminish significantly until a second peak is observed after females give birth during late July into August (Hansen and Brode 1993; Wylie et al. 1997; USFWS 1999; Hansen 2004b). Giant garter snakes then remain actively foraging and occasionally courting until the onset of cooler fall temperatures.

Giant garter snakes feed on small fishes, tadpoles, and small frogs (Hansen 1980; USFWS 1999), specializing in ambushing prey underwater (Brode 1988). Historically, giant garter snakes preyed on native species such as the thick-tailed chub (*Gila crassicauda*) and California red-legged frog (*Rana aurora draytonii*), which have been extirpated from the giant garter snake's current range), as well as the pacific treefrog (*Pseudacris regilla*) and Sacramento blackfish (*Orthodox microlepidus*) (Cunningham 1959; Rossman et al*.* 1996; USFWS 1999). Giant garter snakes now utilize introduced species, such as small bullfrogs (*Rana catesbeiana*) and their larvae, carp (*Cyprinus carpio*)*,* and mosquitofish (*Gambusia affinis*). While juveniles probably consume insects and other small invertebrates, giant garter snakes are not known to consume larger terrestrial prey such as small mammals or birds.

Large vertebrates, including raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), red foxes (*Vulpes vulpes*), gray foxes (*Urocyon cinereoargentius*), river otters (*Lutra canadensis*), opossums (*Didelphis virginiana*), harriers (*Circus cyaneus*), hawks (*Buteo* spp.), herons (*Ardea herodius*, *Nycticorax nyctycorax*), egrets (*Ardea alba*, *Egretta thula*), and American bitterns (*Botaurus lentiginosus*) prey on giant garter snakes (USFWS 1999). In areas near urban development, giant garter snakes may also fall prey to domestic or feral house cats (G. E. Hansen pers. comm.). In permanent waterways, introduced predatory game fishes, such as bass (*Micropterus* spp.), sunfish (*Lepomis* spp.), and channel catfish (*Ictalurus* spp.), prey on giant garter snakes and compete with them for smaller prey (Hansen 1998; USFWS 1993).

Giant garter snakes coexist with the valley garter snake (*Thamnophis sirtalis fitchi*). In limited instances, both may be found together with the mountain garter snake (*Thamnophis elegans elegans*), a subspecies of western terrestrial garter snake, in locations where this species' range extends to the floor of the Central Valley. The extent of competition among these species is unknown but, generally, differences in habitat use and foraging behavior allow their coexistence (C; USFWS 1999).

A.5.4 Species Distribution and Population Trends

A.5.4.1 Distribution

The current known distribution of giant garter snakes is variable, and extends from near Chico in Butte County south to the Mendota Wildlife Area in Fresno County. Occurrences of giant garter snakes are not known from the northern portion of the San Joaquin Valley north to the eastern fringe of the Sacramento-San Joaquin River Delta, where the floodplain of the San Joaquin River is limited to a relatively narrow trough (Hansen and Brode 1980; USFWS 1993). The resulting gap of approximately 100 kilometers (km) (62.3 miles) separates the southern and northern populations, with no giant garter snakes known from the lowland regions of Stanislaus County (California Natural Diversity Database [CNDDB] 2004; Hansen and Brode 1980). Scattered records within the Sacramento-San Joaquin River Delta suggest that giant garter snakes may have occupied this region at one time, but longstanding reclamation of wetlands for intense agricultural applications has eliminated most suitable habitat (CNDDB 2004; Hansen 1986). Recent records within the Sacramento-San Joaquin Delta are haphazard, and repeated surveys have failed to identify any extant population clusters in the region (Hansen 1986; Patterson and Hansen 2002; Patterson 2003). Recent occurrence records indicate that, within this range, garter snakes are distributed in 13 unique population clusters coinciding with historical flood basins, marshes, wetlands, and tributary streams of the Central Valley (Hansen and Brode 1980; Brode and Hansen 1992; USFWS 1999). These populations are isolated, without protected dispersal corridors to other adjacent populations, and are threatened by land use practices and other human activities, including development of wetland and suitable agricultural habitats.

One of these 13 extant giant garter snake populations, the northern Yolo Basin population is distributed along the northeastern edge of the Yolo Basin near the Sacramento River. Yolo County is well within the Central Valley proper and includes the floodplains of the Sacramento River as well as those of Cache, Willow, and Putah Creeks. Upon receding, these creeks may have provided the wetland habitat and prey utilized by giant garter snakes during the spring and summer active season. The historical distribution of giant garter snakes in Yolo County is unclear, however, with the majority of sightings made only in recent decades (Hansen 1986; CNDDB 2007).

Giant garter snakes are documented in two distinct concentrations along the eastern edge of Yolo County (CNDDB 2007; Hansen 2006, 2007a, 2008; Wylie et al. 2004; Wylie and Martin 2005; Wylie and Amarello 2006). The first concentration lies in the northeastern portion of Yolo County, northwest of Knights Landing and in the southern end of the Colusa Basin near Sycamore Slough and the Colusa Basin Drainage Canal. Wylie and Amarello (2006) report a population density in the Colusa Basin Drainage Canal of 20±3 snakes/km during 2006, falling within 2003 and 2004 confidence intervals, noting, however, that local distribution appears to have shifted away from areas formerly in rice production that have either been fallowed or converted to other crop types. The second concentration lies in the east-central portion of Yolo County, with records in the Yolo Bypass east of Conaway Ranch near the Tule Canal, the Willow Slough/Willow Slough Bypass from Conaway Ranch south to the Yolo Wildlife Area, the Davis Wetlands complex south of Conaway Ranch between the Willow Slough Bypass and the Yolo Bypass, the Yolo Wildlife Area along the east edge of the Yolo Bypass west levee, and the adjacent ricelands east of the Yolo Wildlife Area. Surveys conducted in 2005, 2006, and 2007 resulted in captures of 34, nine, and one unique individual(s), respectively, in the Yolo Wildlife Area; eight, 18, and eight unique individuals, respectively, in the adjacent ricelands; and 36 unique individuals (2007 only) in the Davis Wetlands complex (Hansen in. litt. 2006, 2007, 2008). Hansen (2006, 2007a, 2008) reports an even distribution within size classes, estimating local populations ranging from 8 ± 2.6877 (95 percent confidence interval (C.I.) = 7 to 20) to 57 ± 9.53 (95 percent C.I. = 45 to 84) in the Yolo Wildlife Area; 5 ± 0.4932 (95 percent C.I. = 5 to 5) to 17 ± 5.9655 (95 percent C.I. = 12 to 39) in the adjacent ricelands; and from 26 ± 21.2829 (95 percent C.I. = 11 to 120) to 67 ± 59.7094 (95 percent C.I. = 22 to 322) within the Davis Wetlands Complex (Hansen 2006, 2007a, 2008). Queries of the online databases of the California Academy of Sciences (2008) and Museum of Vertebrate Zoology (2008) yielded one additional occurrence record (CAS 178594) situated within downtown Davis; however, the stated location for this record (a frontage road one mile east of the Yolo Causeway) conflicts with the stated coordinates, leaving the true location unclear.

Evidence that giant garter snakes may once have been distributed throughout the easterly reaches of Yolo County is illustrated by reported sightings in portions of Solano County adjacent to Yolo County, in South Fork Putah Creek near Davis, and in the Liberty Farms region of the Yolo Basin. Repeated attempts to assess local distribution suggest that both the Liberty Farms and Putah Creek populations are probably extirpated (Hansen 1986; Wylie and Martin 2005; D. Kelly pers. comm.).

Genetic analyses of tissue samples collected from giant garter snakes in the Yolo Wildlife Area and adjacent ricelands are ongoing. Engstrom (2007) reports that the Yolo Basin population is genetically very similar to those of the Natomas and Middle American Basins, but that genetic diversity within the Yolo Basin is lacking, which is typical of recently colonized populations. Engstrom reports, however, that there appears to be very little gene flow between the Yolo Basin and neighboring populations, and that ongoing migration into the Yolo Basin is not significant.

A.5.4.2 Population Trends and Abundance Estimates

Prior to listing in 1971, giant garter snakes were known from 16 localities, representing nine distinct populations based on available literature and museum records (Hansen and Brode 1980; USFWS 1993). Range-wide status surveys of the giant garter snake conducted during the mid-1970s and 1980s indicate that they have been extirpated from the San Joaquin Valley south of Mendota in Fresno County, an area comprising as much as one-third of the snake's former range (Fitch 1940; Hansen and Brode 1980; Rossman and Stewart 1987; Stebbins 2003). Once plentiful in areas such as Mendota, Los Baños, and Volta, giant garter snakes are now known from only a small number of localities in the southern aspect of their range (USFWS 1999; Dickert 2003; Hansen 2007b). Giant garter snakes have not been documented from Burrell in Fresno County northward to Stockton since prior to 1980 and now appear to be most abundant in regions of the northern Sacramento Valley that are dominated by rice agriculture (USFWS 1993, 1999; CNDDB 2007).

Abundances and densities of giant garter snakes vary with context of habitat; they are lowest in managed seasonal marshes (dry in summer, flooded in winter for waterfowl habitat), greatest in natural marshes, and intermediate in rice fields (Wylie et al. 2011). In general, giant garter snakes select areas with a dense network of canals, often in close proximity to rice agriculture, with a low density of streams and close to open water and wetlands, compared to available environments in the Sacramento Valley (Halstead et al. 2010).

Most density estimates for giant garter snakes have been derived from linear trapping transects along canals, linear wetlands, or ecotones between deep water and upland habitat. Standard survey methodology for giant garter snake entails transects consisting of 50 floating aquatic funnel traps (Casazza et al. 2000) located along the open water/terrestrial or open water/emergent vegetation interface in areas of standing or slow-moving water and, where possible, emergent aquatic vegetation. Traps are spaced approximately 10 meters (33 feet) apart, resulting in traplines of approximately 500 meters (1,640 feet).

Lineal densities of individuals captured per transect (and extrapolated to lineal miles of habitat) can be converted into two-dimensional densities (snakes/acre) in two ways: First, the “area of influence” around a transect may provide a small-scale reference based on the spatial behavior of snakes (Wylie et. al 2010). Thus, a trapline is typically estimated to adequately sample the number of snakes present in an area of 100 meters on either side of the transect, or a total area of 200 meters by 500 meters = 100,000 square meters (approximately 25 acres). Thus, using the “area of influence” approach, snake densities reported per lineal mile are based on a total area of 80 acres.

Secondly, on a landscape scale, the density of the number of snakes captured along lineal structures (e.g., canals, shorelines) is derived from the overall density of conveyances per acre of surrounding habitat. Thus, the number of snakes per lineal mile is multiplied with the number of lineal miles of canal per acre of snake habitat. This measure is perhaps a more meaningful estimator for landscape and population-level measurements of giant garter snake densities in agricultural areas, where rice paddies and conveyance channels are both considered habitat. But such densities are more challenging to derive for more complex natural and restored wetlands, due to the contorted shoreline and the difficulty to delineate habitat in emergent marshes and wetlands.

Hansen and Brode (1993) estimated a local population size of 1,000 snakes per square mile (1.56 snakes per acre) of rice lands based on year-to-year mark recapture rates (U.S. Fish and Wildlife Service 1999). Giant Garter snake population densities (snakes per lineal mile of rice irrigation canal) in Yolo county ranged from 13 (95 percent C.I. = 11 to 32) to 92 (95 percent C.I. = 72 to 135) in the Yolo Wildlife Area; 8 (95 percent C.I. = 8 to 8) to 27 (95 percent C.I. = 19 to 63) in the adjacent ricelands; and from 42 (95 percent C.I. = 18 to 193) to 108 (95 percent C.I. = 35 to 518) within the Davis Wetlands Complex (Hansen in. litt. 2006, 2007, 2008). For the Colusa Drain and adjacent rice habitat, a mean density of 22.6 snakes per lineal mile of survey was determined for three consecutive years (Wylie and Amarello 2008). The U.S. Geological Survey (USGS) (Wylie et al. 2000a, 2000b, 2001, 2002, 2004) reported linear densities in selected trapping areas ranging from 13 (95 percent C.I. = 10–19) to 88 (95 percent C.I. not reported) giant garter snakes per linear mile from 1999 to 2003 in the Natomas Basin. Mean landscape-level densities of giant garter snakes reported from the Natomas Basin (all habitats combined) range from 5.1 to 22.7 giant garter snakes per linear mile (Table 1) and have fluctuated considerably among the years.

Table A-1. Giant Garter Snake Densities (Individuals Captured per Mile Surveyed) Reported in Rice and Other Wetland Habitats from Various Sites in the Sacramento Valley, 1999–2010

Location	Habitat	Individuals Captured	Miles	Individuals per Mile	Reference
Badger Creek (southern Sacramento County)	Natural wetlands	103	0.5	221.0	Wylie et al 2010
Colusa NWR	Managed wetlands	22	1.1	20.2	Wylie et al 2010
Colusa NWR	Restored wetlands				
Gilsizer Slough (Sutter County)	Rice	67	1.8	37.8	Wylie et al 2010
Colusa Drain (2003)	Rice	40	2.4	16.8	Wylie and Amarello 2008
Colusa Drain (2004)	Rice	24	2.4	10.0	Wylie and Amarello 2008
Colusa Drain (2006)	Rice	30	2.4	12.4	Wylie and Amarello 2008
Natomas Basin	Rice	141	4.1	34.1	Wylie et al 2010
Butte and Glenn Counties	Rice	28	3.5	7.5	Wylie et al. 2011
Natomas Basin Average (1999-2004)	All	NA	NA	22.7	Jones and Stokes 2005
Natomas Basin 2009	All	155	19. 3	8.0	Jones and Stokes 2010
Natomas Basin 2010	All	112	22.1	5.1	ICF 2011

Note:
NWR = National Wildlife Refuge.

In general, higher densities of snakes were recorded in linear drainage and irrigation features associated with rice, compared with managed or seasonal marsh habitats (ICF 2011). The availability of managed marsh habitat has been deemed important for giant garter snakes when they emerge from winter dormancy and begin feeding, dispersing, and mating – at which times rice fields and other aquatic habitats are not available (ICF 2011). Core home range size of radio-tagged female garter snakes (Valcarel et al. 2011) were smaller in rice habitats and overlapped considerably more, compared to those in restored wetlands in Gilsizer Slough (Sutter County).

A.5.4.3 Giant Garter Snake Habitat Types and Populations in the Yolo NHP Area

The NHP geospatial database was developed from the California Department of Fish and Wildlife (DFW) Wildlife Habitat Relationships (WHR) database, which identifies vegetation communities according to their function as habitat for the giant garter snake. Aquatic habitat availability is the primary determinant of giant garter snake abundance; therefore, this analysis only considers aquatic habitats as an obligate habitat prerequisite for the species. For the purpose of this analysis, and to facilitate the crosswalk of modeled habitat types with those reported in the literature (e.g., Wylie et al. 2010) aquatic habitat was categorized as follows:

Rice: Rice agriculture has become a major habitat for giant garter snakes in the Central Valley (Hansen and Brode 1993). Within the giant garter snake focal areas of the NHP Plan Area (i.e., Planning Units 11, 12, 13, and 19), rice land habitat is an important element of the species' life history. The primary giant garter snake habitat within rice lands are the conveyance channels and irrigation canals, which provide foraging and movement habitat and which ensure spatial connectivity of habitat and populations within the rice agricultural landscape. Studies indicate that despite the presence of ditches or drains, giant garter snakes will generally abandon aquatic habitat that is not accompanied by adjacent shallow-water wetlands or rice fields (Jones and Stokes 2008; Wylie et al. 2006). Giant garter snakes tend to expand their foraging activities from the canals and ditches into rice fields soon after the rice plants emerge above the water's surface, and they continue to use the fields until the water is drained during late summer or fall (Hansen and Brode 1993). During the winter period, banks along the ditches provide crucial hibernacula that are protected from flooding. Thus, within rice lands, a greater density of canals and irrigation structures is expected to support higher densities of giant garter snakes, due to a greater and more stable prey base and the presence of habitat refugia in times when some canals are dry or during maintenance events. In addition, complex habitat structure providing cover from predation and perhaps locally lower predation rates may also contribute to higher giant garter snake densities. Isolated patches of habitat containing small, discrete snake populations would likely result where this aquatic connectivity is lost.

Wylie et al (2011) provide the currently best available landscape-level estimates of giant garter snake density in rice-dominated agricultural areas, based on captures and recaptures at 44 transects along linear canals within rice fields and in managed wetlands in Butte and Glenn County from 2008 through 2010. To make the results of Wylie et al (2011) more applicable to the rice area in the Plan Area, the total density of snakes per lineal mile of canal habitat from all transects, including those that did not result in snake captures, was calculated. Density estimates ($\bar{x}$ =7.48, sd = 8.10, range = 0 to 19.65) were calculated from data provided by Wylie et al. (2011). These estimates are among the lowest estimates compared to other recent studies in adjacent areas (Table 1), but probably are realistic estimates for a large landscape area, since Wylie's et al. (2011) study included transects that did not yield captures. Wylie et al. (2011) established a lower confidence interval boundary of 0.2 snakes per ha (= 0.49 per acre) at the study site with the lowest overall density of snakes (excluding sites that had no snake captures), which translates into a low estimate of 6.34 snakes/mile for occupied sites. This estimate is also well within the range of data for giant garter snakes in Sacramento Valley (Table 1). An upper estimate of snake density was derived as the mean plus one standard deviation from Wylie et al. (2011). Thus, a high estimate of the area-wide density of snakes was calculated as ($\bar{x}$ + sd) = 15.58 snakes/mile. The distribution of giant garter snakes in the Plan Area is probably clumped and likely disjunct (Glenn Wylie, pers. comm.), with large areas of unoccupied habitat interspersed by patches of higher population densities. Such distributions have been related to historical (Paquin et al. 2006) and spatial dynamics of habitat manipulations and conveyance management (Hansen and Brode 1983). In addition, the presence and abundance of prey and non-native and native predators (e.g., bull frogs, predatory fish, egrets, and herons) may also affect the metapopulation structure of giant garter snakes in the Plan Area.

Based on 117 miles of drainage canals within rice lands in the Colusa Basin Subpopulation (Planning Units 12 and 13) and 32 miles in the Willow Slough/Yolo Bypass Subpopulation (Planning Units 11 and 19), and the conservative mean estimate of 7.48 snakes per lineal mile of canals, which takes into account currently unoccupied habitat, a total estimate of giant garter snakes for the 29,470 acres of riceland of the relevant Planning Units is 1,122 giant garter snakes, or 0.039 snakes per acre of rice. This density estimate compares well with the landscape level estimate of 0.41 snakes per acre derived from Wylie et al. (2010). Although the habitat model for giant garter snake also included irrigated croplands and seasonal managed wetlands, for the purpose of estimating snake population size, these habitat types were assumed not to provide year-round stable habitat and thus were not included for the calculation of a population estimate.

Seasonal/Managed Wetlands: Most emergent wetland types and vegetation associations in the in the Colusa Basin Subpopulation (Planning Units 12 and 13) and the Willow Slough/Yolo Bypass Subpopulation (Planning Units 11 and 19) are considered marginal habitat, as they are flooded primarily during winter only. Hence, they may not provide the warm water summer habitat for giant garter snake but rather lower-quality winter cold water foraging habitat and put snakes at risk in their winter hibernacula. There are 4,490 acres of managed wetlands within the NHP giant garter snake conservation focal areas in Planning Units 11, 12, 13, and 19 NHP habitat mapping units classified as Freshwater Emergent Vegetation. Based on visual estimates from summer aerial imagery (September 2011), approximately 3,600 acres of these mapped seasonal wetlands are winter flooded (80 percent of fresh emergent wetland cover type), but considerable inaccuracies and resolution incongruence exist. No densities of giant garter snakes were assigned to these acreages because they are not expected to provide summer aquatic habitat for the species.

Summer Flooded/Perennial Wetlands: Wetlands that are flooded during summer or are perennial provide the highest quality habitat for giant garter snake. Since existing summer-flooded, perennial or natural wetlands could not be distinguished from the fresh emergent wetland data layer in the NHP geographic information system (GIS) database, it was necessary to estimate the proportion of summer flooded wetlands that potentially provide garter snake habitat functions. The percentage of habitat that is summer flooded managed/seasonal wetlands was identified by overlaying the NHP habitat GIS layer for managed wetlands and estimating the proportion in each parcel that could be considered summer flooded or perennial wetland from 2011 aerial imagery. Approximately 900 acres were considered summer flooded permanent or seasonal wetlands that may be expected to provide habitat functions for giant garter snake.

Only one local density estimate (i.e., 20.2 snakes/mile of transect) exists for giant garter snakes in managed wetlands from a study on the Colusa NWR, which was translated into a density of 0.25 individuals/acre (based on a 100 m buffer on each side of the transect as described by Wylie et. al 2011). Based on a density of 0.25 snakes per acre, the population estimate for the estimated summer flooded or perennial wetlands in the conservation focal areas is 900x.25 = 225 snakes.

Restored Wetlands: Wetlands restored specifically for giant garter snake habitat provide an opportunity to produce high densities of snakes. Ideally, these habitats function as natural perennial wetlands and provide year-round habitat function for the species. Studies of restored wetlands specifically as habitat for giant garter snake are only just beginning. Local density estimates for giant garter snakes in restored wetlands in the Colusa Wildlife Refuge range from 48 to 194 snakes per mile depending on the trapping location on the Refuge, similar to values in a previous year (87-169/mile) (Wylie et al. 2002). Framed by a minimum density estimate of 0.063 snakes/acre (or 5.8 snakes/mile) (ICF 2010, 2011) and a conservative maximum density value of 0.46 snakes/acre (37.6 snakes/mile) (Wylie et al. 2010), an average landscape-level density estimates from all studies (except natural wetlands) (Wylie 2010) results in a mean of 0.21 snakes/acre of restored wetland (sd=0.137), with a low to high estimate ($\bar{x} \pm sd$) of 0.073 to 0.348 snakes/acre.

A.5.4.4 Plan Area Population Estimate Summary

No systematic density evaluation or survey of giant garter snakes in the NHP Plan Area has been conducted to date. Thus, an estimate of a total population size of giant garter snakes cannot be derived based on systematic demographic studies. Instead, landscape-level densities observed in multiple studies were used to estimate population sizes, based on the acreage or spatial extent of the respective habitat type. Population estimation was separated by habitat type, based on the different observed densities of giant garter snakes in rice and seasonal/managed wetlands. The distribution of giant garter snake aquatic habitat types by Planning Unit and subpopulation is presented in Table 2, and resulting population estimates are presented in Table 3.

Table A-2. Acreage of Giant Garter Snake Aquatic Habitat

	Colusa Basin Subpopulation			Willow Slough/Yolo Bypass Subpopulation[a]			
Aquatic Habitat Type	**PU 12**	**PU13**	**Subtotal**	**PU11**	**PU19**	**Subtotal**	**Total**
Rice – miles of canals	113	4	117	28	4	32	149
Rice - acreage	20,045	1,592	21,637	6,535	1,298	7,833	29,470
Managed/seasonal wetland	840	3,063	3,903	587	0	587	4,490
Managed summer flooded and perennial wetlands	168	612.6	780.6	117.4	0	117.4	898
Total acreage	**20,885**	**4,655**	**25,540**	**7,122**	**1,298**	**8,420**	**33,960**

[a] Excluding the Yolo Bypass (Planning Units 17 and 18) within which no conservation actions are proposed by the Implementing Entity.

Table A-3. Giant Garter Snake Population Estimate by Subpopulation and Habitat Type

	Colusa Basin Subpopulation			Willow Slough/Yolo Bypass Subpopulation[a]			
Aquatic Habitat Type	**PU 12**	**PU13**	**Subtotal**	**PU11**	**PU19**	**Subtotal**	**Total**
Rice	845	30	875	209	30	239	1115
Managed/seasonal wetland – winter flooded	0	0	0	0	0	0	0
Managed summer flooded and perennial wetlands	42	153	195	29	0	29	225
Total number of snakes	**887**	**183**	**1,070**	**239**	**30**	**269**	**1,339**

[a] Excluding the Yolo Bypass (Planning Units 17 and 18) within which no conservation actions are proposed by the Implementing Entity.

A.5.5 Threats to the Species

Continued loss of wetland or other suitable habitat resulting from agricultural and urban development constitutes the greatest threat to this species' survival. Conversion of Central Valley wetlands for agriculture and urban uses has resulted in the loss of as much as 95 percent of historical habitat for the giant garter snake (Wylie et al. 1997). In areas where the giant garter snake has adapted to agriculture, maintenance activities such as vegetation and rodent control, bankside grading or dredging, and discharge of contaminates, threaten their survival (Hansen and

Brode 1980; Brode and Hansen 1992; Hansen and Brode 1993; USFWS 1999; Wylie et al. 2004). Within agricultural areas, giant garter snakes are also threatened by fluctuations in the amount and locations of rice production, and by the conversion of rice lands to other crop types. Giant garter snakes are subject to mortality through loss or degradation of habitat; predation of juvenile giant garter snakes by introduced predators; elimination of giant garter snakes or prey species by pesticides and other toxins; road mortality; maintenance and modification of agricultural ditches, drains, and flood control systems; and flooding (Hansen 1986; USFWS 1999). Snakes remaining in rice fields are subject to threats from mechanical harvesting, including disrupted foraging, thermoregulating, or direct mortality; the extent of these threats is unknown (USFWS 2006). For many snake species, chemoreceptivity plays an integral role in habitat (Clark 2004) and mate selection (Shine et al. 2003; O'Donnell et al. 2004) in snakes' ability to navigate through their habitat, find overwintering sites, and locate mates. In developed areas, threats of vehicular mortality also are increased. Paved roads likely have a higher rate of mortality than dirt or gravel roads due to increased traffic and traveling speeds, and as many as 31 giant garter snake traffic mortalities have been reported during a four-year period in the Natomas Basin (Hansen and Brode 1993).

The loss of wetland habitat is compounded by elimination or compaction of adjacent upland and associated bankside vegetative cover, as well as water fouling; these conditions are often associated with cattle grazing (Thelander 1994). While cattle grazing and irrigated pastures may provide the summer water that giant garter snakes require, high stocking rates may degrade habitat by removing protective plant cover and underground and aquatic retreats such as rodent and crayfish burrows (Hansen 1986; USFWS 1999). Studies of wandering garter snakes (*Thamnophis elegans vagrans*) in Northern California have shown population numbers to be much higher in areas where grazing was excluded (Szaro et al. 1985). Radiotelemetry studies in perennial wetlands where grazing was differentially excluded show that giant garter snakes avoid areas where grazing is frequent (Hansen 2002). Cattle grazing may, however, provide an important function in controlling invasive vegetation that can compromise the overall value of wetland habitat (Hansen 2002).

Giant garter snakes are also threatened by the introduction of exotic species. Examinations of gut contents confirm that introduced bullfrogs (*Rana catesbeiana*) prey on juvenile giant garter snakes throughout their range (Treanor 1983; Dickert 2003; Wylie et al. 2003). While the extent of this predation and its effect on population recruitment is poorly understood, estimates based on preliminary data from a study conducted at Colusa National Wildlife Refuge suggests that 22 percent of neonate (newborn) giant garter snakes succumb to bullfrog predation (Wylie et al. 2003). Other studies of bullfrog predation on snakes have documented bullfrogs ingesting other species of garter snakes up to 80 centimeters (31.5 inches) long, resulting in a depletion of this size-class within the population (Bury and Wheelan 1984). Introduced predatory game fishes, such as black bass (*Micropterus* spp.), sunfish (*Lepomis* spp.), and channel catfish (*Ictalurus* spp.), prey on giant garter snakes and compete with them for smaller prey (Hansen 1988; USFWS 1993).

Selenium contamination and impaired water quality have been identified as a threat to giant garter snakes, particularly in the southern portion of their range (USFWS 1999). While little data are available regarding the effects of specific contaminants, the bioaccumulative properties of selenium in the food web have been well documented in the Kesterson National Wildlife Refuge area (Saiki and Lowe 1987; Ohlendorf et al. 1988; Saiki and May 1988; Saiki et al. 1991; USFWS 1999).

Recent findings demonstrate that giant garter snakes are extant within Yolo County (CNDDB 2007; Hansen 2006, 2007a, 2008; Wylie et al. 2003, 2004, 2006). However, little is known of their regional distribution or their population status throughout the remainder of Yolo County. While some estimates are available (e.g., Hansen and Brode 1993; Wylie et al. 2004), giant garter snake population sizes and densities are not well known throughout their range. Differential dispersal and home range patterns between males and larger females who spend the majority of the active season gestating young are not reported. Lifetime dispersal patterns of both neonates and adults of this species are unknown.

Until uncertainties regarding population structure, population dynamics, and the strength, frequency, and direction of environmental fluctuations and edge effects are resolved, it is impossible to establish population numbers as a delisting criterion for this species (USFWS 1999). Current criteria for assessing the species' status include the quality and distribution of available habitat and the presence of both young and adults, indicating a stable population structure within known populations (USFWS 1993, 1999).

Throughout the Central Valley, GIS modeling has been used to analyze microhabitat characteristics and suitability of aquatic and upland habitats for the giant garter snake (Hansen 2003). Modeling includes the use of 23 distinct habitat variables correlated with giant garter snake life history and ecological requirements. Data are maintained within a comprehensive database, which is updated in response to changes in land use or habitat management. Coverage currently includes all navigable waterways within California Department of Boating and Waterways Aquatic Weed Control Division's Water Hyacinth and *Egeria densa* Control Program service areas, spanning the Central Valley from the Port of Sacramento in Sacramento County south to the Mendota Pool area in Fresno and Madera Counties, and in select areas within Sacramento, Sutter, and Yuba Counties.

In the Central Valley, rice fields have become important habitat for giant garter snakes. Irrigation water typically enters the rice lands during April along canals and ditches. Giant garter snakes use these canals and their banks as permanent habitat for both spring and summer active behavior and winter aestivation. Where these canals are not regularly maintained, lush aquatic, emergent, and streamside vegetation develops prior to the spring emergence of giant garter snakes. This vegetation, in combination with cracks and holes in the soil, provides much-needed shelter and cover during spring emergence and throughout the remainder of the summer active period.

Rice is planted during spring, after the winter fallow fields have been cultivated and flooded with several inches of standing water. In some cases, giant garter snakes move from the canals and ditches into these rice fields soon after the rice plants emerge above the water's surface, and they continue to use the fields until the water is drained during late summer or fall (Hansen and Brode 1993). It appears that the majority of giant garter snakes move back into the canals and ditches as the rice fields are drained, although a few may overwinter in the fallow fields, where they hibernate within burrows in the small berms separating the rice checks (Hansen 1998).

While within the rice fields, the snakes forage in the shallow warm water for small fish and the tadpoles of bullfrogs and treefrogs. For shelter and basking sites, giant garter snakes utilize the rice plants, vegetated berms dividing the rice checks, and vegetated field margins. Gravid (pregnant) females may be observed within the rice fields during summer, and at least some giant garter snakes are born there (Hansen and Brode 1993; Hansen 1998). Suitability of rice fields for giant garter snakes may vary by crop type. Wild rice species (e.g., *Zizania* spp.) may reach 5 to 6 feet in height, obscuring sunshine and limiting opportunities for snakes to thermoregulate. White or brown rice species are shorter in stature, providing superior basking opportunities.

Water is drained from the fields during late summer or fall by a network of drainage ditches. These ditches are sometimes routed alongside irrigation canals and are often separated from the irrigation canals by narrow vegetated berms that may provide additional shelter. Drainage typically occurs one month prior to harvest for white or brown rice and two to three weeks prior to harvest for wild rice crops (D. Sills pers. comm.). Remnants of old sloughs also may remain within rice-growing regions, where they serve as drains or irrigation canals. Giant garter snakes may use vegetated portions along any of these waterways as permanent habitat. Studies indicate that despite the presence of ditches or drains, giant garter snakes will generally abandon aquatic habitat that is not accompanied by adjacent shallow-water wetlands (Hansen 2008, Jones and Stokes 2008, Wylie et al. 2006), underscoring the important role that this crop plays in this species' life history.

Central Valley wetland conservation occurs through a combination of both public and privately managed refuges, mitigation banks, and duck clubs, creating a large network of wetland preserves throughout the historical range of the giant garter snake. A large percentage of these wetland conservation efforts, however, are geared toward waterfowl management, often placing greater emphasis on winter water rather than the summer water upon which giant garter snakes depend (G. Hansen pers. comm.; USFWS 1999). With proper consideration given to design, location, and management, these efforts might also significantly benefit the giant garter snake and other wetland-dependent species (USFWS 1999).

Under the 1999 *Draft Recovery Plan for the Giant Garter Snake* (*Thamnophis gigas*), initiation of the delisting process is anticipated by 2028, given that defined recovery criteria are adequately met. To accomplish the recovery of this species, the U.S. Fish and Wildlife Service emphasizes habitat protection; public participation, outreach, and education; habitat management and restoration; surveying and monitoring; and continued research (USFWS 1993).

A.5.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.5.3, *Habitat Requirements and Ecology* (Figure A-5). The model parameters were limited to regions east of Highway 113 and Interstate 5 and include the following.

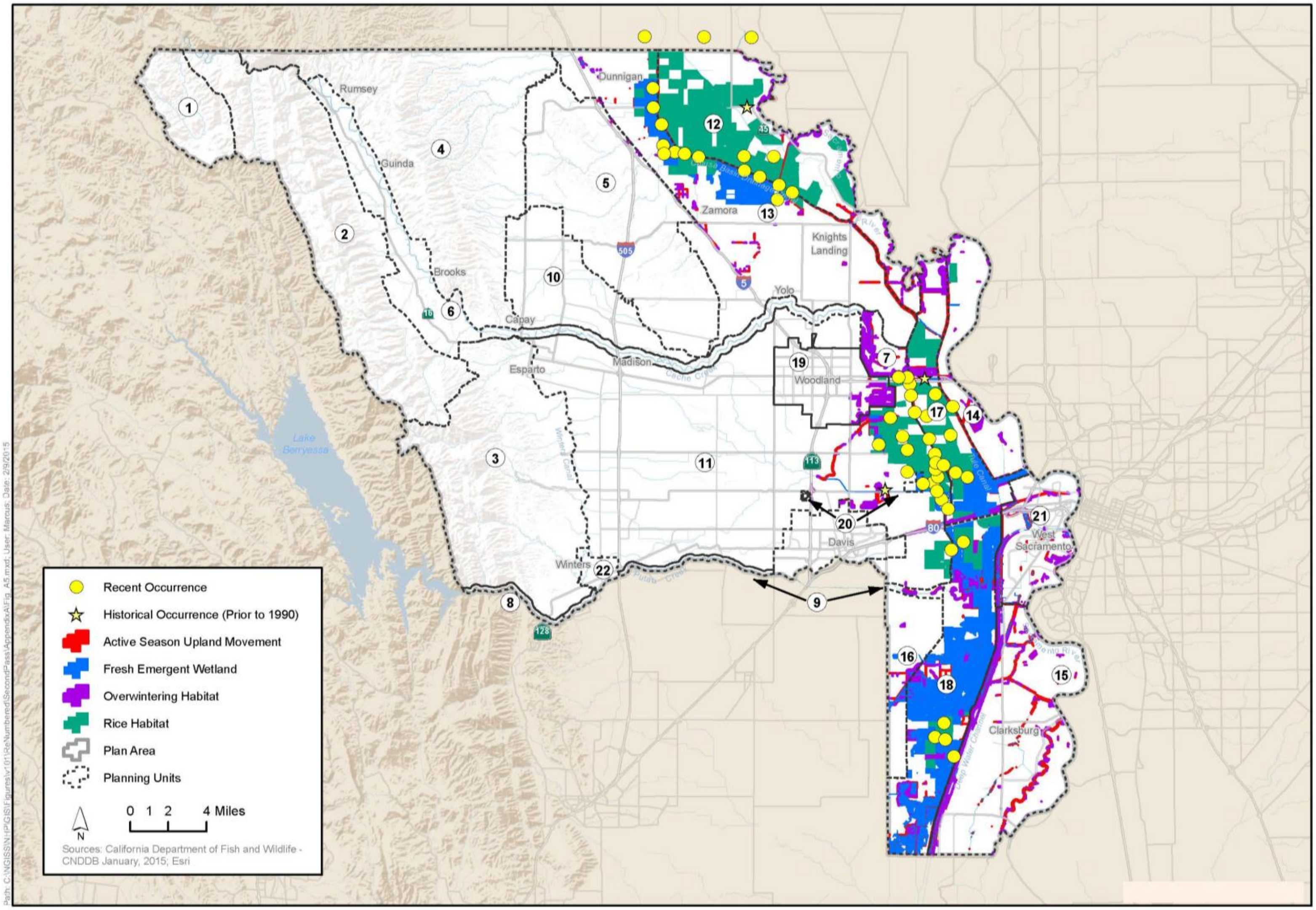
- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., California Natural Diversity Database [CNDDB]; U.S. Geological Survey; Eric Hansen).
- Rice Habitat: Based on the known distribution of giant garter snake within the Plan Area (Figure A-19). This habitat includes all mapped rice land that occur east of Highway 113 and east of Interstate 5 from its junction with Highway 113. Mapped rice land includes associated water conveyance channels.
- Fresh Water Emergent Habitat: Based on the known distribution of giant garter snake within the Plan Area (Figure A-19) this habitat includes all mapped fresh emergent wetland that occurs east of Highway 113 and east of Interstate 5 from its junction with Highway 113. Freshwater emergent habitat is generally seasonal or managed wetlands that may support inclusions of perennial wetland.
- Active Season Upland Movement: This habitat includes all potentially suitable active season upland movement habitat adjacent to modeled rice, open water, and fresh emergent wetland

land cover types with the potential to provide basking and short-term refuge. This habitat was modeled by selecting all natural vegetation types that occur within 200 feet of modeled rice and fresh emergent wetland land cover types (Hansen 1986; Wylie et al. 1997; USFWS 1999). Note that if habitat in this category remains outside the winter flood zone it may also be used for overwintering.

- Overwintering Habitat: This habitat includes all potentially suitable overwintering habitat outside of the active season upland movement habitat that may provide long-term refuge during the winter. This habitat was modeled by selecting all natural vegetation types that occur between 200 feet and 820 feet from modeled rice and fresh emergent wetland land cover types (Hansen 1986, Wylie et al. 1997, USFWS 1999).

Aquatic Habitat: This habitat type includes all aquatic features that might be used by the giant garter snake. This habitat was modeled by selecting all open water features that occur east of Highway 113 and east of Interstate 5 from its junction with Highway 113. Larger water features including Cache and Putah Creeks, the Sacramento River, and the Deep Water Channel were excluded along with water features surrounded by development without surrounding upland habitat. (Hansen 1986, Wylie et al. 1997, USFWS 1999).

Figure A-5. Giant Garter Snake Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

Path: C:\NGIS\SWHP\GIS\Figures\v1011\ReNumbered\SecondPass\AppendixA\Fig_A5.mxd; User: Marcus; Date: 2/9/2015

A.6 Swainson's Hawk (*Buteo swainsoni*)

© LorrieJo Williams

A.6.1 Listing Status

Federal: Bird of Conservation Concern (U.S. Fish and Wildlife Service [USFWS] 2008).

State: Threatened.

Recovery Plan: None.

A.6.2 Species Description and Life History

Swainson's hawk (*Buteo swainsoni*) is a long-winged, medium-sized soaring raptor, (48 to 56 centimeters [19 to 22 inches] and 693 to 1367 grams [24.46 to 48.26 ounces]) that nests and roosts in large trees in flat, open grassland or agricultural landscapes. Females average larger than males, but there are no distinguishing plumage characteristics for separating the sexes.

Swainson's hawk is characterized by its long, narrow, and tapered wings held in flight in a slight dihedral shape. The body size is somewhat smaller, thinner, and less robust than other *Buteos*, although the wings are at least as long as other *Buteos*. This body and wing shape allows for efficient soaring flight and aerial maneuverability, important for foraging, which Swainson's hawks do primarily from the wing, and during courtship and inter-specific territorial interactions.

There are three definitive plumage morphs: light, rufous, and dark. However, there are numerous intermediate variations between these plumage morphs. The two most distinguishing plumage characteristics are a dark breast band and the contrasting darker flight feathers and lighter wing lings on the underwings, giving most individuals a distinctive bicolored underwing pattern. These characteristics are most pronounced in lighter morph birds and become less so as the plumage darkens, and are indistinguishable in the definitive dark morph, which is completely melanistic. All three definitive plumage morphs are present in the Central Valley with a relatively large proportion of the population categorized as intermediate morph, with varying amounts of streaking or coloration in the belly and wing linings.

A.6.2.1 Seasonal Patterns

Swainson's hawks arrive on their breeding grounds in the Central Valley from early March to early April. The breeding season extends through mid-to-late August, when most young have fledged and breeding territories are no longer defended. By late August pre-migratory groups begin to form. The fall migration begins early- to mid-September. By early October, most Swainson's hawks have migrated out of the Central Valley. Central Valley Swainson's hawks winter from Central Mexico, to northern and central South America (Bradbury et al. in preparation). This differs from what is known about the migratory pattern and wintering grounds of Swainson's hawk populations outside of the Central Valley, most of which take a different migratory route and winter entirely in southern South America, with the largest wintering populations known to occur in northern Argentina (England et al. 1997).

A.6.2.2 Reproduction

Swainson's hawks exhibit a high degree of nest site fidelity, using the same nests, nest trees, or nesting stands for many years (England et al. 1997). Pairs are monogamous and may maintain bonds for many years (England et al. 1997). Immediately upon arrival onto breeding territories, breeding pairs begin constructing new nests or repairing old ones. One to four eggs are laid in mid- to late April followed by a 30- to 34-day incubation period. Nestlings begin to hatch by mid-May followed by an approximately 20-day brooding period. The young remain in the nest until they fledge in 38 to 42 days after hatching (England et al. 1997). Studies conducted in the Sacramento Valley indicate that one or two, and occasionally three, young typically fledge from successful nests (Estep in preparation). The rate of young fledged per nest in the Central Valley is among the lowest recorded in the entire species range. This geographic difference in reproductive success may be related to the reliance on small voles that may not meet the high energetic demands of breeding adults and developing young compared to the diets that include a higher proportion of gophers, rabbits, ground squirrels and other larger mammals consumed in other locations (S. England pers. comm.). In Yolo County, fledging rates ranged from 1.15 to 1.96 young per successful nest from 1988 to 2000 (Table 1) (Estep in preparation).

After fledging, young remain near the nest and are dependent on the adults for about four weeks, after which they permanently leave the breeding territory (Anderson et al. in preparation).

A.6.2.3 Home Range/Territory Size

Home ranges are highly variable depending on cover type, and fluctuate seasonally and annually with changes in vegetation structure (e.g., growth, harvest) (Estep 1989; Woodbridge 1991; Babcock 1995). Smaller home ranges consist of high percentages of alfalfa, fallow fields, and dry pastures (Estep 1989; Woodbridge 1991; Babcock 1995). Larger home ranges were associated with higher proportions of cover types with reduced prey accessibility, such as orchards and vineyards, or reduced prey abundance, such as flooded rice fields. Swainson's hawks regularly forage across a very large landscape compared with most raptor species. Data from Estep (1989) and England et al. (1995) indicate that it remains energetically feasible for Swainson's hawks to successfully reproduce when food resources are limited around the nest and large foraging ranges are required. Radio-telemetry studies indicate that breeding adults in the Central Valley routinely forage as far as 30 kilometers (km) (18.7 miles) from the nest (Estep 1989; Babcock 1995).

Home ranges (calculated as minimum convex polygons) for 12 Swainson's hawks in the Central Valley, including six in Yolo County, averaged 27.6 square kilometers (km^2)(10.7 square miles [mi^2]) (range: 3.36 to 87.18 km^2 [1.3 to 33.7 mi^2) (Estep 1989). Using similar methods, four Swainson's hawks in West Sacramento averaged 40.5 km^2 (15.6 mi^2) (range: 7.2 to 76.6 km^2 [2.8 to 29.6 mi^2]), and included fields planted in grain, alfalfa, tomatoes, and safflower, as well as fallow fields (Babcock 1995).

Swainson's hawks in the central region of the Central Valley (including Yolo County) had the shortest distances between nests of those reported in England et al. (1997); on average, nests were 1.14 km (0.7 miles) apart (Estep 1989). Nesting density in the Central Valley was calculated at 30.2 pairs/100 km^2 (11.7 pairs/100 mi^2) (range: 21.4 to 39.1 km^2; [8.3 to 15.1 mi^2]) (England et al. 1995). This high nest density was attributed to widely available, uniformly distributed optimal foraging habitat and relatively abundant nesting sites along narrow riparian corridors, farm shelterbelts, roadside trees, remnant groves, and isolated trees. Results from a 2007 baseline survey of nesting Swainson's hawks in Yolo County indicate a nesting density within the survey area (excluding the higher elevation portions of the county of 98 pairs/100 km^2 (37.8/100 mi^2), the highest nesting density reported for this species (Estep 2008).

A.6.2.4 Foraging Behavior and Diet

Swainson's hawks hunt primarily from the wing, searching for prey from a low-altitude soaring flight, 30 to 90 meters (98.4 to 295.2 feet) above the ground and attack prey by stooping toward the ground (Estep 1989). This species is also highly responsive to farming activities that expose and concentrate prey, such as cultivating, harvesting, and disking. During these activities, particularly late in the season, Swainson's hawks will hunt behind tractors searching for exposed prey. Other activities, such as flood irrigation and burning, also expose prey and attract foraging Swainson's hawks.

In the Central Valley, Swainson's hawks feed primarily on small rodents, usually in large fields that support low vegetative cover (to provide access to the ground) and high densities of prey (Bechard 1982; Estep 1989). These habitats include hay fields, grain crops, certain row crops, and lightly grazed pasturelands. Fields lacking adequate prey populations (e.g., flooded rice fields) or those that are inaccessible to foraging birds (e.g., vineyards and orchards) are rarely used Estep 1989; Babcock 1995; Swolgaard 2004).

Meadow vole (*Microtus californicus*) is the principal prey item taken by Swainson's hawks in the Central Valley (Estep 1989). Pocket gopher (*Thomomys bottae*) is also an important prey item. Other small rodents, including deer mouse (*Peromyscus californicus*) and house mouse (*Mus musculus*) are also taken along with a variety of small birds, reptiles, and insects.

During late summer, the diet of post-breeding adults and juveniles includes an increasing amount of insects, including grasshoppers and dragonflies. Dragonflies may constitute a major proportion of the diet of post-breeding and migrant birds. In the Central Valley during summer, dragonfly species that swarm in large numbers and that are a potentially important, abundant food source are common green darner (*Anax junius*), spot-winged glider (*Pantalahy hymenaea*), and wandering glider (*Pantala flavescens*). In alfalfa and corn crops in Idaho, post-breeding flocks also forage primarily on grasshoppers (Johnson et al. 1987). Dragonflies are also the primary prey for wintering birds in Argentina (Jaramillo 1993).

Following their arrival back on the breeding grounds, Swainson's hawks again shift their diet to include larger prey such as small rodents, rabbits, birds, and reptiles (England et al. 1997). This shift to a higher quality diet is prompted by the nestlings' nutritional demands during rapid growth and the adults' high energetic costs of breeding.

A.6.3 Habitat Requirements and Ecology

A.6.3.1 Nesting

Throughout much of its range, both in North and South America, the Swainson's hawk inhabits grasslands, prairies, shrub-steppes, and agricultural landscapes, including dry and irrigated row crops, alfalfa and hay fields, pastures, and rangelands. They nest in trees most often in riparian woodlands and farm shelterbelts (England et al. 1997), as well as in urban/suburban areas with large trees adjacent to suitable foraging habitat (England et al. 1995; James 1992). Suitable nest trees are usually deciduous and tall (up to 30.48 meters [100 feet]); but in suburban/urban areas, most nest trees are conifers (England et al. 1997; England et al. 1995). Nests are built of sticks sometimes several feet in diameter. They are generally placed in the uppermost and outermost branches that will support the nest, often in mistletoe clumps (England et al. 1997).

In the Central Valley, Swainson's hawks usually nest in large native trees such as valley oak (*Quercus lobata*), cottonwood (*Populus fremontii*), walnut (*Juglans hindsii*), and willow (*Salix* spp.), and occasionally in nonnative trees such as eucalyptus (*Eucalyptus* spp.). Nests occur in riparian woodlands, roadside trees, trees along field borders, isolated trees, small groves, and on the edges of remnant oak woodlands. Stringers of remnant riparian forest along drainages contain the majority of known nests in the Central Valley (Estep 1984; Schlorff and Bloom 1984; England et al. 1997). This appears to be a function of nest tree availability, however, rather than dependence on riparian forest. Nests are usually constructed as high as possible in the tree, providing protection to the nest as well as visibility from it.

Tables 1 and 2 indicate the nesting habitat results from the 2007 baseline survey (Estep 2008). Riparian habitat was the most frequently used nesting habitat type, followed by roadside tree rows, isolated trees, and rural residential trees. Valley oak (*Quercus lobata*) was the most frequently used nest tree species, followed by Fremont cottonwood (*Populus fremontii*), walnut (*Juglans hindsii*), willow (*Salix* spp.), and eucalyptus trees (*Eucalyptus* spp.).

Table A-4 Nesting Habitat Associations of Swainson's Hawk Territories in the Yolo County Study Area, 2007

Nesting Habitat Type	Number of Territories	Percent of Total
Riparian (natural)	106	36.6
Roadside Tree Row	39	13.4
Riparian (channelized)	36	12.4
Isolated Tree	32	11.0
Rural Residential	26	9.0
Tree Row	19	6.6
Isolated Roadside Tree	15	5.2
Eucalyptus Grove	6	2.1
Oak Grove	4	1.4
Urban	3	1.0
Cottonwood Grove	1	0.3
Savanna	1	0.3
Farmyard	1	0.3
Mixed Grove	1	0.3
Total	**290**	**100**

Table A-5. Nest Tree Species used by Nesting Swainson's Hawks in the Yolo County Study Area, 2007

Tree Species	Number of Active Nest Sites	Percent of Total
Valley Oak	101	35.7
Cottonwood	76	26.9
Walnut	33	11.7
Willow	32	11.3
Eucalyptus	26	9.2
Pine	7	2.5
Locust	4	1.4
Redwood	2	0.7
Sycamore	2	0.7
Total	**283**	**100**

A.6.3.2 Foraging

Swainson's hawks are essentially plains or open-country hunters, requiring large areas of open landscape for foraging. Historically, the species used the grasslands of the Central Valley and other inland valleys, and valley oak savanna with and understory of *Elymus triticoides*. With substantial conversion of these grasslands to farming operations, Swainson's hawks have shifted their nesting and foraging into those agricultural lands that provide low, open vegetation for hunting and high rodent prey populations.

Foraging habitat value is a function of patch size (i.e., Swainson's hawks are sensitive to fragmented landscapes; use will decline as suitable patch size decreases), prey accessibility (i.e., the ability of hawks to access prey depending on the vegetative structure), and prey availability (i.e., the abundance of prey populations in a field). In the Central Valley, agricultural land use or specific crop type determines the foraging value of a field at any given time. Cover types were evaluated by Estep (1989) and ranked based on these factors. However, suitability ranking is based on a variety of site-specific issues and at a landscape level should be characterized only on a general basis. On a site-specific level – important for land management purposes to maximize foraging value – individual cover types can be assessed based on site-specific and management conditions.

Important land cover or agricultural crops for foraging are alfalfa and other hay, grain and row crops, fallow fields, dryland pasture, and annual grasslands. The matrix of these cover types across a large area creates a dynamic foraging landscape as temporal changes in vegetation results in changing foraging patterns and foraging ranges.

Hay crops, particularly alfalfa, provide the highest value because of the low vegetation structure (high prey accessibility), relatively large prey populations (high prey availability), and because farming operations (e.g., weekly irrigation and monthly mowing during the growing season) enhance prey accessibility. Most row and grain crops are planted in winter or spring and have foraging value while the vegetation remains low, but become less suitable as vegetative cover and density increases. During harvest, vegetation cover is eliminated while prey populations are highest, significantly enhancing their suitability during this period. Some crop types, such as rice, orchards, and vineyards, provide little to no value because of reduced accessibility and relatively low prey populations.

A.6.4 Species Distribution and Population Trends

A.6.4.1 Distribution

In North America, Swainson's hawks nest in the grassland plains and agricultural regions from southern Canada (and possibly in the northern provinces and territories, and Alaska) to northern Mexico. Other than a few documented small wintering populations in the United States (Herzog 1996; England et al. 1997), the species winters primarily in the Pampas region of Argentina. The Central Valley population winters between Mexico and central South America (Bradbury et al. in preparation).

Early accounts described Swainson's hawk as one of the most common raptors in California, occurring throughout much of lowland the portions of the state (Sharp 1902). Since the mid-1800s, native habitats that supported the species have undergone a gradual conversion to agricultural or urban uses. Today, native grassland habitats are virtually nonexistent in the state, and only remnants of the once vast riparian forests and oak woodlands still exist (Katibah 1983). While the species has successfully adapted to certain agricultural landscapes, this habitat loss has caused a substantial reduction in the breeding range and in the size of the breeding population in California (Bloom 1980; England et al. 1997). Current breeding populations occur primarily in the Central Valley, but also in the Klamath Basin, the northeastern plateau, Owens Valley, and rarely in the Antelope Valley (Grinnell and Miller 1944; Bloom 1980; Garrett and Dunn 1981). The bulk of the Central Valley population resides in Yolo, Sacramento, Solano, and San Joaquin Counties.

In Yolo County, the species is distributed throughout the low elevation agricultural region east of the Interior Coast Range. Closely associated with agricultural cover type, the distribution of the species generally follows the pattern of hay, grain, and row crops. The majority of nesting pairs occur from several miles north of Woodland south to Putah Creek and east to the Sacramento River. Fewer pairs occur in the predominantly rice growing region in the northeastern portion of the county, in the orchard region in the northwest and southwest portions of the county, and the wetland-dominated areas of the southern panhandle. They generally avoid scrub, chaparral, savannah, or oak-dominated habitats in the western portion of the county. The highest nesting concentrations are north of Woodland to County Road 12; along oak and cottonwood-dominated riparian corridors such as Willow Slough, Putah Creek, and the Sacramento River; and between Davis and Woodland, and west to approximately Interstate 505 and east to the Sacramento River (Estep 2008).

A.6.4.2 Population Trends

Swainson's hawk populations have declined in California, Utah, Nevada, and Oregon (England et al. 1997). Populations in other western states are considered stable. Bloom (1980) reported a statewide estimate of 375 breeding pairs. This was followed by estimates of 550 (California Department of Fish and Game [DFG] 1988) in the late 1980s and 800 to 1,000 breeding pairs in the late 1990s (Swainson's Hawk Technical Advisory Committee 1999). However, none of these estimates was generated using a statistically based statewide survey effort and would be considered less credible than the results of a more statistically valid approach. The most recent statewide population estimate for California is 2,081 breeding pairs (Anderson et al. 2006) and is based on a statistically valid statewide survey effort conducted in 2005 and 2006. While this estimate is higher than the original statewide estimate that led to the state listing of the species (Bloom 1980) and subsequent estimates through the 1980s and 1990s, it represents a substantial decline (50–90 percent) of the historical statewide breeding population in California (Bloom 1980).

Baseline surveys conducted in 2007 located a total of 290 active breeding territories in Yolo County (Estep 2008). This was the first comprehensive baseline of this species in the County, and thus cannot be used to assess a trend in the number of breeding pairs in the County. However, based on the results of a long-term population study conducted in Yolo County since the mid-1980s (Estep in preparation), there appears to have been an upward trend in the number of breeding pairs (Table 3). While this may be at least partially attributed to increasing observer detection skill in the early years of the study, this local population appears to be at least stable with respect to the number of breeding pairs. Whether or not this population is stable based on productivity and recruitment is undetermined.

Table A-6. Swainson's Hawk Activity Data: Yolo County Study Area 1988–2000[a]

Year	Active Territories	Nesting Pairs	Successful Nests	Number of Young	Fledging Rate per Successful Nest
1988	55	48	46	62	1.34
1989	71	61	60	90	1.50
1990	85	72	70	118	1.69
1991	108	95	83	122	1.45
1992	122	110	94	136	1.45
1993	101	80	68	105	1.54
1994	137	128	110	188	1.70
1995	140	110	83	110	1.33
1996	139	101	75	107	1.43
1997	125	78	66	92	1.39
1998	158	103	27	31	1.15
1999	131	127	71	139	1.96
2000	136	126	69	102	1.48

[a] From Estep, J. A. In preparation. Ecology of the Swainson's Hawk in the Central Valley of California.

A.6.5 Threats to the Species

Swainson's hawks face different threats in different portions of their range. In California, causes of population decline are thought to be loss of nesting habitat (Schlorff and Bloom 1984) and loss of foraging habitat to urban development and to conversion to unsuitable agriculture such as orchards and vineyards (England et al. 1997; England et al. 1995). Nestlings are vulnerable to starvation and fratricide (i.e., the larger nestling killing the smaller nestling in times of food stress); predation from other raptors, crows, and ravens cause significant nestling losses. Natural population cycles of voles in central California may be a major factor in reproductive success where vole population crashes suppress reproduction or lead to increased starvation rates of nestlings (J. Estep pers. comm.). In addition, insecticides and rodenticides may contribute to these rates by reducing prey abundance. There is little evidence that adult Swainson's hawks are killed by natural predators, but collisions with moving vehicles and illegal shooting and trapping have been identified as sources of mortality (England et al. 1997).

Well-documented mass poisoning of hundreds or thousands of Swainson's hawks wintering in Argentina (Woodbridge et al. 1995; Goldstein et al. 1996) have led to that country's ban of an insecticide (organophosphate monocrotophos) used on alfalfa and sunflower fields to control grasshopper populations. Levels of dichlorodiphenyldichloroethylene (DDE), a breakdown product of DDT, in Swainson's hawks from the Central Valley may have been high enough to negatively affect reproductive success during the decades when it was used extensively in the United States. However, levels of DDE measured in eggs collected in 1982–1983 were not considered high enough to indicate a health threat (Risebrough et al. 1989).

Where populations are limited by inadequate nesting and foraging habitat, the most effective approach for Swainson's hawk conservation may be in management of agricultural landscapes (Smallwood 1995). Nesting density is greatest in cultivated areas where tree density (Schmutz 1984) and prey availability (Bechard 1982) are highest. Alfalfa fields are among the more valuable foraging habitats in California, even when compared with nonagricultural areas. However, valuable prey species such as pocket gophers (*Thomomys* spp.) and other small mammals may be exterminated in such fields (Smallwood 1995). While agricultural areas may benefit these hawks, fully realizing the conservation potential of cultivated areas to Swainson's hawks will be impaired when prey populations are controlled by means of poisons. Maintenance of critical prey populations is necessary to attain the full benefits of alfalfa fields and other agricultural crops to Swainson's hawks (Smallwood 1995).

In contrast to some agricultural landscapes, Swainson's hawks are absent from or are in very low densities in large expanses of annual grasslands in the Central Valley (Detrich 1996 cited in Woodbridge 1998). These grasslands have high densities of nocturnal, burrowing rodents that are rarely available as prey to Swainson's hawks and have low densities of voles (*Microtus* spp.) and pocket gophers that the hawks prefer (Woodbridge 1998). Because voles are active during the day and live among vegetation, they are especially accessible and important prey for hawks. Restoring perennial grasslands and promoting agriculture that supports high densities of voles and pocket gophers would create or enhance foraging habitat and could potentially expand Swainson's hawk distribution in Yolo County.

Many populations of prey species, especially voles, mice and insects, fluctuate due to annual, seasonal, and local geographic variations in rainfall, predation pressures, natural population cycles, and agricultural practices, including changing crop types, harvesting, applying rodenticides and insecticides, flood irrigating, and disking. The timing of harvesting and disking also strongly affects prey abundance (Woodbridge 1998). The importance of crop types for foraging habitat rest on two variables: abundance of voles and other important prey, and amount of vegetative cover that affects access to prey (Estep 2009). Alfalfa is an important habitat because although it supports lower populations of voles, the amount of vegetative cover is not sufficient to provide much protection to voles from foraging hawks. Tomato and beets fields, in contrast, support high populations of voles, but their higher vegetative cover provides better protection for voles, thereby decreasing those habitats' value. Furthermore, as crops mature, their protective cover for rodents increases, making prey less available to hawks (Bechard 1982; Woodbridge 1998; Estep 2009). In agricultural landscapes, prey abundance and accessibility to hawks continuously change through the breeding season. All of these factors play major roles in reproductive success (J. Estep pers. comm.). To reduce negative effects on regional populations, large areas of optimal foraging habitats should be preserved or managed for populations of Swainson's hawks and their prey (DFG 1994). Better understanding of the dynamics and processes of how agricultural practices affect these populations on a landscape level would help to guide conservation planning.

In areas with suitable foraging habitat that lack Swainson's hawks, surveys of potential nest trees should be conducted to assess whether the hawk population is limited by lack of suitable nest trees. Also, the relationship between Swainson's hawks and locally breeding red-shouldered hawks, red-tailed hawks, and great horned owls should be studied to determine whether competition for nest trees and prey are negatively affecting the Swainson's hawk population or distribution in Yolo County.

A.6.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.6.3, *Habitat Requirements and Ecology* (Figure A-6). The model parameters include the following.

Nesting Habitat: This modeled habitat type includes all potentially suitable nesting habitat and was modeled by selecting all mapped vegetation types as listed below that occur below an elevation of 350 feet outside of Planning Units 3 and 4 (Hofmann pers. comm.). In addition, all remnant woody vegetation outside of blue oak woodland and blue oak foothill pine occurring in isolated patches or isolated trees in agricultural fields or field borders (Tuil 2008) outside of Planning Units 3 and 4 below an elevation of 350 feet were included as potential nesting habitat to the extent that they were mapped. The majority of isolated trees and roadside and field border trees, which are commonly used as Swainson's hawk nest trees, were not mapped and thus the extent and distribution of potential nesting habitat is underestimated. The elevation limit was based on the elevational extent of potential nesting habitat in the Plan Area.

- Eucalyptus
- Valley Oak Woodland
- Fremont Cottonwood – Valley Oak – Willow (Ash – Sycamore) Riparian Forest Not Formally Defined (NFD) Association
- Great Valley Valley Oak Riparian Association
- Mixed Fremont Cottonwood – Willow spp. NFD Alliance
- Mixed Willow Super Alliance
- Valley Oak – Fremont Cottonwood – (Coast Live Oak) Riparian Forest NFD Association
- Valley Oak Alliance –Riparian
- White Alder (Mixed Willow) Riparian Forest NFD Association
- Undifferentiated Riparian Woodland/Forest
- Agricultural Foraging Habitat: This modeled habitat type includes all of the annually cultivated irrigated cropland and semi-perennial hay crops (e.g., alfalfa) listed below that occur at an elevation of 500 feet or lower. While there is a high degree of variability in the suitability of these agricultural crop types, because they rotate annually or periodically, field-level value changes across the landscape each year.
    - All Field Crops
    - All Grain/Hay Crops
    - Pasture (alfalfa)

  - Native Pasture
  - Miscellaneous Grasses
  - Mixed Pasture
  - All Truck and Berry Crops
- Natural Foraging Habitat: This modeled habitat type includes the uncultivated grassland and seasonal wetland land cover types listed below that occur at an elevation of 500 feet or lower. These land cover types generally produce less available microtine prey due to dryer conditions or periodic inundation. While suitable foraging habitat, these types are expected to be used less frequently than cultivated habitats.
  - California Annual Grassland Alliance
  - Upland Annual Grassland and Forbs Formation
  - Alkali Sink
  - Vernal Pool Complex
  - *Carex* spp. *Juncus* spp. Wet Meadow Grasses NFD Super Alliance
  - *Crypsis* spp. Wetland Grasses – Wetland Forbs NFD Super Alliance
  - Undetermined Alliance – Managed
- Modeling limited to Planning Units: 3, 4, 5, 6, 7, 9, 10, 11, 12, 13, 14, 14, 16, 17, 18, 19, 20, 21, 22

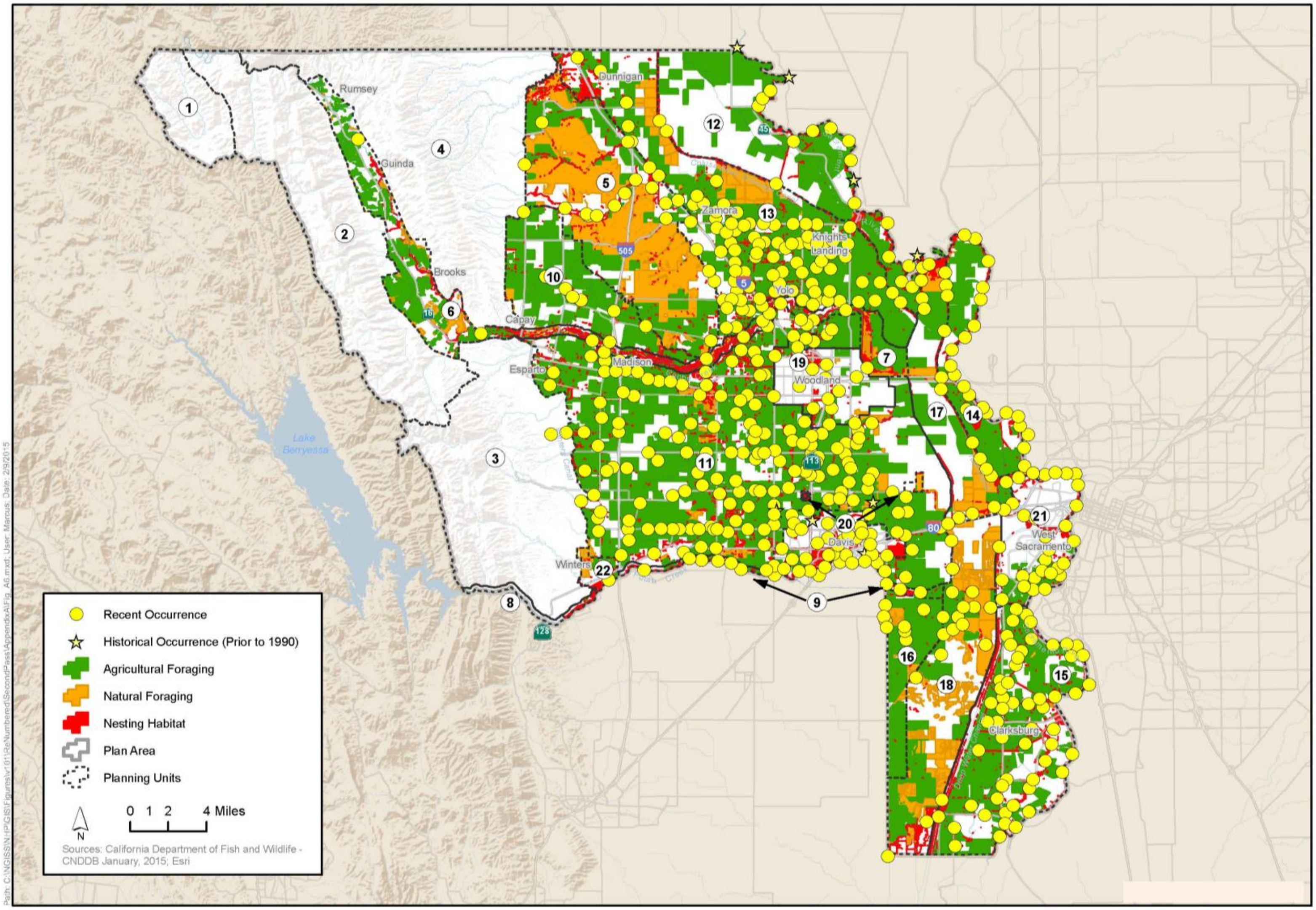
A.6.6.1 Cumulative Nest Locations and Sightings

Figure A-20 displays the cumulative distribution of recent and historical nest locations and sightings (nesting records with lower mapping precision) from a variety of data sources.

- Nest Locations (2007 surveys): Nest locations mapped from 2007 surveys (Estep 2008).
- Other Recent Nest Locations: Location where the nests have relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., California Natural Diversity Database [CNDDB], California Department of Fish and Wildlife [DFW], and Chris DiDio of the University of California, Davis (UC Davis).

Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., CNDDB, California Department of Fish and Game, Chris DiDio of UC Davis, UC Davis Museum of Wildlife and Fish Biology, California eBird, Avian Knowledge Network).

Figure A-6. Swainson's Hawk Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

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A.7 White-Tailed Kite (*Elanus leucurus*)

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A.7.1 Listing Status

Federal: None.

State: Fully Protected.

Recovery Plan: None.

A.7.2 Species Description and Life History

The white-tailed kite (*Elanus leucurus*) is a medium-sized (32- to 38-centimeter) raptor of open grasslands, savannahs, and agricultural areas. It is identified by its unique plumage and habit of hovering while hunting. It has long, narrow, and pointed wings and a long, bright-white tail, face, and underside that contrast with distinctive black patches on the inner wings. Adults also have gray backs and red eyes. The sexes are similar, but the female has a slightly darker back (Dunk 1995).

A.7.2.1 Seasonal Patterns

Although apparently a resident bird throughout most of its breeding range, dispersal occurs during the nonbreeding season, resulting in some range expansion during the winter. Stendell (1972) believed it to be resident, becoming nomadic during periods of low prey abundance. While population changes and local and regional movements appear to be somewhat predictable based on vole and other rodent cycles, it remains unknown whether in Northern California this constitutes a migration movement or nomadic response to changes in the prey populations (Dunk and Cooper 1994).

A.7.2.2 Reproduction

The breeding season from pair bonding to juvenile independence occurs from approximately January to October with peak activity occurring from May through August (Dunk 1995). Nests are constructed of loosely piled sticks and twigs that are lined with grass, straw, or rootlets. The nest is placed near the top of a dense oak, willow, or other tree; usually 6 to 20 meters above ground in trees that vary from 3 to 50 meters in height (Dixon et al. 1957). Females typically lay a clutch of four eggs, with a range of three to six. The female incubates exclusively and performs most brooding while the male provisions the female and nestlings. Eggs are incubated for about 28 days. Young fledge in 35–40 days following hatching, with the peak fledging period occurring in May–June (Erichsen 1995).

A.7.2.3 Home Range/Territory Size

Territory size is variable and regulated primarily by prey abundance and vegetation structure (i.e., accessibility of prey); however, this species also responds to the abundance of interspecific and intraspecific competitors (Dunk 1995; Erichsen 1995). Reported average territory sizes include 1.6–21.5 hectares (ha) (Dunk and Cooper 1994), 19–52 ha with a mean of 29 ha (Waian 1973), and 17–120 ha (Henry 1983). As with other raptors species, particularly those occurring in agricultural habitats, home ranges may overlap and foraging may be limited to a small portion of the total area. This may be a result of competition or fluctuating prey accessibility due to changes in vegetation structure (Henry 1983). Communal roosts are used during the nonbreeding season (Waian and Stendell 1970). Home ranges for nonbreeders is more difficult to determine since communal roosts may be tens of kilometers away (Dunk 1995).

A.7.2.4 Foraging Behavior and Diet

White-tailed kites generally hunt from a central perch over areas as large as 3 square kilometers (km^2) (Warner and Rudd 1975), but foraging usually occurs within 0.8 km from the nest during the breeding season (Hawbecker 1942). Kites are not particularly territorial. The nest site and the immediate surrounding area are defended against other raptors and crows (Pickwell 1930, Dixon et al. 1957). Small wintering territories of about 0.10 km^2 have been documented to be defended as well (Bammann 1975).

The white-tailed kite preys mostly on voles, but also takes other small, diurnal mammals, and occasionally birds, insects, reptiles, and amphibians. Small mammal prey comprises 95 percent of the kite diet (Dunk 1995). It forages in undisturbed, open grasslands, meadows, farmlands and emergent wetlands, ungrazed grasslands, fence rows and irrigation ditches adjacent to grazed lands (Dunk 1995). It soars, glides, and hovers less than 30 meters above the ground in search of prey. It hunts almost exclusively by hovering from 5 to 25 meters in height, with hovering bouts lasting up to 60 seconds. During this time, kites scan the ground searching for prey and watching for potential competitors or predators. The hovering bout ends in a dive to the ground for prey; flight to another location; soaring or interacting with another bird; or flight to the perch (Warner and Rudd 1975).

A.7.2.5 Predation

The primary cause of egg mortality is inclement weather and predation (Stendell 1972). Circumstantial evidence suggests red-tailed hawks may take adults (Pinkston and Caraviotis 1980). Skeletons of immature white-tailed kites with feathers on wings have been found beneath perches used by larger raptors, also suggesting predation (Dunk 1995).

A.7.3 Habitat Requirements and Ecology

A.7.3.1 Nesting

The white-tailed kite inhabits low elevation, open grasslands, savannah-like habitats, agricultural areas, wetlands, and oak woodlands (Dunk 1995). Habitat elements that influence nest site selection and nesting distribution include habitat structure (usually trees with a dense canopy) and prey abundance and availability (primarily the association with meadow vole), while the association with specific vegetation types (e.g., riparian, oak woodland, etc.) appears less important (Erichsen 1995; Dunk 1995). White-tailed kite nests have been documented in a variety of tree species, including valley oak (*Quercus lobata*), Fremont cottonwood (*Populus fremontii*), willow (*Salix* spp.), live oak (*Quercus wislizenii*), box elder (*Acer negundo*), ornamental trees including olive and pine trees, and occasionally in tall shrubs (Dixon et al. 1957; Dunk 1995).

Nest trees appear to be selected on the basis of structure and security, and thus typically have a dense canopy or are within a dense group of trees, such as riparian forest or oak woodland. Kites will occasionally use isolated trees, but this is relatively rare. Most nests in the Sacramento Valley are found in oak/cottonwood riparian forests, valley oak woodlands, or other groups of trees and are usually associated with compatible agricultural foraging habitat, such as pasture and hay crops, compatible row and grain crops, or natural vegetation such as seasonal wetlands and annual grasslands (Erichsen 1995).

Kites often nest in close association with other nesting kites and with several other raptors. These include the Swainson's hawk (*Buteo swainsoni*), red-tailed hawk (*Buteo jamaicensis*), and red-shouldered hawk (*Buteo lineatus*) (particularly in riparian habitats of the Sacramento Valley).

A.7.3.2 Foraging

The white-tailed kite uses a variety of foraging habitat types, but those that support larger and more accessible prey populations are more suitable. The presence and abundance of white-tailed kites are strongly correlated with the presence of meadow voles (Stendell 1972). As a result, population cycles of meadow voles can also influence nesting and wintering abundance of white-tailed kites. Cover types that appear to be preferred include alfalfa and other hay crops, irrigated pastures, and some cultivated habitats, particularly sugar beets and tomatoes, both of which can support relatively large populations of voles (Estep 1989) and which have been highly correlated with kite nest site densities (Erichsen et al. 1994). Kites also forage in dry pastures, annual grasslands, rice stubble fields, and occasionally in orchards (Erichsen 1995).

Winter foraging habitat is similar to breeding season foraging habitat (particularly the association with agricultural habitats and vole populations); however, there is less association with riparian forests and woodlands.

A.7.4 Species Distribution and Population Trends

A.7.4.1 Distribution

The white-tailed kite was threatened with extinction in North America during the early twentieth century (Eisenmann 1971). Until the 1960s, the species was considered declining throughout its North American range, but since then has recovered in some areas. Currently, the distribution of the species includes the East Coast and southeast United States, the southwest United States from Texas to California, and north to Washington State, and from Mexico to South America (Dunk 1995). Relatively stable resident populations occur in California, portions of coastal Oregon and Washington, southern Florida, southern Texas, and portions of northern Mexico. The species is considered rare in remaining portions of its North American range. Range expansion has also been noted in some Central American locales (Eisenmann 1971).

White-tailed kite has been reported from most of the open, lowland habitats in Yolo County. The species is underreported in the California Natural Diversity Database (CNDDB 2009) with only six nest sites reported, all in the vicinity of Davis. A total of 13 nest sites was reported during a survey of the lowland portion of Yolo County conducted in 2007 (Estep 2008). Most were found in riparian areas, including three along Putah Creek, three along Willow Slough, two along Dry Slough, one along the Sacramento River, one along the Willow Slough Bypass, and one along the Knights Landing Ridge Cut. Two nonriparian sites included one in West Sacramento and one near Dunnigan. Whisler (pers. comm., 2015) reported several suburban nests in east and north Davis and the Willowbank area (planning unit 20), El Macero Golf Course, and UC Davis during 2001 and 2002. No trend information for Yolo County is available.

A.7.4.2 Population Trends

California populations were also thought to be seriously declining prior to the 1960s, likely due to habitat loss, shooting, and possible egg collecting (Pickwell 1930; Waian and Stendell 1970). From the 1940s to the 1970s, populations and distribution increased (Fry 1966, Waian and Stendell 1970, Eisenmann 1971) due to protection from shooting and possibly due to increasing agricultural development, which may have increased rodent habitat and expanded the foraging range of white-tailed kite (Eisenmann 1971; Small 1994). In the Sacramento Valley, the kite has increased predominantly in irrigated agricultural areas where meadow vole (*Microtus californicus*) populations are found (Warner and Rudd 1975).

California is currently considered the breeding range stronghold for white-tailed kite in North America, with nearly all areas up to the western Sierra Nevada foothills and southeast deserts occupied (Small 1994; Dunk 1995). It is common to uncommon and a year-round resident in the Central Valley, other lowland valleys, and along the entire length of the coast (Dunk 1995).

Although white-tailed kite is probably resident through most of its breeding range, dispersal occurs during the non-breeding season, leading to a winter range expansion that includes most of California (Small 1994; Dunk 1995).

While white-tailed kite populations may have recovered to some extent since the 1960s as a result of agricultural crop conversions in Yolo County, the species is also subject to interspecific competition with nesting great-horned owls, Swainson's hawks, red-tailed hawks, and red-shouldered hawks, which can result in territory abandonment or nest failure. Erichsen (1995) reported six of 13 kite nest failures in riparian areas due to displacement by nesting Swainson's hawks.

A.7.5 Threats to the Species

A.7.5.1 Urbanization/Fragmentation

Urbanization, including residential and commercial development and infrastructure development (roads and oil, water, gas, and electrical conveyance facilities) is one of the principal causes of continuing habitat loss for white-tailed kite and is a continuing threat to remaining populations, particularly in rapidly urbanizing areas in the Sacramento Valley. Urbanization permanently removes habitat and results in permanent abandonment of nesting territories. Proximity to urban areas also influences kite occurrence. While there are examples of kites nesting and roosting in urban areas, in general, the species is intolerant of noise and human activities and will abandon nesting areas that are subject to increasing levels of human disturbances. Kites are also sensitive to habitat fragmentation. Low density urbanization or isolation of habitats, even if relatively large patches remain undisturbed, also leads to territory abandonment.

A.7.5.2 Agricultural Crop Conversion

As noted above, white-tailed kite populations are closely associated with rodent abundance and accessibility, which can be influenced by crop patterns. Kite populations have recovered to some extent in California due in part to the expansion of compatible agricultural types. The conversion to crop patterns that do not support sufficient rodent prey or that restrict accessibility to prey can result in the abandonment of traditionally active territories.

A.7.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.7.3, *Habitat Requirements and Ecology* (Figure A-7). The model parameters include the following. CNDDB Location: These are locations where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (CNDDB).

- Nesting Habitat: This habitat type includes all potentially suitable nesting habitat, which was modeled by selecting all mapped vegetation types as listed below that occur below an elevation of 500 feet. In addition, all remnant woody vegetation occurring in isolated patches or isolated trees in agricultural fields or field borders (Yolo County Remnant Woody Vegetation mapping project)[1] were included as potential nesting habitat.
- Primary Foraging Habitat: This habitat includes all potentially suitable foraging habitat on the valley floor that is of higher value because these vegetation types are nearer to nesting habitat and have the physical structure and planting/harvesting patterns to make higher density prey available to white-tailed kites. This habitat was modeled by selecting all mapped pasture types (including alfalfa) and annual grasslands, that occur at an elevation of 500 feet or lower and are within 1 mile of modeled nesting habitat and reported nesting location in all ecoregions.
- Secondary Foraging Habitat: This habitat includes all potentially suitable foraging habitat that is also nearer to nesting habitat but has crop and vegetation communities that are used less frequently than those in the Primary Foraging category. This habitat was modeled by selecting all mapped vegetation types as listed below that occur at an elevation of 500 feet or lower and are within 1 mile of modeled nesting habitat and reported nesting location in all ecoregions.

A.7.6.1 Nesting Habitat – Vegetation Types

- Blue Oak Woodland
- Blue Oak – Foothill Pine
- Eucalyptus
- Valley Oak Woodland
- Fremont Cottonwood – Valley Oak – Willow (Ash – Sycamore) Riparian Forest NFD Association
- Great Valley – Valley Oak Riparian Association
- Mixed Fremont Cottonwood – Willow spp. NFD Alliance
- Mixed Willow Super Alliance

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[1] GIS layer prepared by J. Tuil in 2008 for Yolo County NHP.

- Valley Oak – Fremont Cottonwood – (Coast Live Oak) Riparian Forest NFD Association
- Valley Oak Alliance – Riparian
- White Alder (Mixed Willow) Riparian Forest NFD Association
- Undifferentiated Riparian Woodland/Forest

A.7.6.2 Primary Foraging Habitat – Vegetation Types

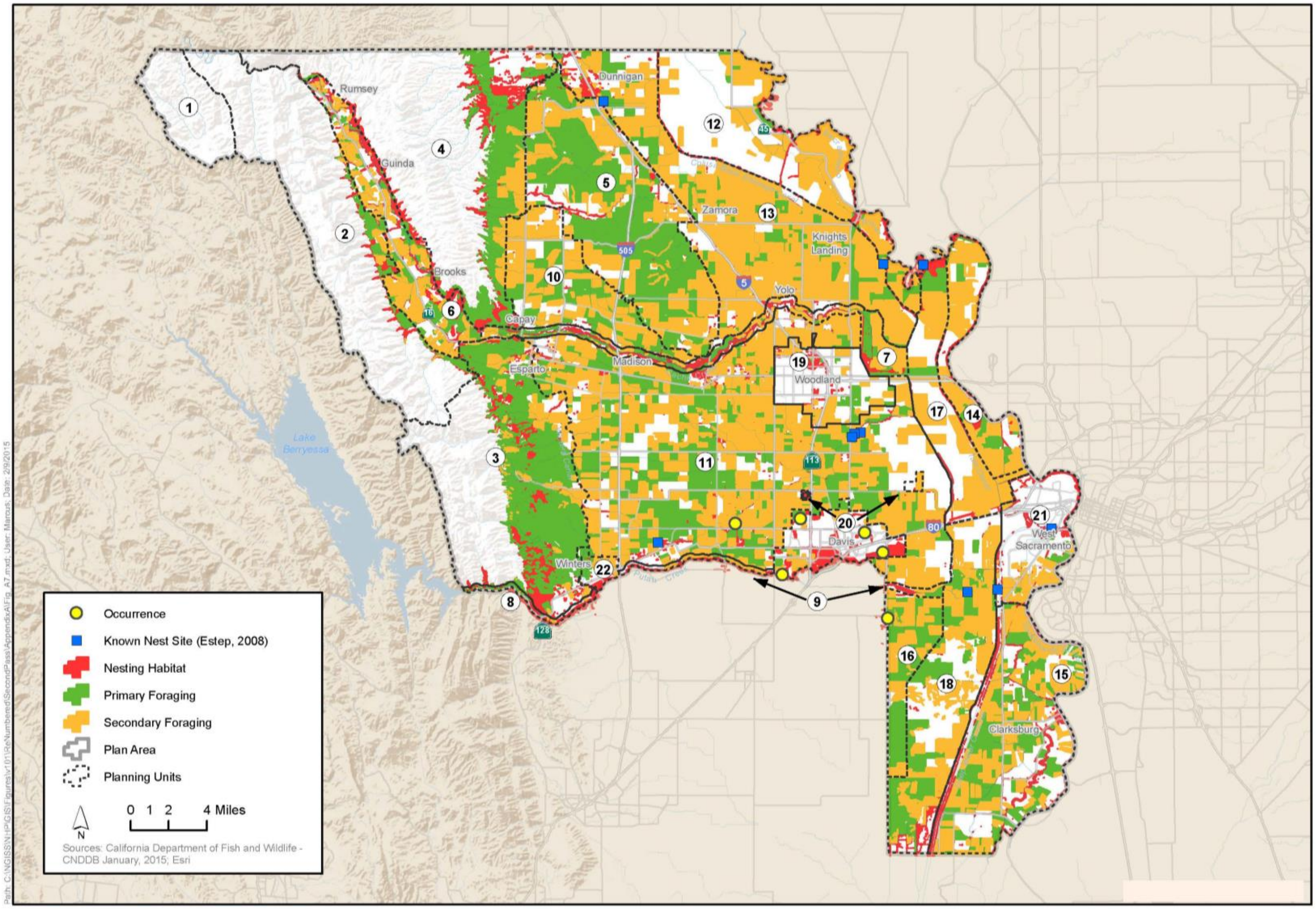
- All pasture types (including alfalfa)
- Annual grassland

A.7.6.3 Secondary Foraging Habitat – Vegetation Types

- *Crypsis*
- Carex
- Undetermined Alliance – Managed
- Alkali Sink
- Vernal Pool Complex
- Grain/Hay Crops
- Field Crops

Truck/Berry Crops

Figure A-7. White-Tailed Kite Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

Path: C:\NGISSNHP\GIS\Figures\v101\ReNumbered\SecondPass\AppendixA\Fig_A7.mxd; User: Marcus; Date: 2/9/2015

A.8 Western Yellow-Billed Cuckoo (*Coccyzus americanus*)

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A.8.1 Listing Status

Federal: Threatened.

State: Threatened.

Recovery Plan: None.

A.8.2 Species Description and Life History

A.8.2.1 Description

The western yellow-billed cuckoo (*Coccyzus americanus*) is a medium-sized bird about 30 centimeters (11.8 inches) in length with a wingspan of 38–43 centimeters (15–17 inches). The species has a slender, long-tailed profile, with a fairly stout and slightly down-curved bill, which is blue-black with yellow on the base of the lower mandible. Plumage is grayish-brown above and white below, with red primary flight feathers. The tail feathers are boldly patterned with characteristic rows of large white spots on the underside. The legs are short and bluish-gray. Adults have a narrow, yellow eye ring. Juveniles resemble adults, except the tail patterning is less distinct, and the lower bill may have little or no yellow (Hughes 1999).

A.8.2.2 Seasonal Patterns

In California on the Sacramento River, birds arrive onto breeding territories; pair formation occurs from late June to mid-July following the northward migration from South America and is followed by nest building and raising of young (Halterman 1991). The species is restricted to the mid-summer period for breeding presumably due to a seasonal peak in large insect abundance (Rosenberg et al. 1982). To accommodate this, development of young is very rapid with a breeding cycle of 17 days from egg-laying to fledging. Following a relatively short period of post-fledging juvenile dependency, cuckoos migrate out of California from approximately mid-August to early September. The species migrates to South America during the nonbreeding season and is thus not present in the Central Valley between October and May.

A.8.2.3 Reproduction

The pair constructs a flimsy twig nest which is typically 5 to 40 feet above the ground in dense canopy cover. Nests in the riparian forest along the South Fork of the Kern River consisted of twigs and were lined with roots and dried leaves and were rimmed with pine needles. Clutch size is usually three to four eggs, rarely five (Bent 1940). Both the female and the male incubate the eggs, which lasts for 10 to 11 days (Hamilton and Hamilton 1965). Both parents also share incubating and brooding duties and provision young with food. Young develop very rapidly and fledge from six to eight days post-hatching. Parental care continues for an additional three to four weeks before the southern migration begins (Halterman 1991).

In the well-studied Kern River population, it was found that 70 percent of western yellow-billed cuckoo pairs were monogamous, while the remaining 30 percent included a helper at the nest (Laymon 1998). When prey is abundant, cuckoos increase clutch size and may lay eggs in nests of other western yellow-billed cuckoo pairs and other nests of other species (Fleischer et al. 1985; Laymon 1998; Hughes 1999). Further, the Kern River studies determined that cuckoos tend to lay more eggs when they are able to feed nestlings a high percentage diet of katydids, and they tend to fledge more young when prey are easily and quickly captured (Laymon 1998).

A.8.2.4 Home Range/Territory Size

Limited information is available on home range and territory size. Territory size at the South Fork Kern River ranged from 8 to 40 hectares (ha) (20 to 100 acres) (Laymon and Halterman 1985), and on the Colorado River as small as 4 ha (10 acres) (Laymon and Halterman 1989). Patch size, type and quality of habitat, and prey abundance largely determine the size of territories (Halterman 1991).

Western yellow-billed cuckoos are loosely territorial and do not defend territories, but given uniform habitat they are regularly spaced through the landscape (Laymon 1998). Laymon (1980) found nests placed as close as 60 meters (197 feet) apart along the Sacramento River in an area where foraging habitat was abundant but nesting habitat was extremely limited. Breeding densities at the South Fork Kern River from 1985 to 1996 averaged 0.85 pairs/40 ha and ranged from a low of 0.15 pairs/40 ha in 1990 to a high of 1.4 pairs/40 ha in 1993 (Laymon unpublished data in Laymon 1998).

A.8.2.5 Foraging Behavior and Diet

Western yellow-billed cuckoos are primarily foliage gleaners (Laymon 1998). The typical strategy is to slowly hop from limb to limb in the canopy searching for movement of prey. They also sally from perches to catch flying insects or drop to the ground to catch grasshoppers or tree frogs (Laymon 1998).

Food resources vary greatly from year to year and significantly affect reproductive success (Laymon et al. 1997). Cuckoos forage within the riparian canopy primarily on slow-moving insects. The principal food item is green caterpillar (primarily sphinx moth larvae) (44.9 percent), with lesser amounts of katydids (21.8 percent), tree frogs (23.8 percent), and grasshoppers (8.7 percent). The diet also includes cicadas, dragonflies, butterflies, moths, beetles, and spiders (Laymon et al. 1997). Primary food items, particularly sphinx moth larvae, are associated with cottonwood trees and likely explain high reported use of cottonwood trees as foraging habitat for cuckoos (Laymon and Halterman 1985).

A.8.3 Habitat Requirements and Ecology

The western yellow-billed cuckoo is a riparian obligate species. Its primary habitat association is willow-cottonwood riparian forest, but other species such as alder (*Alnus glutinosa*) and box elder (*Acer negundo*) may be an important habitat element in some areas, including occupied sites along the Sacramento River (Laymon 1998). Nests are primarily in willow trees; however, other species are occasionally used, including cottonwood and alder. Along the Sacramento River, English walnut trees and more rarely prune, plum, and almond trees in adjacent orchards have also been reportedly used for nesting (Laymon 1980). Several nests on the Sacramento River were draped with wild grape (Gaines and Laymon 1984; Laymon 1998). Nest site height in willow trees average 4.3 meters (14.1 feet), but those in cottonwood trees have been reported at 30 meters (98.4 feet). Canopy cover is typically dense (averaging 96.8 percent at the nest) and large patch sizes (generally greater than 20 ha [49.4 acres] are typically required (Laymon 1998).

While western yellow-billed cuckoos nest primarily in willow (*Salix* spp.) trees, cottonwood (*Populus fremontii*) trees are important as foraging habitat, particularly as a source of insect prey. All studies indicate a highly significant association with relatively expansive stands of mature cottonwood-willow forests, especially dynamic riverine habitats where the river is allowed to meander and willows and cottonwoods can regenerate on point bars and stream banks (Grecco 2008). However, western yellow-billed cuckoos will occasionally occupy a variety of marginal habitats, particularly at the edges of their range (Laymon 1998). Continuing habitat succession has also been identified as important in sustaining breeding populations (Laymon 1998). Meandering streams that allow for constant erosion and deposition create habitat for new rapidly-growing young stands of willow, which create preferred nesting habitat conditions. Channelized streams or levied systems that do not allow for these natural processes become over-mature and presumably less optimal (Grecco 2008).

Along the Sacramento and Feather Rivers, primary factors influencing nest site selection include the presence of cottonwood/willow riparian forest, patch size, and density of understory vegetation. Laymon and Halterman (1989) found a significant trend toward increased occupancy with increased patch size. In California, except for the population along the Colorado River, cuckoos occupied 9.5 percent of 21 sites 20 to 40 ha in extent, 58.8 percent of 17 sites 41 to 80 ha in extent, and 100 percent of 7 sites greater than 80 ha in extent (Laymon and Halterman 1989). On the Sacramento River, Halterman (1991) found that the extent of patch size was the most important variable in determining occupancy.

A.8.4 Species Distribution and Population Trends

A.8.4.1 Distribution

There are two currently recognized subspecies, *C.a. occidentalis*, found west of the Rocky Mountains and *C.a. americanus*, found in deciduous forests east of the Rocky Mountains. There is a continuing debate over the taxonomic separation of the two subspecies, which is based primarily on morphological and plumage differences (Banks 1988; Franzreb and Laymon 1993), and more recently on genetics studies initiated by the U.S. Fish and Wildlife Service during the status review for federal listing.

The range of western yellow-billed cuckoo historically extended from southern British Columbia to the Rio Grande in northern Mexico, and east to the Rocky Mountains (Bent 1940). Currently the only known populations of breeding western yellow-billed cuckoo are several disjunct locations in California, Arizona, and western New Mexico (Halterman 1991). Western yellow-billed cuckoos winter in South America from Venezuela to Argentina after a southern migration that extends from August to October (Laymon and Halterman 1985). They migrate north in late June and early July (DeSchauensee 1970).

In California, where much of its historical range has been greatly reduced, western yellow-billed cuckoos still occur in isolated sites in the Sacramento Valley from Tehama to Sutter Counties, along the South Fork of the Kern River, and in the Owens Valley, Prado Basin, and Lower Colorado River Valley (Gaines and Laymon 1984; Laymon 1998).

A.8.4.2 Population Trends

Studies conducted since the 1970s indicate that there may be fewer than 50 breeding pairs in California (Gaines 1977; Laymon and Halterman 1987; Halterman 1991; Laymon et al. 1997). While a few occurrences have been detected elsewhere recently, including the Eel River, the only locations in California that currently sustain breeding populations include the Colorado River system in Southern California, the South Fork Kern River east of Bakersfield, and isolated sites along the Sacramento River in Northern California (Laymon and Halterman 1989; Laymon 1998).

Declines in numbers of the western yellow-billed cuckoo in California are a result of "removal widely of essential habitat conditions," as described by Grinnell and Miller (1944). These declines have continued primarily in the San Joaquin Valley, north coast, and central coast (where the populations had been extirpated by 1977) (Gaines and Laymon 1984), and the species was nearly extirpated in the Lower Colorado River Valley by 1999. In the Sacramento Valley, only 1 percent of the species' historical habitat remains to support a small population estimated at only 50 pairs in 1987 and 19 pairs in 1989 (Laymon and Halterman 1989). Population estimates based on surveys conducted in 1999 are similar to those from the 1980s (66 FR 38611). Because no surveys have been conducted since 1999, the current status of the Sacramento Valley population is not known.

A.8.4.3 Distribution and Population Trends in the Plan Area

The historical distribution of western yellow-billed cuckoo extended throughout the Central Valley, where the species was considered common (Belding 1890). In the mid-1940s, Grinnell and Miller (1944) still considered the Central Valley distribution to extend from Bakersfield to Redding. While there are few historical records from Yolo County, presumably the species nested within the county along the west side of the Sacramento River and possibly along smaller tributary drainages, including Putah Creek, Willow Slough, and Cache Creek.

Since 1965, there have been nine records of western yellow-billed cuckoo in Yolo County, including the following:

- Willow Slough in 1965
- Sacramento River in 1977
- Elkhorn Regional Park in 1982
- Gray's Bend in 1997
- City of Davis in 2001
- Putah Creek Sinks in June 2005
- Cache Creek Settling Basin in July 2005
- Fremont Weir in June 2006
- Fremont Weir in July 2006

These records were reported in Gaines (1974), Yolo Audubon Society Checklist Committee (2004), Yolo Audubon Society (2005), and by Steve Hampton.[2] All of these records are presumed to be migrants or nonbreeding individuals. While there are no confirmed breeding records for Yolo County, they are fairly common nesters just across the Sacramento River in Sutter County, especially in riparian forests along the western toe drain of the Sutter Bypass. Up to 15 birds responded to taped vocalizations while canoeing this area in a single day in mid-June 1995 (Beedy pers. obs.).

Very little potential breeding habitat remains in Yolo County, and the mostly channelized and riprapped banks of the Sacramento River provide few opportunities for river meandering and/or riparian restoration that would provide suitable western yellow-billed cuckoo breeding habitat (Grecco 2008). While migrants could potentially use riparian habitats along the Sacramento River and other watercourses, there are few areas that support sufficient contiguous patches of suitable habitat to support breeding cuckoos.

A.8.5 Threats to the Species

Historical declines have been due primarily to the removal of riparian forests in California for agricultural expansion and urban expansion (66 FR 38611). Habitat loss and degradation continues to be the most significant threat to remaining populations. Habitat loss continues as a result of bank stabilization and flood control projects, urbanization along edges of watercourses, agricultural activities, and river management that alter flow and sediment regimes. Fragmentation reduces the ability of an area to sustain a population, leading to local extirpations and the loss of dispersal corridors (66 FR 38611). Nesting cuckoos are sensitive to habitat fragmentation that reduces patch size to less than 100 by 300 meters (Hughes 1999). Fragmentation of occupied habitats could make nest sites more accessible and more vulnerable to predation. Adults have been preyed upon by falcons (Hector 1985), and nestlings have been taken by hawks, jays, grackles (*Quiscalus quiscala*) (Nolan and Thompson 1975; Launer et al. 1990) and by various snake and mammal species (Nolan 1963). Predation is a significant source of nest failures, which have been recorded at 80 percent in some areas (Hughes 1999). In addition, pesticide use associated with agricultural practices may also pose a long-term threat to cuckoos. Pesticides may affect behavior and cause death or potentially affect prey populations (Hughes 1999; 66 FR 38611).

Overuse by livestock has been a major factor in the degradation and modification of riparian habitats in the western United States. The effects include changes in plant community structure and species composition, and relative abundance of species and plant density. (Wiggins 2005). Harris et al. (1986) believed that termination of grazing along portions of the South Fork of the Kern River in California was responsible for increases in riparian vegetation.

Another likely factor in the loss and modification of the western yellow-billed cuckoo is the invasion by the exotic tamarisk (*Tamarisk* sp.). The spread and persistence of tamarisk has resulted in significant changes in riparian plant communities. In monotypic tamarisk stands, the most striking change is the loss of community structure. The multi-layered community of herbaceous understory, small shrubs, middle-layer willows, and overstory deciduous trees is often replaced by one monotonous layer. Plant species diversity has declined in many areas and relative species abundance has shifted in others. Other effects include changes in percent cover, total biomass, fire cycles, thermal regimes, and perhaps insect fauna (Rosenberg et al. 1991; Busch and Smith 1993). Conversion to tamarisk typically coincides with reduction or complete loss of bird species strongly

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[2] http://www.geocities.com/rainforest/canopy/6181/yolo.html.

associated with cottonwood-willow habitat, including the western yellow-billed cuckoo (Hunter et al. 1987; Hunter et al. 1988; Rosenberg et al. 1991).

West Nile virus is spreading throughout portions of the western United States and poses a threat to bird species. The National Wildlife Health Center of the U.S. Geological Survey (USGS) has identified the western yellow-billed cuckoo as a species that may be affected by West Nile virus (USGS 2003).

Significant data gaps relating to many aspects of the life history of the western yellow-billed cuckoo exist. Data gaps include spacing parameters, the capacity for producing offspring, sources of mortality, mating system dynamics, and population structure. Brood parasitism by the western yellow-billed Cuckoo requires further study to identify the physiological and behavioral controls associated with the production of extra eggs. The current extent and causes of eggshell thinning and the effects of pesticides on cuckoos and the availability of prey need to be understood (Laymon 1998). Furthermore, detailed censuses of declining western populations must continue to determine locations of remnant populations and viable sizes necessary for future conservation programs (Laymon 1980).

A habitat model developed by Gaines (1974) for the western yellow-billed cuckoo in the Sacramento Valley includes the following: patch size of at least 25 acres, at least 100.5 meters (330 feet) wide and 302 meters (990 feet) long, within 100.5 meters (330 feet) of surface water, and dominated by cottonwood/willow gallery forest with high-humidity microclimate. Halterman and Laymon (1989) further refined the model by classifying habitat patch sizes for suitability. A willow-cottonwood forest patch greater than 604 meters (1,980 feet) wide and greater than 81 ha (200 acres) is classified as optimum habitat; a patch 201 to 603.5 meters (660 to 1,980 feet) wide and 41.5 to 81 ha (102.5 to 200 acres) is suitable; a patch 100.5 to 201 meters (330 to 660 feet) wide and 20 to 40 ha (50 to 100 acres) is marginal, and smaller patches are unsuitable. Management objectives for the Sacramento Valley include six subpopulations of 25 pairs each to maintain viable populations sizes (Laymon 1998). To achieve this goal, it would be necessary to establish or preserve at least 6,070 ha (15,000 acres) of optimum/suitable habitat. As of 1998, only 2,367 ha (5,850 acres) of habitat were considered suitable (Laymon 1998).

Many large riparian areas along the Sacramento River in Tehama County and along the Feather River in Yuba and Sutter Counties appear to be unoccupied but apparently represent suitable habitat for western yellow-billed cuckoo (Gaines and Laymon 1984). In addition, factors determining local population fluctuations need to be fully understood in order to guide effective management actions to increase and stabilize populations at local carrying capacity.

A.8.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.8.3, *Habitat Requirements and Ecology* (Figure A-8). The model parameters include the following.

- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., Yolo Audubon Society records).
- Nesting/Foraging Habitat: This habitat includes all potentially suitable habitat. This habitat was modeled by selecting all mapped vegetation types as listed below that occur in patch sizes of 25 acres or greater and have a width of at least 330 feet.

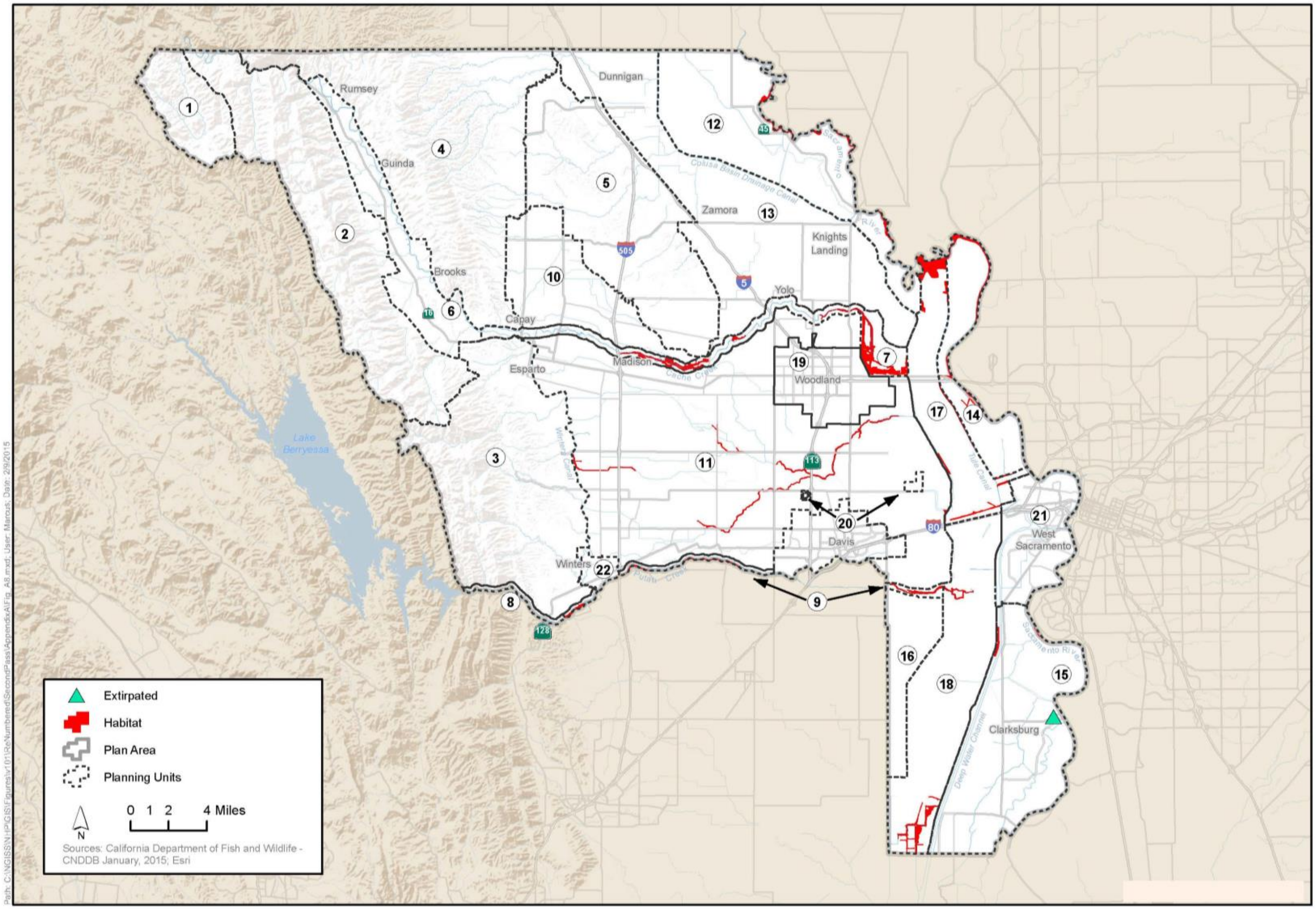
- Limited modeling to Planning Units: 7, 8, 9, 11, 12, 14, 15, 17, 18.

A.8.6.1 Nesting/Foraging Habitat – Vegetation Types

- Fremont Cottonwood – Valley Oak – Willow (Ash – Sycamore) Riparian Forest Not Formally Defined (NFD) Association
- Mixed Fremont Cottonwood – Willow spp. NFD Alliance
- Mixed Willow Super Alliance
- White Alder (Mixed Willow) Riparian Forest NFD Association

Undifferentiated Riparian Woodland/Forest

Figure A-8. Western Yellow-Billed Cuckoo Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

Path: C:\NGISS\NHP\GIS\Figures\v101\ReNumbered\SecondPass\AppendixA\Fig_A8.mxd; User: Marcus; Date: 2/9/2015

A.9 Western Burrowing Owl (*Athene cunicularia hypugaea*)

© Tom Greer

A.9.1 Listing Status

Federal: Species of Conservation Concern (U.S. Fish and Wildlife Service [USFWS] Regions 1, 2, and 6) (USFWS 2002).

State: Species of Special Concern.

Recovery Plan: None.

A.9.2 Species Description and Life History

Western burrowing owls (*Athene cunicularia hypugaea*) inhabit much of the western United States and southern interior of western Canada (Haug et al. 1993). They are unique among the North American owls in that they nest and roost in burrows. This small owl stands about 22.86 centimeters (9 inches) tall. The sexes are similar (although females are often slightly darker than males) with distinct oval facial ruff, white eyebrows, yellow eyes, and long stilt-like legs. Wings are relatively long (51–61 centimeters [20–24 inches]) and somewhat rounded. The owl is sandy colored with pale white spots on the head, back, and upperparts of the wings and white-to-cream with barring on the breast and belly (Haug et al*.* 1993).

A.9.2.1 Seasonal Patterns

Burrowing owls are resident in northern California. The breeding season (defined as from pair bonding to fledging) generally occurs from February to August with peak activity occurring from April through July (Haug et al*.* 1993). Pairs may be resident at breeding sites throughout the year or migrate out of the breeding area during the nonnesting season. Some individual birds only winter in the region. Thus, the demographics of this species in the region are relatively dynamic. Burrowing owls have a strong affinity for previously occupied nesting and wintering habitats. They often return to burrows used in previous years, especially if they had been reproductively successful (Lutz and Plumpton 1999). Additionally, burrowing owls often return as breeding adults to the general area in which they were born. For these reasons, efforts that enhance productivity help to ensure continued use of burrows and territories.

Migration patterns vary among burrowing owls. As noted above, in Northern California burrowing owls are generally year-round residents although some may migrate from or migrate to other regions during winter. Those burrowing owls that do migrate often return to the same nesting territories in successive years.

A.9.2.2 Reproduction

Adults begin pair bonding and courtship in February through March. Following pair formation, a nest is established in the natal burrow and females lay a clutch of six to 11 eggs. Average clutch size is seven to nine. Eggs are incubated entirely by the female for a period of between 28 and 30 days. During this time, the female is provisioned with food by the male. Following hatching, the young remain in the natal burrow for two to four weeks, after which they begin to emerge from the burrow and can be observed roosting at the burrow entrance. The female begins hunting as young become less dependent. Adults also often relocate chicks to satellite burrows presumably to reduce the risk of predation (Desmond and Savidge 1998) and possibly to avoid nest parasites (Dechant et al. 2003). After approximately 44 days, young leave the natal burrow and by 49–56 days begin to hunt live insects. On average, three to five young fledge, but fledging rates can range from a single chick to as many as eight or nine (Lutz and Plumpton 1999). During this time, the juveniles expand their range and may find cover in the satellite burrow. The juveniles continue to be provisioned by the adults until mid-September when they molt into adult plumage and begin to disperse (Landry 1979). King and Belthoff (2001) report that dispersing young use satellite burrows in the vicinity of their natal burrows for about two months after hatching before departing the natal area.

A.9.2.3 Home Range/Territory Size

Few valid measures of territory or home range size of burrowing owls have been published; home range has not often been measured directly (e.g., via telemetry studies), and is highly subject to observer bias or equipment effect. Accordingly, caution is warranted when interpreting home range estimates. Gervais et al. (unpublished 2000 report) estimated that the mean minimum convex polygon (MCP) home range estimates for 22 burrowing owls in Fresno and Kings Counties, California was 1.89 square kilometers (km^2) (467 acres). Haug and Oliphant (1990) estimated that the mean MCP for six owls in Saskatchewan was 2.41 km^2 (595 acres).

In Colorado, Plumpton and Lutz (D. Plumpton pers. comm.) recorded densities of nesting burrowing owls that ranged from 21 to 34 pairs on roughly 9.06 km^2 (2,240 acres) of available habitat (i.e., 0.43 $km^2$and 0.26 km^2 [106 and 65 acres]/pair, respectively). Thomsen (1971) estimated territory size based on nearest-neighbor distances between nest burrows, producing a result of six pairs of owls averaging 0.008 km^2, with a range of between 0.0004 to 0.016 km^2 (1.98 acres; range: 0.1 to 4.0 acres). The preceding values demonstrate the disparity among studies, the different values attained when using different methods of estimating abundance, and the risk in relying on the results of a single study.

A.9.2.4 Foraging Behavior and Diet

Burrowing owls are active day and night and will hunt throughout the 24-hour day, but are mainly crepuscular, hunting mostly at dusk and dawn, and are less active in the peak of the day. They tend to hunt insects in daylight and small mammals at night. They usually hunt by walking, running, hopping along the ground, flying from a perch, hovering, and fly-catching in mid-air.

Burrowing owls tend to be opportunistic feeders. Large arthropods, mainly beetles and grasshoppers, comprise a large portion of their diet. In addition, small mammals, especially mice and voles (*Microtus, Peromyscus*, and *Mus* spp.) are also important food items. Other prey animals include reptiles and amphibians, young cottontail rabbits, bats, and birds, such as sparrows and horned larks. Consumption of insects increases during the breeding season (Zarn 1974; Tyler 1983; Thompson and Anderson 1988; John and Romanow 1993; Green et al. 1993; Plumpton and Lutz 1993a). Productivity may increase in proportion to the amount of mice and voles in the diet (D. Plumpton unpublished data).

As with most raptors, burrowing owls select foraging areas based on prey availability as well as prey abundance. Prey availability (the ability of a raptor to detect prey) decreases with increasing vegetative cover, thus foraging habitat suitability decreases with increasing grass height or vegetative density.

A.9.3 Habitat Requirements and Ecology

Burrowing owls are found in open, dry grasslands, agricultural and range lands, and desert habitats often associated with burrowing animals (Klute et al. 2003). They also occupy golf courses, airports, road and levee embankments, and other disturbed sites where there is sufficient friable soil for burrows (Haug et al. 1993). Because they typically use the burrows created by other species, particularly the California ground squirrel (*Spermophilus beecheyi*), presence of these species is usually a key indicator of potential occurrence of burrowing owl (Gervais et al*.* 2008). Burrowing owls in cismontane California were likely historically associated with herbaceous vegetation suppressed by tule elk herds.

A.9.3.1 Nesting

In northern California, most nest sites occur in abandoned ground squirrel burrows; however, other mammal burrows and various artificial sites, such as culverts, pipes, rock piles, and artificially constructed burrows are also used (Gervais et al. 2008). Burrowing owls generally select sites in relatively sandy habitats that allow for modification of burrows and maximize drainage. In addition to providing nesting, roosting, and escape burrows, ground squirrels improve habitats for burrowing owls in other ways. Burrowing owls favor areas with short, sparse vegetation (Coulombe 1971; Haug and Oliphant 1990; Plumpton and Lutz 1993b) to facilitate viewing and hunting, which is typical around active sciurid colonies. Additionally, burrowing owls may select areas with a high density of burrows (Plumpton and Lutz 1993b). Typical habitats are treeless, with minimal shrub cover and woody plant encroachment, and have low vertical density of vegetation and low foliage height diversity (Plumpton and Lutz 1993b). While occupied burrows are sometimes found in flat landscapes, often in elevated mounds created by burrowing activity, they are also commonly found on hillsides, levee slopes, or other vertical cuts, probably to facilitate drainage and maximize visibility. Nest sites are also often associated with nearby perches, including stand pipes, fences, or other low structures.

Optimal nesting locations are within an open landscape with level to gently sloping topography, sparse or low grassland or pasture cover, and a high density of burrows.

Burrowing owls are tolerant of human-altered open spaces, such as areas surrounding airports, golf courses, and military lands, where burrows are readily adopted (Thomsen 1971; Gervais et al. 2008). Burrowing owls may select areas adjacent to unimproved and improved roads (Brenckle 1936; Ratcliff 1986); a modest volume of vehicle traffic does not appear to significantly affect behaviors or reproductive success (Plumpton and Lutz 1993c). In the South San Francisco Bay region, in the Sacramento area, and in several locations in and around the City of Davis, burrowing owls nest and winter in highly human-affected environments and can adjust to most types of human activity if habitats remain in a suitable condition.

The dimensions of the nest burrow vary with location, age of burrow, and the species that originally excavated it. Typical burrows constructed by ground squirrels are from 3 to 6 inches in diameter and extend underground at a gradual downward slope from 3 to 10 feet with an enlarged cavity at the end of the burrow. Burrow entrances are often adorned with various objects as well as feathers and pellets. The burrow is often lined with grass or other material (Haug et al. 1993).

Burrowing owls are solitary nesters or may nest in loose colonies – usually from 4 to 10 pairs (Zarn 1974); however, larger colonies have been documented. Most pairs occupy a natal burrow and at least one additional satellite burrow.

As semi-colonial owls, colony size is indicative of habitat quality. Colony size is also positively correlated with annual site reuse by breeding burrowing owls; larger colonies (those with more than five nesting pairs) are more likely to persist over time than colonies containing fewer pairs or single nesting pairs (DeSante et al. 1997). Nest burrow reuse by burrowing owls has been well documented (Martin 1973; Gleason 1978; Rich 1984; Plumpton and Lutz 1993b; Lutz and Plumpton 1999). Former nest sites may be more important to continued reproductive success than are mates from previous nest attempts (Plumpton and Lutz 1994). Past reproductive success may influence future site re-occupancy by burrowing owls. Female burrowing owls with large broods tend to return to previously occupied nest sites, while females that failed to breed or produced small broods may change nest territories in subsequent years (Lutz and Plumpton 1999).

In general, burrowing owls show a high degree of nest site fidelity and reuse the same nesting burrows and satellite burrows for many years if left undisturbed (Haug et al. 1993).

A.9.3.2 Foraging

Burrowing owls forage in open grasslands, pasturelands, agricultural fields and field edges, fallow fields, and along the edges of roads and levees. Vegetation is low to maximize visibility and access. Short perches such as fence posts are often used to enhance visibility. While they will defend the immediate vicinity of the nest, burrowing owls will often forage in common areas (Haug et al. 1993).

A.9.4 Species Distribution and Population Trends

A.9.4.1 Distribution

There are two subspecies of burrowing owls in North America (Dechant et al. 2003). The breeding range of *A. cunicularia floridana* is restricted to Florida and adjacent islands. The breeding range of *Athene cunicularia hypugaea* extends south from southern Canada throughout most of the western half of the United States and south to central Mexico. The winter range is similar to the breeding range except that most owls from the northern areas of the Great Plains and Great Basin migrate south and southern populations are resident year-round (Haug et al. 1993).

Burrowing owls were once widespread and generally common over western North America, in treeless, well-drained grasslands, steppes, deserts, prairies, and agricultural lands (Haug et al. 1993). The owl’s range has contracted in recent decades, and populations have been generally diminished in some areas.

In California, burrowing owls are widely distributed in suitable habitat throughout the lowland portions of the state; however, occupied sites have ranged from 200 feet below sea level at Death Valley to above 12,000 feet at Dana Plateau in Yosemite National Park (California Department of Fish and Game [DFG] 2000; Gervais et al. 2008). In southern California, the species is fairly common along the Colorado River Valley (Rosenberg et al. 1991) and in the agricultural region of the Imperial Valley. Only small, scattered populations are thought to occur in the Great Basin and the desert regions of southern California (DeSante et al. 1997). Burrowing owl breeding populations have greatly declined along the California coast, including the southern coast to Los Angeles, where these owls have been eliminated from virtually all private land, and occur only in small populations on some federal lands (Trulio 1997; Garrett and Dunn 1981). Breeding populations in Central California include the southern San Francisco Bay south of Alameda and Redwood City, the interior valleys and hills in the Livermore area, and the Central Valley (DeSante et al. 1997; Gervais et al. 2008).

The current distribution of burrowing owls in Yolo County is localized primarily in remaining low elevation uncultivated areas, such as the grasslands along the western edge of the Central Valley, the pasturelands in the southern panhandle, and the Yolo Bypass Wildlife Area. Other sites include some urban and semi-urban areas, particularly in and around the City of Davis, and other scattered locations associated with edges of cultivated lands.

While comprehensive surveys of the plan area have not been conducted, coordinated surveys have been undertaken in portions of the county. The majority of recent information is a result of these efforts, including monitoring surveys in and around the City of Davis (McNerney pers. comm.); surveys conducted by the California Department of Fish and Game at the Yolo Bypass Wildlife Area; and surveys coordinated by the Burrowing Owl Preservation Society in coordination with the Institute of Bird Populations on 12 selected 5-square-mile survey blocks in Yolo County in 2007 and 2014 (Wilkerson pers. comm., Catherine Portman pers. comm.). Additional data is gathered and reported incidentally by knowledgeable individuals from other areas of the County.

The results of these surveys and incidental reports indicate that the majority of known burrowing owl breeding locations are in the southern portion of Yolo County, centered in and around the City of Davis, the Yolo Bypass Wildlife Area, and the southern panhandle. A total of 50 breeding pairs were reported in Yolo County in 2007 (Table A9-1), and surveys of these same sites in 2014 indicated that only 15 breeding pairs were present in these locations. These data represent only reported sightings from several locations in Yolo County where surveys were conducted and data were recorded and made available. This summary does not represent the total number of burrowing owl breeding pairs in the county. However, it does represent the most significant known breeding areas for burrowing owl in Yolo County.

During 2010 and 2011, there were 6 documented burrowing owl nests northeast of Davis along the north side of CR 28H between CR 102 and 104 (Whistler pers. comm.). During 2015, Whisler observed only one pair of burrowing owl north of CR 28H, just west of CR 104. This pair was in the former ConAgra (Hunt-Wesson) property nesting on a dirt mound.

A.9.4.2 Population Trends

Overall population trend throughout the subspecies' North American range is reportedly declining. James (1993) reports that 54 percent of the areas sampled reported declining burrowing owl populations. Breeding Bird Surveys (BBS) conducted between 1980 and 1989 also report significant declines in many areas (Haug et al. 1993).

Burrowing owl was formerly common or abundant throughout much of California, but a decline noticeable by the 1940s (Grinnell and Miller 1944) has continued to the present time. The decline has been almost universal throughout California. Conversion of grasslands and pasturelands to incompatible crop types and the destruction of ground squirrel colonies have been the main factors causing the decline of the burrowing owl population (Zarn 1974; Gervais et al. 2008). Assimilation of poisons applied to ground squirrel colonies also affects burrowing population levels (Gervais et al. 2008).

A census of burrowing owls from 1991 to 1993 (DeSante et al. 1997) estimated there were approximately 10,000 pairs of burrowing owls in California. Over 70 percent of the owls in California are in the Imperial Valley, an area that represents less than 2 percent of the state's landmass (D. Plumpton pers. comm.). Numbers have been declining for decades in several areas of the state. Owls are extinct or have been reduced to very low numbers in several parts of the state, including coastal southern California and parts of the San Francisco Bay area. The statewide census indicated there has been a 50 percent decline in numbers of owls and the number of breeding groups in some parts of the state from the 1980s to 1990s.

Although California has a significant burrowing owl population, development pressures and recent population trends suggest that the species may continue to be extirpated from large portions of its range in California during the next decade. In the San Francisco Bay area, burrowing owls are commensal with the California ground squirrel and reside in undeveloped grassland remnants amid a rapidly expanding human population. An estimated 167 nesting pairs (1.8 percent of California's population) remain (all figures as of 1991, based on DeSante et al. [1997]), representing a decline of approximately 50 percent since the mid-1980s. In the southern California coastal population, burrowing owls have been almost entirely extirpated from private lands and are now found only on a few undeveloped federal lands, where an estimated 260 nesting pairs (3 percent of California's population) persist. An estimated 2,224 nesting pairs exist in the Central Valley (24 percent of California's population). Burrowing owls are mostly commensal with the round-tailed ground squirrel (*Spermophilus tereticaudus*) in the Imperial Valley, where burrowing owls are almost completely relegated to irrigation canal banks and where an estimated 6,570 nesting pairs (71 percent of California's population) remain (all data from DeSante et al. 1997, presented also in Barclay et al. 1998).

Table A-7. Breeding Season Burrowing Owl Occurrences Reported from Yolo County in 2007

Location	No. of Breeding Pairs	No. of Unpaired Singles	Total No. of Adults	No. of Young
Davis city limits	21	6	48	61
Yolo Bypass Refuge	19		38	60
Davis vicinity	4	4	12	
Woodland vicinity	3		6	
South panhandle	3		6	11
Total	**50**	**10**	**110**	**132**

There is evidence that the overall population in the county has declined based on severe declines or extirpations of known colonies. For example, the owl colony on the University of California, Davis campus had declined from 22 pairs in 1981 to one pair in 1991, then rebounded to several pairs in the late 1990s (Johnson pers. comm.). Another colony of 10 pairs documented in 1976 near the Yolo County Airport had been eliminated when the location was flooded in 1983 to create a pond (California Natural Diversity Database [CNDDB] 2007). More recently, a small colony on the north side of Winters was displaced by grading activities in preparation of a new development project.

However, burrowing owls have increased or continue to be relatively stable during the last several years in other areas, such as the Mace Ranch Preserve and the Wildhouse agricultural buffer and golf course (McNerney pers. comm.) in the Davis area. Habitat restoration efforts by the California Department of Fish and Wildlife (DFW) at the Yolo Bypass Wildlife Area may also be responsible for the increase in reported occurrences of owls at that location. Thus, in some areas owls appear to respond favorably to protection and restoration efforts.

A.9.5 Threats to the Species

Urbanization, including residential and commercial development and infrastructure development (roads and oil, water, gas, and electrical conveyance facilities) is one of the principal causes of habitat loss for burrowing owls and is a continuing threat to remaining northern California populations. Urbanization permanently removes habitat and has led to permanent abandonment of many burrowing owl colonies in the developing portions of the Central Valley, Bay Area, and throughout the state (Gervais et al. 2008).

Burrowing owls have shown a high level of tolerance for human encroachment, degradation of native habitats, and fragmentation of habitats (Gervais et al. 2008). Owls will often continue to occupy traditional sites as long as essential habitat elements remain present and until the extent of available habitat is reduced below the species' habitat requirement thresholds. Some burrowing owls nest on the edges of agricultural areas and forage in suitable agricultural landscapes, such as recently harvested fields, alfalfa and other hay fields, irrigated pastures, and fallow fields. The conversion of these fields to incompatible crop types, such as orchards, vineyards, and other crops that are not conducive to burrowing owl foraging, reduce available foraging habitat and lead to abandonment of traditional nesting areas. Many burrowing owl nests are known to occur along the outside slope or at the toe of levees. Levee stability practices for flood control, including vegetation removal, grading, and reinforcing with rock can destroy burrowing owl nesting habitat.

Although burrowing owls are relatively tolerant of low levels of human activity, human-related impacts such as shooting and burrow destruction adversely affect this species (Zarn 1974; Haug et al. 1993). Rodent control, particularly along levees and roadsides, can decimate ground squirrel populations and ultimately reduce available nesting and cover habitat for burrowing owls. Artificially enhanced populations of native predators (e.g., gray foxes, coyotes) and introduced predators (e.g., red foxes, cats, dogs) near burrowing owl colonies can also be a significant local problem. Burrowing owls also get tangled in loose fences, abandoned wire, fishing line, rat traps, and other materials.

The overall effect of population-level threats (e.g., habitat conversion or ground squirrel eradication) is of much greater concern than sources of individual mortality (e.g., shooting or vehicle collisions), as these former forces operate at a population, regional, and/or range-wide level. As obligate burrow nesters that do not excavate their own burrows, burrowing owls are largely dependent on burrowing mammals that have no legal status or protection, and are commonly and purposefully eradicated by humans. Whereas individual mortality cumulatively represents a significant number of individuals, a population that is secure and productive can offset these losses. Conversely, populations that are failing because of population-level effects cannot be sustained even in absence of direct sources of individual mortality. In California, significant economic development pressures exist, and habitat conversion for human purposes continues to degrade the abundance and quality of owl nesting habitat (Barclay et al. 1998). Few provisions exist to protect habitats over time. As a result, burrowing owls appear to be declining throughout most of California.

Important conservation milestones, such as the investigation and rejection of the case for changing the status of the burrowing owl to either threatened or endangered at the state or federal levels, have been reached in recent years. Significant data gaps exist in regard to migration, dispersal from nesting sites, and other aspects of annual movements. Small body size and habit of dwelling in burrows make the burrowing owl a poor choice for study using radio telemetry. Accordingly, much of what is known is the result of leg-banding studies that rely on visual detection or physical recapture of previously banded owls. These results are very specific to location, based on small sample sizes, and subject to observer effects. Accordingly, these data are not reliable for inference across the range of these owls, and should not be extrapolated to a specific location. Anecdotal accounts offer the most locality-specific data on dispersal, but few reliable data exist.

Burrowing owls are known to reoccupy habitats over their lifespan, if these habitats remain suitable (Rich 1984; Lutz and Plumpton 1999). Accordingly, preservation of large areas of consistently suitable habitat is the most important management and conservation option available. These habitats will include native grasslands that also support the native suite of species—including ground squirrels—that dig burrows, and prey such as voles, mice, ground beetles, and grasshoppers.

A.9.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.9.3, *Habitat Requirements and Ecology* (Figure A-9). The model parameters include the following.

- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (e.g., CNDDB, Burrowing Owl Preservation Society, City of Davis, Yolo Basin Wildlife Area).
- Primary Habitat: This habitat includes all potentially suitable habitat in preferred natural habitats, pastures, and other open or barren areas on the lower slopes and valley floors. This habitat was modeled by selecting all mapped land cover types as listed below, where they occur in the Central Valley, Dunnigan Hills, and Yolo Bypass ecoregions.
- Other Habitat: This habitat includes selected pasture types, where uncultivated field borders may be suitable for potential nesting burrows and fields may be suitable for foraging. This habitat was modeled by selecting all pasture types except for turf farms, within the Central Valley, Dunnigan Hills, and Yolo Bypass ecoregions.
- Added Land Cover that was had the vegetation type 'Semi-Agriculture/Incidental to Agriculture' that was within 50' of habitat that was modeled with the aforementioned criteria.

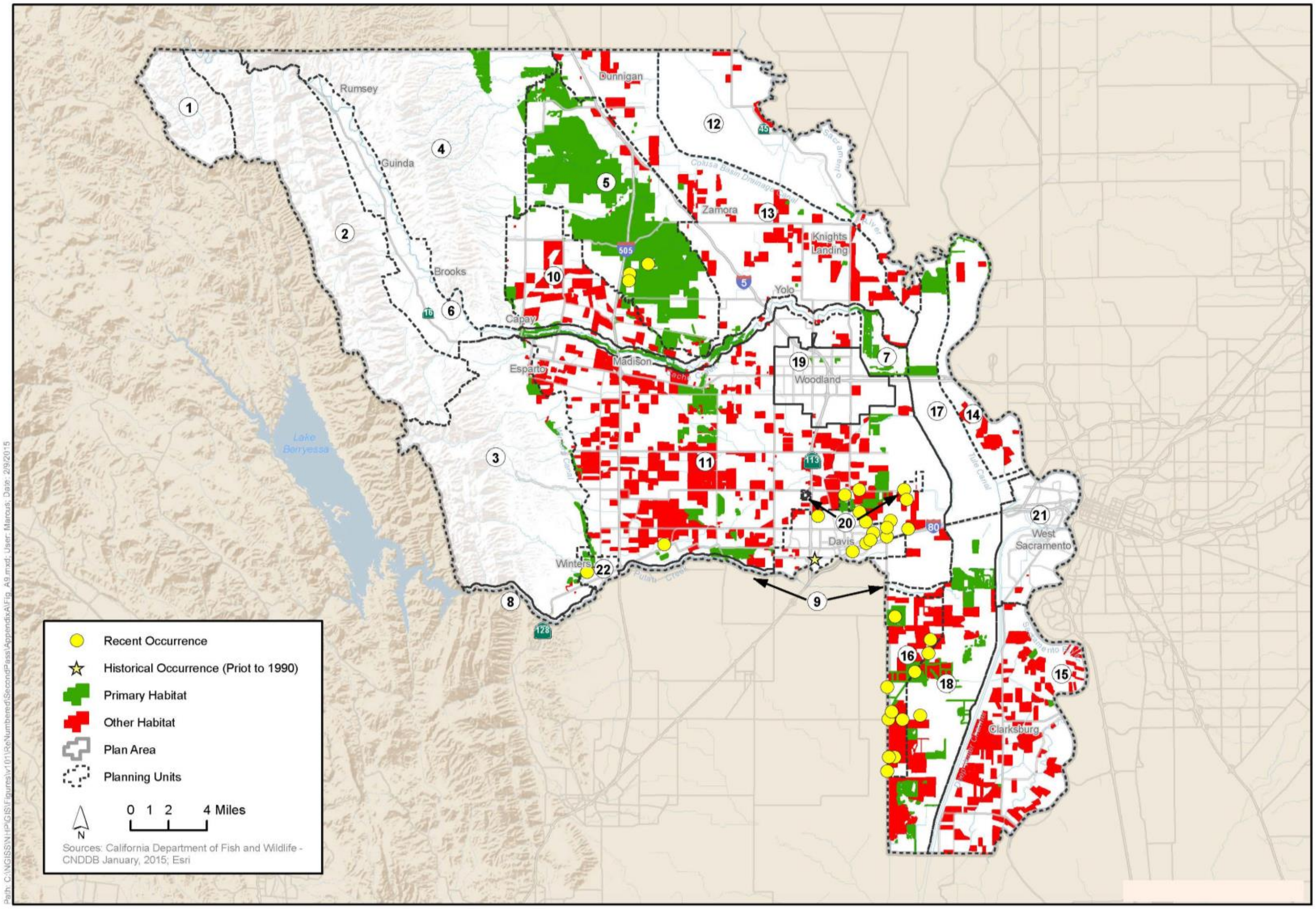
A.9.6.1 Primary Habitat – Vegetation Types

- California Annual Grasslands Alliance
- Upland Annual Grasslands and Forbs Formation
- Barren – Anthropogenic
- Native Pasture

A.9.6.2 Other Habitat – Vegetation Types

- Mixed Pasture
- Miscellaneous Grasses (grown for seed)
- Alfalfa

Figure A-9. Western Burrowing Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

A.10 Least Bell's Vireo (*Vireo bellii pusillus*)

A.10.1 Listing Status

Federal: Endangered.

State: Endangered.

Recovery Plan: The U.S. Fish and Wildlife Service (USFWS) prepared a *Draft Recovery Plan for the least Bell's Vireo (*Vireo bellii pusillus*)* in 1998 (USFWS 1998).

A.10.2 Species Description and Life History

Least Bell's vireo is the smallest subspecies of the Bell's vireo (*Vireo bellii*). The Bell's vireo can range from 4.3 to 4.7 inches (11–12 centimeters) in length and has a wingspan of 7.1 inches (18 centimeters). It weighs approximately 0.2–0.4 ounce (7–10 grams) (Brown 1993). It is drably colored and indistinctly marked. The least Bell's vireo is the grayest subspecies of Bell's vireo and has very little yellow or green in its plumage.

A.10.2.1 Seasonal Patterns

Least Bell's vireos are migratory and usually arrive to their California breeding grounds in mid-March to early April from their wintering grounds in Mexico. Observations of banded birds suggest that returning adult breeders may arrive earlier than first-year birds by a few weeks (Kus unpublished data in Kus 2002a). Least Bell's vireos begin departing for their wintering grounds by late July but are generally present on their breeding grounds until late September (Garrett and Dunn 1981; Salata 1983).

A.10.2.2 Reproduction

Egg-laying begins one to two days after nest completion. Typically three to four eggs are laid. Average clutch sizes of nonparasitized nests observed with complete clutches have ranged from 3.1 to 3.9 in recent years. Both parents share in incubation, which takes approximately 14 days. After hatching, nestlings are fed by both parents for 10 to 12 days until fledging (USFWS 1998). Adults continue to care for the young at least two weeks after fledging when territorial boundaries may be relaxed as family groups range over larger areas. Fledglings usually remain in the territory or its vicinity for most of the season. Least Bell's vireo pairs may attempt up to five nests in a breeding season, although most fledge young from only one or two. Few nests are initiated after mid-July. Long-term annual rates of hatching success (the percentage of eggs laid that hatch) have ranged from 53 to 83 percent in the major study populations at the San Luis Rey, Santa Margarita, and Tijuana Rivers. The annual average number of fledglings produced per pair has ranged from 0.9 to 4.5, with long-term averages ranging between 1.8 and 3.2 (USFWS 1998).

Nests are typically placed in the fork of a tree or shrub branch in dense cover within 3 to 6 feet (1 to 2 meters) of the ground. Both members of the pair construct the cup-shaped nest from leaves, bark, willow catkins, spider webs, and other material, in about four to five days. The female selects the nest site (Bent 1950; Barlow 1962). Nests are placed in a wide variety of plant species, but the majority are placed in willows (*Salix* spp.) and mule fat (*Baccharis glutinosa*). Nests tend to be placed in openings along the riparian edge, where exposure to sunlight allows the development of shrubs.

A.10.2.3 Home Range/Territory Size

Territory size ranges from 0.5 to 7.5 acres, but on average are between 1.5 and 2.5 acres in southern California (USFWS 1998). Newman (1992) investigated the relationship between territory size, vegetation characteristics, and reproductive success for populations in San Diego County, but found no significant factors that could account for the variability in territory size found at his sites. Spatial differences in riparian habitat structure, patch size, and numerous other factors result in differences in the density of territories within and between drainages. Embree (1992) concluded that patch size and crowding did not influence least Bell's vireo reproductive success, at least not through the mechanisms of singing rates and attraction of predators.

A.10.2.4 Foraging Behavior and Diet

Least Bell's vireos are insectivorous and prey on a wide variety of insects, including bugs, beetles, grasshoppers, moths, and especially caterpillars (Chapin 1925; Bent 1950). They obtain prey primarily by foliage gleaning (picking prey from leaf or bark substrates) and hovering (removing prey from vegetation surfaces while fluttering in the air). Foraging occurs at all levels of the canopy but appears to be concentrated in the lower to mid-strata, particularly when pairs have active nests (Grinnell and Miller 1944; Goldwasser 1981; Gray and Greaves 1981; Salata 1983; Miner 1989). Miner (1989) determined that least Bell's vireo foraging time across heights was not simply a function of the availability of vegetation at those heights; rather; it represented an actual preference for the 3-to-6–meter zone. Foraging occurs most frequently in willows (Salata 1983; Miner 1989), but occurs on a wide range of riparian species and even some non-riparian plants that may host relatively large proportions of large prey (Miner 1989).

A.10.3 Habitat Requirements and Ecology

A.10.3.1 Nesting

The least Bell's vireo is an obligate riparian breeder that typically inhabits structurally diverse woodlands, including cottonwood-willow woodlands/forests, oak woodlands, and mule fat scrub (USFWS 1998). Two features appear to be essential for breeding habitat: (1) the presence of dense cover within 3 to 6 feet (1 to 2 meters) of the ground, where nests are typically placed; and (2) a dense stratified canopy for foraging (Goldwasser 1981; Gray and Greaves 1981; Salata 1981, 1983; RECON 1989). While least Bell's vireo typically nests in willow-dominated areas, plant species composition does not seem to be as important a factor as habitat structure.

Early successional riparian habitat typically supports the dense shrub cover required for nesting and a diverse canopy for foraging. While least Bell's vireo tends to prefer early successional habitat, breeding site selection does not appear to be limited to riparian stands of a specific age. If willows and other species are allowed to persist, within five to 10 years they form dense thickets and become suitable nesting habitat (Goldwasser 1981; Kus 1998). Tall canopy tends to shade out the shrub layer in mature stands, but least Bell's vireo will continue to use such areas if patches of understory exist. In mature habitat, understory vegetation consists of species such as California wild rose (*Rosa californica*), posion oak (*Toxicodendron diversiloba*), California blackberry (*Rubus ursinus*), grape (*Vitis californica*), and perennials that can conceal nests. Nest site characteristics are highly variable and no features have been identified that distinguish nest sites from the remainder of the territory (Hendricks and Rieger 1989; Olson and Gray 1989; RECON 1989).

A.10.3.2 Foraging

Least Bell's vireos forage primarily within and at all levels of the riparian canopy (Salata 1983); however, they will also use adjacent upland scrub habitat, in many cases coastal sage scrub. In addition to use as foraging habitat, these areas also provide migratory stopover grounds and dispersal corridors for non-breeding adults and juveniles (Kus and Miner 1989; Riparian Habitat Joint Venture [RHJV] 2004). Vireos along the edges of riparian corridors maintain territories that incorporate both habitat types, and a significant proportion of pairs with territories encompassing upland habitat place at least one nest there (Kus and Miner 1989).

Little is known about least Bell's vireo wintering habitat requirements. They are not exclusively associated with riparian habitat during winter, and can occur in mesquite scrub vegetation to a greater degree than riparian areas in winter (Kus unpublished data in USFWS 2006). Least Bell's vireo may also occur in palm groves or along hedgerows associated with agriculture and rural residential areas.

A.10.4 Species Distribution and Population Trends

A.10.4.1 Distribution

The least Bell's vireo is one of four subspecies of Bell's vireo and is the only subspecies that breeds entirely in California and northern Baja California. *V. bellii arizonae* is found along the Colorado River and may occur on the California side, but otherwise occurs throughout Arizona, Utah, Nevada, and Sonora, Mexico.

1. A riparian obligate, the historical distribution of the least Bell's vireo extended from coastal southern California through the San Joaquin and Sacramento valleys as far north as Tehama County near Red Bluff. The Sacramento and San Joaquin valleys were considered the center of the species' historical breeding range supporting 60 to 80 percent of the historical population (51 FR 16474). The species also occurred along western Sierra foothill streams and in riparian habitats of the Owens Valley, Death Valley, and Mojave Desert (Cooper 1861 and Belding 1878 in Kus 2002a; Grinnell and Miller 1944). The species was reported in Grinnell and Miller (1944) from elevations ranging from -175 feet in Death Valley to 4,100 feet at Bishop, Inyo County. These and other historical accounts described the species as common to abundant, but no reliable population estimates are available prior to the species' federal listing in 1986. The last known nesting pair of LBVI in the Sacramento Valley was observed in 1958 (Cogswell 1958, Goldwasser 1978).

   During 2010-2013, least Bell's vireo surveys were conducted in the Putah Creek Sinks located in the Yolo Bypass Wildlife Area (Whisler 2013, 2015). The focus of this study was to determine whether least Bell's vireos were breeding in the Putah Creek Sinks. The field survey methods were consistent with the U.S. Fish and Wildlife Service (2001) least Bell's vireo survey guidelines and the Yolo Audubon Society's Yolo County Breeding Bird Atlas survey methods.

Least Bell's vireos were observed during the 2010 and 2011 breeding seasons; none were detected during 2012, and one individual was observed in May 2013. Brown-headed cowbirds were common in the survey area during each year.

During 2010, two pairs of least Bell's vireos were observed in the survey area along with one or two additional individuals. Both pairs of vireos were observed performing courtship activities and territorial defense against other least Bell's vireos. On April 26, an adult least Bell's vireo was observed carrying nesting material. There was no evidence of successful nesting by least Bell's vireos. No obvious signs of nesting (e.g., active nests, fledglings, or adults carrying food) were observed during the surveys. The territories were occupied throughout the typical nesting season (April through mid-August).

In 2011**, t**he two 2010 least Bell's vireo territories were occupied by two least Bell's vireo pairs. The male in each pair was observed singing and defending the territory, signs of breeding behavior. Courtship activities were observed in one of the two pairs. One male was also defending its territory from a third adult. There were no further least Bell's vireo detections in late July or August of 2011.

There were no least Bell's vireo detections during 2012. Apparently the birds did not return to the survey area or they were not detected. One vireo was detected in 2013 on May 9, but none were detected after that date. 2015 surveys are ongoing (Whisler et al. 2015).

A.10.4.2 Population Trends

Coinciding with widespread loss of riparian vegetation throughout California (Katibah 1984), Grinnell and Miller (1944) began to detect population declines in the Sacramento and San Joaquin Valley region by the 1930s. Surveys conducted in late 1970s (Goldwasser et al. 1980) detected no least Bell's vireos in the Sacramento-San Joaquin Valleys, and the species was considered extirpated from the region. By 1986, the USFWS determined that least Bell's vireo had been extirpated from most of its historical range and numbered approximately 300 pairs statewide (51 FR 16474). The historical range was reduced to six California counties south of Santa Barbara, with the majority of breeding pairs in San Diego County (77 percent), Riverside County (10 percent), and Santa Barbara County (9 percent) (51 FR 16474).

Since federal listing in 1986, populations have gradually increased and the species has recolonized portions of its historical range. Increases have been attributed primarily to riparian restoration and efforts to control the brood parasite brown-headed cowbird (Kus 1998 and Kus and Whitfield 2005 in Howell et al. in press). By 1998, the total population was estimated at 2,000 pairs and recolonization was reported along the Santa Clara River in Ventura County, the Mojave River in San Bernardino County, sites in Monterey and Inyo counties (Kus and Beck 1998; Kus 2002a; USFWS 2006), and a single nest reported from Santa Clara County near Gilroy in 1997 (Roberson et al. 1997). Still, the distribution remained largely restricted to San Diego County (76 percent) and Riverside County (16 percent) (USFWS 2006).

By 2005, the population had reached an estimated 2,968 breeding pairs (USFWS 2006) with increases in most Southern California counties and San Diego County (primarily Camp Pendleton Marine Corps Base) supporting roughly half of the current population (USFWS 2006).

Distribution and Population Trends in the Plan Area

Two singing least Bell's vireo males were detected, positively identified, and photographed in the southern portion of the Yolo Bypass Wildlife Area in Yolo County in mid-April 2010 and have subsequently returned in the spring of 2011 (J. P. Galván pers. comm.). The next closest recent record occurred in June 2005 and was approximately 66 miles south of the current record at the San Joaquin River National Wildlife Refuge in the San Joaquin and Tuolumne River floodplain (Howell et al. in press). In June 2005, least Bell's vireos were detected nesting at the San Joaquin River National Wildlife Refuge, west of Modesto in Stanislaus County, the first nesting record of the species in the Central Valley in over 50 years (Howell et al. in press). A single breeding pair nested at the refuge in 2005, 2006, and 2007. The pair successfully nested in 2005 and 2006 and the nest was depredated in 2007. No least Bell's vireos were detected in 2008 or 2009 (Howell et al. in press).

A.10.5 Threats to the Species

A major factor leading to declines in populations of least Bell's vireo is the loss and degradation of riparian woodland habitat throughout the species' range. Habitat loss and degradation can occur through clearing of vegetation for agriculture, timber harvest, development, or flood control. Flood control and river channelization eliminates early successional riparian habitat that least Bell's vireo (and many other riparian focal species) use for breeding. Dams, levees and other flood control structures hinder riparian reestablishment, creating more "old-growth" conditions (dense canopy and open understory) that are unfavorable to breeding vireos. Finally, habitat degradation encourages nest predation and parasitism. Agricultural land uses and water projects not only directly destroy habitat, but may also reduce water tables to levels that inhibit the growth of the dense vegetation least Bell's vireo prefer (RJHV 2004). Grazing can also have a significant effect on riparian vegetation (Sedgwick and Knopf 1987). Cattle and other livestock can trample vegetation and eat seedlings, saplings, shrubs, and herbaceous plants. This can lead to a reduction in cover and nesting sites, and affect insect prey populations. Insecticides may also be a threat to this species since it is insectivorous and its greatest declines are in areas with intensive agriculture (Holstein 2003).

Brood parasitism from brown-headed cowbirds (*Molothrus ater*) has a major negative impact on least Bell's vireo. Livestock grazing has reduced and degraded the lower riparian vegetation favored by the Least Bell's Vireo (Overmire 1962) and provided foraging areas for the brown-headed cowbird. Row crops and orchards also provide feeding grounds for the parasite. By as early as 1930, nearly every least Bell's vireo nest found in California hosted at least one cowbird egg (USFWS 1998). Since a parasitized nest rarely fledges any vireo young, nest parasitism of least Bell's vireo results in drastically reduced nest success (Goldwasser 1978; Goldwasser et al. 1980; Franzreb 1989; Kus 1999; Kus 2002b).

Predation is a major cause of nest failure in areas where brown-headed cowbird nest parasitism is infrequent or has been reduced by cowbird trapping programs. Most predation occurs during the egg stage. Predators likely include western scrub jays (*Aphelocoma californica*), Cooper's hawks (*Accipiter cooperii*), gopher snakes (*Pituophis melanoleucus*) and other snake species, raccoons (*Procyon lotor*), opossums (*Didelphis virginiana*), coyotes (*Canis latrans*), long-tailed weasels (*Mustela frenata*), dusky-footed woodrats (*Neotoma fuscipes*), deer mice (*Peromyscus maniculatus*), rats (*Rattus* spp.), and domestic cats (*Felis domesticus*) (Franzreb 1989).

A.10.6 Species Habitat Model and Location Data

The habitat model for this species was based on known recent sightings and the distribution of land cover types that are known to support its habitat as described above in Section A.27.3, *Habitat Requirements and Ecology* (Figure A-27).

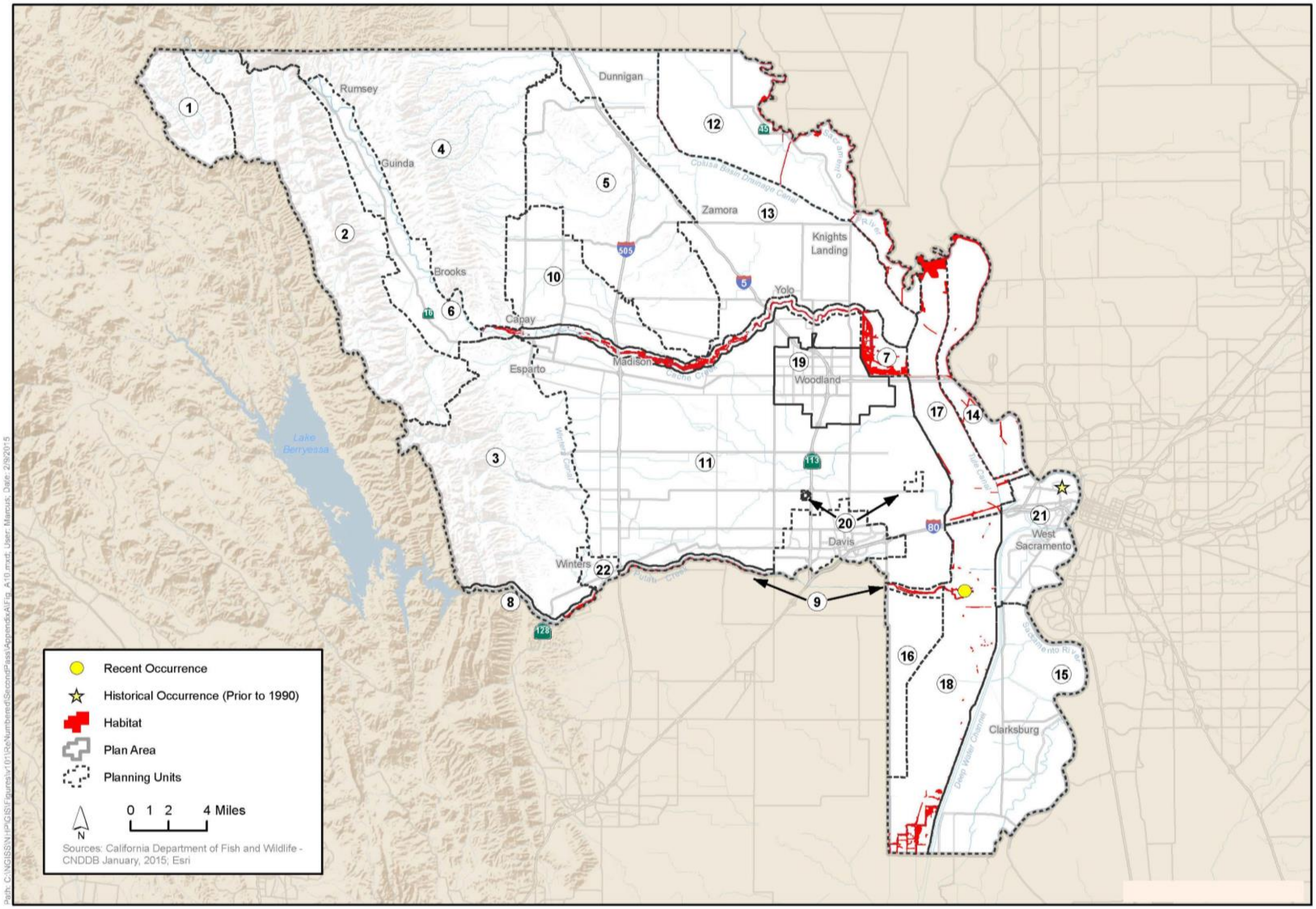
The model parameters include the following.

- Known Recent Sightings: Location where the species has relatively recently (post-January 1, 1980) been documented according to one or more species locality records databases (e.g., California Natural Diversity Database [CNDDB], BIOS, University of California, Davis Museums collections, etc.).
- Nesting/Foraging Habitat: This habitat includes all potentially suitable breeding and foraging riparian areas and was modeled by selecting all mapped vegetation types as listed below.
- Limited modeling to Planning Units: 7, 9, 12, 14, 17, 18.

A.10.6.1 Nesting/Foraging Habitat – Vegetation Types

- Blackberry Not Formally Defined (NFD) Super Alliance
- Coyote Bush
- Fremont Cottonwood – Valley Oak – Willow (Ash – Sycamore) Riparian Forest NFD Association
- Mixed Fremont Cottonwood – Willow spp. NFD Alliance
- Mixed Willow Super Alliance
- White Alder (Mixed Willow) Riparian Forest NFD Association
- Undifferentiated Riparian Bramble
- Undifferentiated Riparian Woodland/Forest

Figure A-10. Least Bell's Vireo Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

Path: C:\NGIS\SNHP\GIS\Figures\v101\ReNumbered\SecondPass\AppendixA\Fig_A10.mxd; User: Marcus; Date: 2/9/2015

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A.11 Bank Swallow (*Riparia riparia*)

A.11.1 Listing Status

Federal: None.

State: Threatened.

Recovery Plan: Recovery Plan: Bank Swallow *(Riparia riparia)* (California Department of Fish and Game [DFG] 1992).

A.11.2 Species Description and Life History

The bank swallow (*Riparia riparia*) breeds throughout much of the Northern Hemisphere and migrates to spend the winter months in South America, Africa, and southern Asia. It is the smallest of the North American swallows (approximately 13 centimeters [5.12 inches] long). Bank swallows are distinguished from other swallows by their distinctive, complete brown breast band, contrasted against white underparts and its dark brown upper parts. Sexes are similar and cannot be distinguished based solely on plumage characteristics (DFG 1992).

A.11.2.1 Seasonal Patterns

Bank swallows arrive in California from their wintering grounds in the southern Amazon basin from mid-March to May and reestablish breeding colonies shortly after arrival. During spring migration, the first individuals arrive in California in mid-March, with numbers peaking in May; during fall migration, the first individuals leave in late July, with a few birds remaining until mid-September (Humphrey and Garrison 1987; Garrison 1999; Garrison 2002). After breeding, bank swallows join mixed-species flocks of swallows that congregate at wetlands and other areas with high concentrations of aerial insect prey, until they depart California for their southward migration in August and September.

A.11.2.2 Reproduction

Bank swallows nest in colonies in vertical cliffs, most often in lowland riverbanks, coastal bluffs, open pit mines, and roadcuts (DFG 1992). Following a short courtship, both sexes spend four to five days digging a nest burrow in soft sand/loam strata. Females typically lay four or five eggs, and feed their young at the nest until the young fledge in 18 to 20 days later. Banks swallows are primarily monogamous, and each pair tends one nest. However, extra-pair copulations are frequent which enhances the genetic diversity of a brood and colony (Garrison 1999).

A.11.2.3 Home Range/Territory Size

Bank swallows actively defend nest burrows and the immediate vicinity of individual burrows. They defend the area around an occupied burrow early in the nesting period. Females select burrows and frequently reject burrows excavated by males until a burrow is suitable for nesting. Thus, typically the number of burrows outnumbers the pairs of bank swallows in a given colony (Garrison 1999).

A.11.2.4 Foraging Behavior and Diet

Bank swallows often join mixed-species flocks of swallows while foraging over water, meadows, bogs, and other sites where concentrations of aerial insects can be found. At nesting colonies, they forage mostly within 200 meters (656 feet) of their nesting burrows, but this range can vary depending on the distance to good foraging areas. Analysis of contents of 394 stomachs from throughout Canada and the United States disclosed 33.5 percent ants, bees, and wasps; 26.6 percent flies; 17.9 percent beetles; 10.5 percent mayflies; 8 percent bugs; and a few dragonflies, butterflies, and moths (Garrison 1999, 2002).

A.11.3 Habitat Requirements and Ecology

A.11.3.1 Nesting

Important breeding habitat characteristics include soil moisture, texture, orientation of bank face, bank height, verticality (slope) of the face, and proximity of the colony to foraging areas (DFG 1992). Bank swallow colonies are often found in fine silt and sandy loam soils (DFG 1992) represented as three main types: sea cliffs, or hard consolidated sand; river banks of sand and sandy earth; and actively worked sand and gravel pits (Hickling 1959 as cited in DFG 1992). In California, bank swallows most often nest in steep earthen riverbanks subject to frequent winter erosion events. Nest sites consist of burrows dug into a vertical earthen bank 45 to 90 centimeters (cm) (17.72 to 35.43 inches) deep, 5 cm (1.97 inches) high, and 7.6 cm (2.99 inches) wide (Garrison 1999). Sites with grassland adjacent to vertical banks are considered of highest suitability (Garcia et al. 2008).

Unique combinations of optimal habitat characteristics may dictate the size and success of individual bank swallow colonies. Burrows that remain available from a previous season may be used in subsequent years. Bank swallow nesting colonies range in size from relatively small (10 burrows) to very large (3,000 burrows) (DFG 1992). Suitable burrows for nesting are at least 1 meter (3.3 feet) above ground or water for predator avoidance, and heights of occupied colony banks in California averaged 3.3 meters (10.83 feet) (SD = 1.7, range 1.3 to 7.3, n = 23) (Garrison 2002).

A.11.3.2 Foraging

Bank swallows are aerial insectivores that forage over lakes, ponds, rivers and streams, meadows, fields, pastures, and bogs (Garrison 1999). Grasslands and croplands immediately adjacent to colonies also provide foraging habitat for bank swallows (DFG 1992). Adult birds foraging along the Sacramento River typically forage within 50 to 200 meters (164 to 656 feet) of the colony location (Garrison 1998), and the normal maximum foraging distance can be as great as 8 to 10 kilometers (5.0 to 6.2 miles) (Mead 1979).

A.11.4 Species Distribution and Population Trends

A.11.4.1 Distribution

During the summer months in the western hemisphere, bank swallows range throughout most of Alaska and Canada, southward from eastern Montana to Nevada, and eastward across the United States to Georgia. They are variably distributed throughout California, Texas, and New Mexico. Within California, regular breeding of the Bank Swallow occurs in Siskiyou, Shasta, and Lassen Counties, and along the Sacramento River from Shasta County south to Yolo County (DFG 2000). Other subspecies are also widespread and common in Europe, Asia, and Africa (Garrison 1999). Bank swallows winter primarily in South America, especially in the southern Amazon Basin and Pantanal (Garrison 1999), although a few winter along the Pacific coast of Mexico (Howell and Webb 1995).

A.11.4.2 Population Trends

Bank swallows historically nested throughout the lowlands of California (Grinnell and Miller 1944). The species once bred at coastal sites from Santa Barbara County south to San Diego County. They have now disappeared as a breeding bird from Southern California (Garrett and Dunn 1981). The historical population along the Sacramento River was most likely larger than it is today, but no population data exist from that era (DFG 1992).

The colonial nesting habits of the bank swallow and the short-lived nature of colony sites make it difficult to consistently census the species accurately from point counts on Breeding Bird Surveys (Garrison 1999), so trends reported from that data set are not informative. According to DFG (2000), estimates of breeding pairs in Sacramento River habitats dropped from 13,170 in 1986 to 5,770 in 1997. In 1998, the number of breeding pairs dropped to 4,990 before rebounding in 1999 to 8,210 pairs. Since 2000, numbers have fluctuated between 6,320 and 8,530 pairs (Garcia et al*.* 2008). Population size can vary greatly over relatively short time periods because of the poor durability of nesting sites and weather-influenced mortality on wintering grounds (Garrison 1999).

A.11.4.2.1 Distribution and Population Trends in the Plan Area

In Yolo County, colonies ranging from 10 to 400 burrows were observed along the Sacramento River and Cache Creek in 1987 (California Natural Diversity Database [CNDDB] 2005). Breeding occupancy was estimated as ranging 10 to 70 percent at the various colonies. However, many of the colonies were unoccupied or inactive. During a survey in 2000, four colonies totaling 488 burrows were found along the Sacramento River in Yolo County between Verona and Knight's Landing (R. Schlorff and C. Swolgaard unpublished data). Assuming an occupancy rate of 45 percent, as used by California Department of Fish and Wildlife (DFW) (Wright et al. 2011), this population was estimated at 202 pairs. An active colony persisted along Cache Creek in a gravel quarry until at least 2001 (Yolo Audubon Society 2004).

April 10, 2011, Whisler (pers. comm. 2015) observed bank swallows nest-building in the bank of the cross-channel from the Port of West Sacramento to the Sacramento River. The colony failed when the Sacramento River rose from heavy rains that spring. This was likely the southernmost colony along the Sacramento River, and in the most urban area along the Sacramento River. No colonies have been detected since then (Whisler pers. comm. 2015).

A.11.5 Threats to the Species

In California, the loss of nesting habitat is the most significant threat to bank swallows. Nesting habitat is lost through conversion of natural waterways to flood control channels, stabilization of riverbanks for flood control, and other activities that change the natural flow of rivers and prevent the creation of new nesting habitat. Bank stabilization projects are currently the single greatest threat to the state's largest bank swallow population, which breeds along the Sacramento River from Shasta to Yolo counties (Garrison 1998). These projects have had a significant effect on nesting habitat when banks are sloped to 45 degrees and include large rocks. Colony sites are also destroyed by road building and by increased regulation of water flow from reservoirs that can reduce needed winter bank erosion (to maintain vertical banks) or increase summer flows, which can flood nests and intensify erosion during the breeding season (Humphrey and Garrison 1987; Garrison 1999; Garcia et al. 2008). Destruction of nest sites or burrow collapse due to natural or human-related alteration of banks has been found to be the most significant, direct cause of mortality. Bank swallow young and eggs are the primary victims of this type of mortality (DFG 1992). In addition, gopher snakes (*Pituophis melanolencus*) are a significant predator of eggs and nestlings, and raptors such as peregrine falcons (*Falco peregrinus*) and American kestrels (*F. sparverius*) may take young and adults (DFG 1992).

Other factors that affect swallow populations include fluctuations in the genetic structure of a population; demographic factors such as recruitment rates, sex ratios, and survivorship; climate; and catastrophic events, including flooding, drought, fire, and epidemics (DFG 1992). Bank swallows are generally tolerant of human disturbance in the general vicinity of colonies (Garrison 1999).

A habitat suitability index model was developed to evaluate habitat for breeding colonies within the continental United States (Garrison 1989). The model assumed that a bank suitable for a nesting colony must be at least 5 meters (16.7 feet) long; that suitable foraging habitat occurs within 10 kilometers (6 miles) of the colony; that insect prey are not limited; and that optimal colony locations are in vertical banks, greater than 1 meter (3.3 feet) tall, greater than 25 meters (83 feet) long, and consisting of suitable soft soils (sand, loamy sand, sandy loam, loam, and silt loam) in strata greater than 0.25 meter (0.8 foot) wide. The habitat variables incorporated into the model included soil texture class and width in strata, slope of bank, height of bank, and length of bank.

A significant data gap exists in regard to locations of recently occupied bank swallow colony sites and population sizes in Yolo County, especially along Cache Creek. More information is also needed to assess the effects of pesticides and other contaminants, predation, and local river dynamics and flood control projects on the swallows and their nesting colonies.

Extinction probabilities of bank swallow colonies along the Sacramento River decreased with proximity to the nearest grassland, decreased with colony size, and increased with maximum water discharge (Moffatt et al. 2005). Creation of vertical banks in friable sandy soils and road cuts can directly benefit the bank swallow if large rocks (rip-rap) are not placed on the slopes. Artificial banks and enhanced natural banks were built along Sacramento River to mitigate loss of colony sites from flood control projects (Garrison 1991). The artificial banks provided some initial success in that bank swallows occupied artificial and enhanced sites for a few years following construction. Nestlings at the artificial and enhanced colonies were produced at levels similar to natural sites. However, these colonies were abandoned after three years because maintenance activities such as vegetation removal and bank maintenance were conducted on the sites, thereby rendering them unsuitable as bank swallow habitat (Garrison 1991).

Habitat enhancement is feasible, but to ensure suitable quality of artificial banks, the sites must be maintained. Habitat enhancement is currently considered inappropriate for the long-term maintenance of bank swallows because maintenance, such as excavation with hand tools, is costly to maintain and monitor over time (Garrison 1991; DFG 1992).

A recovery plan written for the bank swallow in California proposed long-term strategies to preserve bank swallow habitat including developing set-back levees and a riverine meander-belt, preserving major portions of remaining habitat, and developing reach-by-reach habitat maintenance strategies based on the results of a population analysis of the Sacramento River population outlined in the recovery plan (DFG 1992).

The population of bank swallows inhabiting the Sacramento River and its major tributaries are the core of the State’s population. These areas, therefore, provide the most important habitat for the long-term maintenance and recovery of bank swallows (DFG 1992). The population analysis in the recovery plan (DFG 1992) indicated that “the risk of low numbers in some years was substantial for the Sacramento River bank swallow population and, under most modeled conditions, was considerably higher than the risk of near local extinction.”

A.11.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.11.3, *Habitat Requirements and Ecology* (Figure A-11).

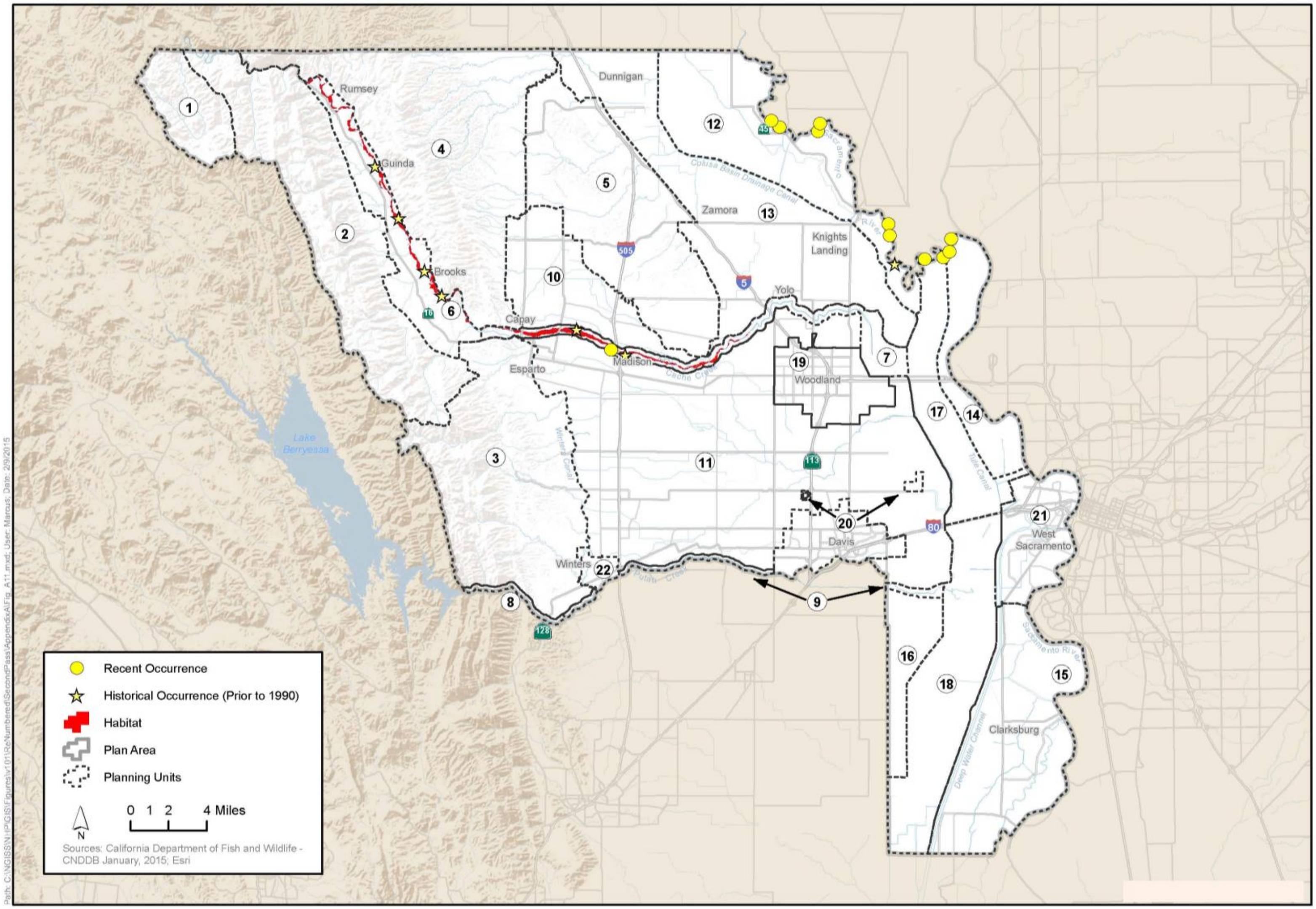
The model parameters include the following.

- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Location where the species has relatively recently (post-January 1, 1990) been documented according to one or more species locality records databases (i.e., California Natural Diversity Database [CNDDB], Ed Whisler, John Sterling, Chris Alford).
- Nesting Habitat: This habitat includes all potentially suitable breeding habitat in stream channels with suitable nesting substrate of vertical and friable river banks that are free of rip-rap. This habitat was modeled by selecting all mapped land cover types as listed below that occur in the Yolo Bypass, Central Valley and Capay Valley ecoregions.
- Limited modeling to the following Planning Units: 6, 7, 12, 14, 17.

A.11.6.1 Breeding – Land Cover Type

- Barren – Gravel and Sand Bars

Figure A-11. Bank Swallow Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

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A.12 Tricolored Blackbird (*Agelaius tricolor*)

A.12.1 Listing Status

Federal: None.

State: Endangered.

Recovery Plan: None; however, a conservation strategy for this species was prepared (Tricolored Blackbird Working Group 2007).

A.12.2 Species Description and Life History

Tricolored blackbirds (*Agelaius tricolor*) form the largest colonies of any North American passerine bird, and these may consist of tens of thousands of breeding pairs (Beedy and Hamilton 1999). Tricolored blackbirds are largely endemic to California and the state is home to more than 95 percent of the global population.

This species closely resembles red-winged blackbird (*Agelaius phoeniceus*), with subtle differences in coloration, bill shape, and overall morphology (Beedy and Hamilton 1999). The adult male is black, with shades of glossy blue, and has a bright red patch on the wing (an epaulet), similar to that of a red-winged blackbird. However, the epaulet of tricolored blackbirds is deeper red with a white lower border, as opposed to an orange-red patch with a yellowish border or no border at all. The adult females are brownish and black, streaked with gray, with small reddish epaulets (rarely visible in the field) and pale gray or whitish chin and throat. Tricolored blackbirds have longer, slightly narrower wingtips and thinner bills than the red-winged blackbirds (Beedy and Hamilton 1999).

A.12.2.1 Seasonal Patterns

Many tricolored blackbirds reside throughout the year in the Central Valley of California (Beedy 2008). However, local populations can move considerable distances, and some are migratory and move from inland breeding locations to wintering habitats in the Sacramento-San Joaquin River Delta and coastal areas. During the breeding season, most birds nest in the San Joaquin Valley and in Sacramento County in their first breeding efforts. They may later move northward into the Sacramento Valley, northeast California, and southern Oregon to nest again (Hamilton 1998; Beedy 2008). Thus, individual tricolored blackbirds may occupy and breed at several sites, or re-nest at the same site, during a given breeding season, depending on environmental conditions and their previous nesting success (Hamilton 1998; Beedy and Hamilton 1999; Meese 2006). In fall, after the nesting season, large roosts form at managed wildlife refuges and other marshes near abundant food supplies such as rice (*Oryza sativa*) and water grass (*Echinochloa crus galli*) (Beedy and Hamilton 1997). During winter, many tricolored blackbirds move out of the Sacramento Valley to the Sacramento-San Joaquin River Delta. Large flocks also winter in the central and southern San Joaquin Valley, and at the dairy farms in coastal areas such as Point Reyes and Monterey County (Beedy and Hamilton 1997). In early March to early April, these flocks move from wintering areas to their breeding colonies in Sacramento County and the San Joaquin Valley (Beedy and Hamilton 1997).

A.12.2.2 Reproduction

Tricolored blackbirds nest colonially, enabling them to synchronize their timing of nest building and egg laying (Beedy and Hamilton 1999). A few breeding colonies documented during fall months (September to November) had more protracted nest-building periods that led to asynchronous egg laying and fledging of young (Orians 1960). Females typically lay three to four eggs and incubate them for 11 to14 days, then both parents feed young until they fledge nine to 14 days after hatching (Beedy and Hamilton 1999).

A.12.2.3 Home Range/Territory Size

As many as 20,000 to 30,000 nests have been recorded in cattail (*Typha* spp.) marshes of 4 hectares or less, with individual nests less than 0.5 meter from each other (Neff 1937; DeHaven et al. 1975). Nest heights range from a few centimeters to about 1.5 meters above water or ground at colony sites in freshwater marshes (Neff 1937) and up to 3 meters in the canopies of willows (*Salix* spp.) and other riparian trees; rarely, they are built on the ground. The species typically selects breeding sites adjacent to open accessible water and places its nests in a protected nesting substrate, often including either flooded or thorny or spiny vegetation. Breeding colonies must have suitable foraging space providing adequate insect prey within a few kilometers (Beedy and Hamilton 1999).

A.12.2.4 Foraging Behavior and Diet

Diets of adult tricolored blackbirds are dependent on geographic location and the availability of local insect foods. Among the most important prey for adults provisioning nestlings include Coleopterans (beetles), Orthopterans (grasshoppers, locusts), Hemipterans (true bugs), other larval insects, and Arachnids (spiders and allies) (Crase and DeHaven 1977; Beedy and Hamilton 1999). The primary diet of a colony depends on the local food availability, and large hatches of dragonflies (Odonata) are especially favorable to this species (Meese pers. comm.); they are also attracted to large outbreaks of grasshoppers (Orians 1961). Adult females require insects to form eggs, and nestlings require insects since they are unable to digest plant materials until they are at least nine days old and ready to leave their nests (Beedy and Hamilton 1999). During the nonbreeding season, tricolored blackbirds often congregate at dairy feedlots to consume grains and other livestock feed, while others forage on insects, grains, and other plant material in grasslands and agricultural fields (Beedy and Hamilton 1999; Skorupa et al. 1980).

A.12.3 Habitat Requirements and Ecology

A.12.3.1 Nesting

Tricolored blackbird colonies require access to water, suitable nesting substrates (including marsh vegetation or thorny or spinous vegetation to protect them from mammalian predators), and foraging habitat with significant populations of insect prey within a few miles (Beedy and Hamilton 1999; Hamilton 2004). Breeding habitat includes diverse wetland and upland and agricultural areas, including those with dense cattails (*Typha* spp.), bulrushes (*Scirpus* spp.), willows (*Salix* spp.), blackberry (*Rubus* spp.), thistles (*Cirsium* and *Centaurea* spp.), and nettles (*Urtica* sp.) (Neff 1937; Hamilton 1998; Beedy and Hamilton 1999). Some of the largest colonies are in silage and grain fields in the San Joaquin Valley, and many are in the vicinity of dairies and feedlots (Hamilton 1998, Beedy and Hamilton 1999).

A.12.3.2 Foraging

Tricolored blackbirds forage in areas that provide abundant insects, including pastures, dry seasonal pools, agricultural fields such as alfalfa and rice, feedlots, and dairies. Tomatoes may occasionally be used as foraging habitat. With the loss of the natural flooding cycle and most native wetland and upland habitats in the Central Valley, breeding tricolored blackbirds now forage primarily in anthropogenic habitats. Tricolored blackbirds have been able to exploit foraging conditions created when shallow flood-irrigation, mowing, or grazing keeps the vegetation at an optimal height (less than 15 centimeters [cm]). Preferred foraging habitats include crops such as rice, alfalfa, safflower, irrigated pastures, and ripening or cut grain fields (e.g., oats wheat, silage) as well as annual grasslands and shrublands (Beedy and Hamilton 1999; Beedy 2008).

In recent years, an increasing percentage and now large majority of adults have foraged on grains provided to livestock as in cattle feedlots and dairies. Tricolored blackbirds also forage in remnant native habitats, including wet and dry vernal pools and other seasonal wetlands, riparian scrub habitats, and open marsh borders. Vineyards, orchards, and row crops (sugar beets, corn, peas, beets, onions, etc.) do not provide suitable nesting substrates or foraging habitats for tricolored blackbirds (Beedy and Hamilton 1999). Both adults feed the nestlings; adults feeding young typically forage within 5 kilometers (km) (3.11 miles) of the colony, but can range up to 13 km (8 miles) from the colony (Beedy and Hamilton 1999).

Some small breeding colonies may occur at private and public lakes, reservoirs, and parks provided that they are near suitable foraging habitats. Many of these colonies are surrounded by shopping centers, subdivisions, and other urban development; adults from such colonies forage in undeveloped uplands nearby.

A.12.4 Species Distribution and Population Trends

A.12.4.1 Distribution

Tricolored blackbirds are endemic to the western edge of North America; however, about 95 percent of the global population resides in California where breeding has occurred in 46 counties (Beedy and Hamilton 1999). Except for a few peripheral sites, the geographic distribution has not declined; breeding colonies in northeastern California, southern Oregon, Washington, western Nevada, and central and western Baja California have been documented (Beedy and Hamilton 1999). While the overall geographic breeding distribution of the species may not have changed since historical times, there are now large gaps in their former range encompassing entire counties (e.g., Kings, San Joaquin, Riverside, San Bernardino counties).

A.12.4.2 Population Trends

The first systematic surveys of the tricolored blackbird’s population status and distribution were conducted by Neff (1937). During a five-year interval, he found 252 breeding colonies in 26 California counties; the largest colonies were in rice-growing areas of the Sacramento Valley. Neff observed as many as 736,500 adults per year (1937) in eight Central Valley counties. The largest colony he observed, in Glenn County, covered almost 24 hectares (59 acres), and contained more than 200,000 nests (about 300,000 adults). Several other colonies in Sacramento and Butte counties contained more than 100,000 nests (about 150,000 adults).

DeHaven et al. (1975) estimated that the overall population size in the Sacramento and northern San Joaquin Valleys had declined by more than 50 percent since the mid-1930s. DeHaven et al. (1975) performed surveys in the areas surveyed by Neff (1937) and observed significant population declines and reductions of suitable habitat since Neff's surveys. Orians (1961) observed colonies of up to 100,000 nests in Colusa, Yolo, and Yuba counties but did not attempt to survey the entire range of the species. Recent statewide censuses have shown dramatic declines in tricolored blackbird numbers in the Central Valley (Beedy and Hamilton 1997; Hamilton et al. 1999; Hamilton 2000; Green and Edson 2004; Cook and Toft 2005). Statewide totals of adults in four late-April surveys covering all recently known colony sites were 369,359 in 1994, 237,928 in 1997, 104,786 in 1999, and 162,508 in 2000 (Hamilton 2000). In April 2004, statewide surveys focused on only those colonies that had supported greater than 2000 adults in at least one previous year. Of 184 sites surveyed, only 33 supported active colonies at the time of the survey. Of the 33 colonies, 13 held greater than 2000 adults each, collectively representing greater than 96 percent of the census total (Green and Edson 2004). A statewide survey performed on April 25 to 27, 2008 found a total of 394,858 adults at 155 sites in 32 counties (Kelsey 2008). The most recent statewide survey for tricolored blackbirds was conducted in 2014, at which time the number of tricolors dropped to 145,135 birds (Meese 2014).

A.12.4.3 Distribution and Population Trends in the Plan Area

In Yolo County, tricolored blackbirds historically bred primarily in marshes with emergent vegetation. The species forages in grasslands, wetlands, and agricultural fields from March through July, but are irregular visitors during the remainder of the year (Yolo Audubon Society Checklist Committee 2004). Recent surveys revealed very few nesting colonies in Yolo County (Meese pers. comm.). Fourteen colonies were documented in the county from 1994 to 2004, with populations estimated from 15 to 1,500 adults. Surveys in 2007 revealed a highly successful colony of more than 30,000 breeding adults in milk thistle on the Conaway Ranch in the Yolo Bypass. This was one of only three documented colonies statewide that were large and successful, and this colony was estimated to have produced about 30,000 young (Meese 2007). Other recent colony sites in the county included: "Bill's Grasslands," a newly-discovered colony located within a patch of Himalayan blackberry approximately one km south of the intersection of County Roads 92B and 15B, that was active in 2006 and again in 2007. This colony was active again in 2012 in a slightly different location off Road 92B. Another colony in milk thistle on County Road 88B, about two km north of State Route 16 that was active in 2005 and 2007, but not in 2006. Four small colonies were also found in the Yolo Bypass in 2005 that have not been occupied since. A historical colony at the Sunsweet Drying facility, just south of County Road 27 and about 1 km west of I-505, has not been active in the past three years (Meese pers. comm.). A total of 1,900 adults were observed at two colonies in the Yolo Bypass during the 2008 statewide survey (Kelsey 2008).

A.12.5 Threats to the Species

A.12.5.1 Habitat Loss and Degradation

The greatest threats to this species are the direct loss and degradation of habitat from human activities (Beedy and Hamilton 1999). Most native habitats that once supported nesting and foraging tricolored blackbirds in the Central Valley have been replaced by urbanization and agricultural croplands unsuited to their needs. In Sacramento County, an historical breeding center of this species, the conversion of grassland and pastures to vineyards expanded from 3,050 hectares in 1996 to 5,330 hectares in 1998 (DeHaven 2000) to 6,762 hectares in 2003 (California Agriculture Statistics Services).[3] Conversions of pastures and grasslands to vineyards in Sacramento County and elsewhere in the species' range in the Central Valley have resulted in the recent loss of several large colonies and the elimination of extensive areas of suitable foraging habitat for this species (Cook 1996; DeHaven 2000; Hamilton 2004).

A.12.5.2 Direct Mortality During Crop Harvest

Entire colonies (up to tens of thousands of nests) in cereal crops and silage are often destroyed by harvesting and plowing of agricultural lands (Beedy and Hamilton 1999; Hamilton 2004; Cook and Toft 2005). While adult birds can fly away, eggs and fledglings cannot. The concentration of a high proportion of the known population in a few breeding colonies increases the risk of major reproductive failures, especially in vulnerable habitats such as active agricultural fields.

A.12.5.3 Predation

Historical accounts documented the destruction of nesting colonies by a diversity of avian, mammalian, and reptilian predators (Beedy and Hamilton 1999). Recently, especially in permanent freshwater marshes of the Central Valley, entire colonies have been lost to black-crowned night-herons (*Nycticorax nycticorax*) and common ravens (*Corvus corax*). Recently, cattle egrets (*Bubulcus ibis*) have been observed preying on tricolored blackbird nests, and at one colony in Tulare County more than 125 egrets were present throughout the breeding season (Meese 2007). Some large colonies (up to 100,000 adults) may lose more than 50 percent of nests to coyotes (*Canis latrans*), especially in silage fields, but also in freshwater marshes when water is withdrawn (Hamilton et al. 1995). Thus, water management by humans often has the effect of increasing predator access to active colonies.

A.12.5.4 Poisoning and Contamination

Various poisons and contaminants have caused mass mortality of tricolored blackbirds. McCabe (1932) described the strychnine poisoning of 30,000 breeding adults as part of an agricultural experiment. Neff (1942) considered poisoning to regulate numbers of blackbirds preying upon crops (especially rice) to be a major source of mortality. This practice continued until the 1960s, and thousands of tricolored blackbirds and other blackbirds were exterminated to control damage to rice crops in the Central Valley. Beedy and Hayworth (1992) observed a complete nesting failure of a large colony (about 47,000 breeding adults) at Kesterson Reservoir, Merced County, and selenium toxicosis was diagnosed as the primary cause of death. At a colony in Kern County, all eggs sprayed by mosquito abatement oil failed to hatch (Beedy and Hamilton 1999). Hosea (1986) attributed the loss of at least two colonies to aerial herbicide applications.

A.12.5.5 Other Conservation Issues

Important information gaps in the ecology of the species include the effects of land use changes on the reproductive success of colonies and on the distribution of wintering birds, the relationship of

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[3] http://www.nass.usda.gov/ca/.

invertebrate prey abundance and brood size, winter distribution, diet, and survival rates, and measures of suitable foraging habitat (Beedy and Hamilton 1999; Meese 2007).

Tricolored blackbirds have been the focus of recent management concern due to population decline, very limited global range, and vulnerability of large breeding colonies to habitat losses, predation, and human-induced impacts. Recommendations for the species conservation (Beedy and Hamilton 1999; Hamilton 2004) include frequent monitoring of breeding and wintering population sizes, colony locations, and reproductive success; protection of colony locations and foraging habitats; protection of colonies on farmland by avoiding harvesting/tilling until young have fledged; providing adequate protection in Habitat Conservation Plans; focusing on dairy-dependence for breeding and wintering populations; developing or restoring breeding habitat near reservoirs, rice fields, alfalfa fields and other optimal foraging habitats; and managing major predators in or near breeding colonies, including common ravens, black-crowned night-herons, cattle egrets, and coyotes when feasible.

A.12.6 Species Habitat Model and Location Data

The habitat model for this species was based on the distribution of land cover types that are known to support its habitat as described above in Section A.31.3, *Habitat Requirements and Ecology* (Figure A-31). The model parameters include the following.

- Known Recent Colonies in Yolo NCCP/HCP Species Locality Database: Location where colonies have relatively recently (post-January 1, 2000) been documented according to one or more species locality records databases (i.e., California Natural Diversity Database [CNDDB], John Kemper, University of California, Davis (UC Davis) Museum of Wildlife and Fish Biology, BIOS, Bob Meese, Avian Knowledge Network).
- Known Recent Sightings in Yolo NCCP/HCP Species Locality Database: Other location where the species has relatively recently (post-January 1, 1990) been documented, but not identified as a colony site, according to one or more species locality records databases (i.e., CNDDB, John Kemper, UC Davis Museum of Wildlife and Fish Biology, BIOS, Bob Meese, Avian Knowledge Network).
- Nesting Habitat: This habitat includes all potentially suitable breeding habitat in natural habitat communities. This habitat was modeled by selecting all mapped vegetation types as listed below that occur in the Yolo Bypass, Central Valley, Capay Valley, and Dunnigan Hills ecoregions.
- Foraging Habitat: This habitat includes all potentially suitable foraging habitat. This habitat was modeled by selecting all mapped vegetation types listed below that occur within 13 km (8 miles) of nesting habitat.

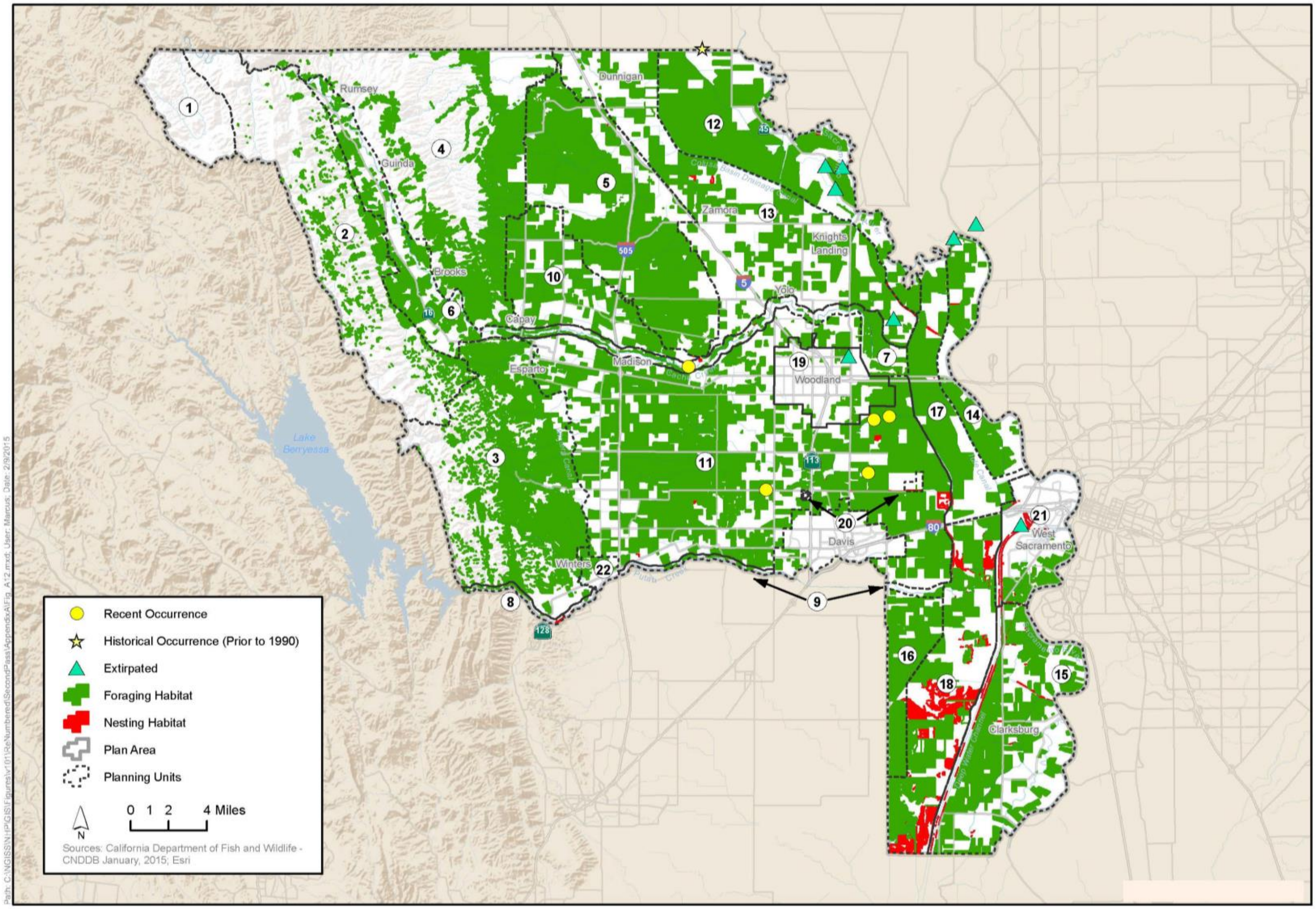
A.12.6.1 Nesting Habitat – Vegetation Types

- Alkali Bulrush – Bulrush Brackish Marsh Not Formally Defined (NFD) Super Alliance
- Bullrush – Cattail Wetland Alliance
- Bulrush – Cattail Fresh Water Marsh NFD Super Alliance
- Blackberry NFD Super Alliance
- Undifferentiated Riparian Bramble and Other

A.12.6.2 Foraging Habitat – Vegetation Types

- All Annual Grassland
- All Pasture
- Safflower and Sorghum
- Grain and Hay Crops
- Rice
- Undetermined Alliance – Managed
- Livestock Feedlots
- Poultry Farms

Figure A-12. Tricolored Blackbird Modeled Habitat and Occurrences



Sources: California Department of Fish and Wildlife - CNDDB January, 2015; Esri

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