Chapter IV: Species Accounts

CHAPTER IV: SPECIES ACCOUNTS

Introduction

The following section describes the status, distribution, population status/trends, natural life history, and population threats associated with the plant and wildlife species listed on the Phase I NCCP/HCP Working List of Covered Species (Table 4). Species identified in the following discussion were chosen for coverage based on their current state or federal listing status, their potential to be listed in the foreseeable future, or their local importance to Phase I ecosystems.

Other special-status species that are not listed on the Working List of Covered Species also may occur within the Phase I Planning Area (see Attachment 3, Appendix III, and Appendix VI). These species were not identified on the Working List of Covered Species due to their low likelihood of becoming state/federally listed in the foreseeable future and/or the ability to manage the population through the adaptive management of natural communities per measures outlined in the NCCP. During the NCCP/HCP planning process species may be added to or removed from the Working List of Covered Species.

Legenere (*Legenere limosa*)

Status

FEDERAL: None.

STATE: None.

OTHER: California Native Plant Society List 1B.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

Legenere is endemic to northern California in the Coast and Cascade Ranges and the Central Valley. Current information indicates 52 extant occurrences, seven extirpated occurrences (including the type locality), and one possibly extirpated occurrence (Jones & Stokes 2002; California Natural Diversity Database 2003; Preston pers. comm.). A majority of the known extant occurrences are concentrated in Solano and Sacramento Counties, with the remainder scattered in Lake, Napa, Placer, San Joaquin, San Mateo, Shasta, Sonoma, Tehama, and Yuba counties (Holland 1983; Platenkamp 1998; California Natural Diversity Database 2003). Legenere is considered extirpated from Stanislaus County (California Natural Diversity Database 2003).

Placer County Phase I Planning Area

Historical

Very little information exists on the historical distribution of legenere within the Phase I Planning Area prior to urbanization. Only two occurrences have been recorded in the CNDDB, and one of these is reported to be extirpated (California Natural Diversity Database 2003).

Current

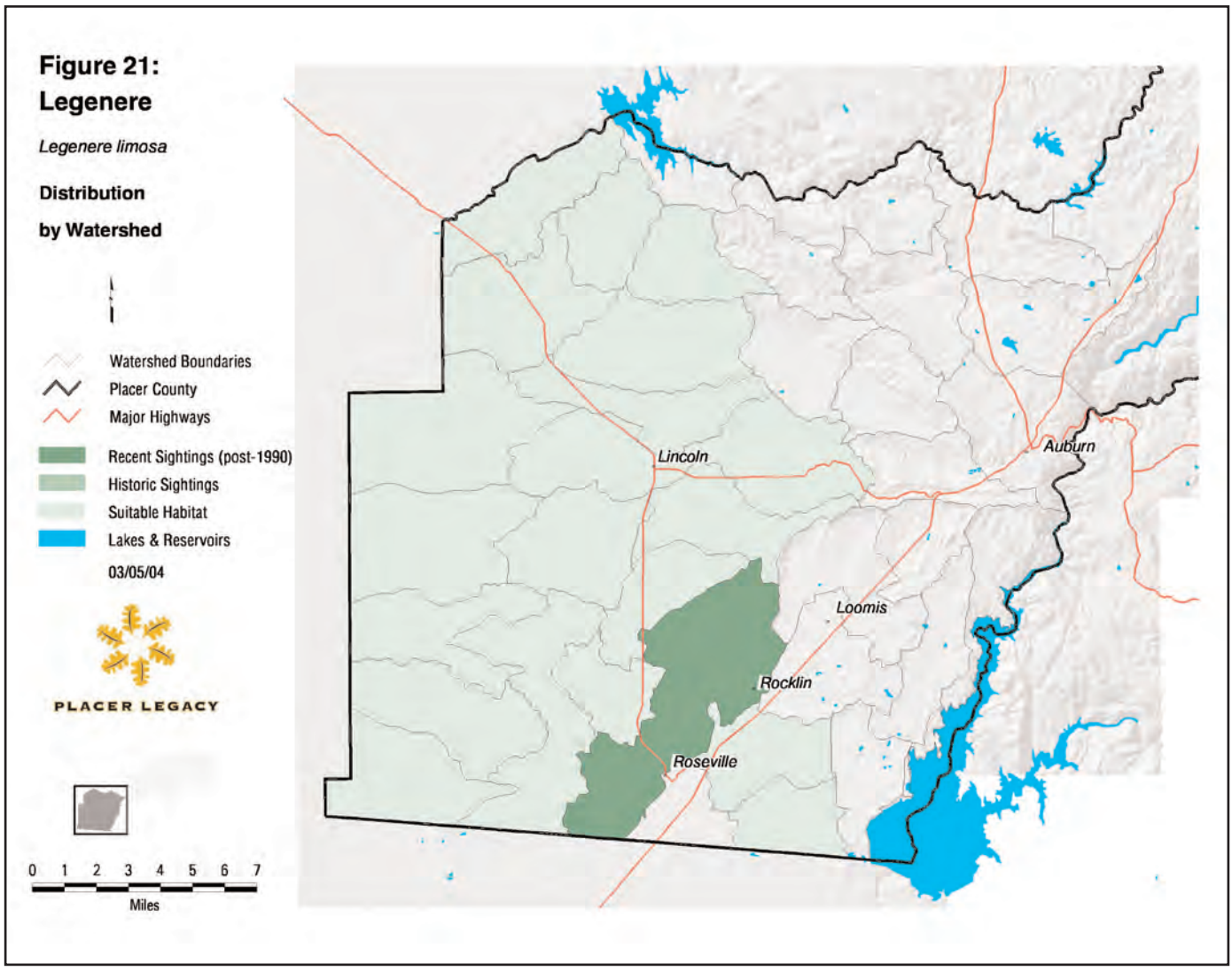
Legenere is currently known from only two extant occurrences in the Phase I Planning Area (Figure 21). One of the occurrences is located north of Pleasant Grove Creek, south of Placer Boulevard, and east of Highway 65. Surveys conducted by Jones & Stokes in 2002 located a second occurrence at the Orchard Creek Conservation Bank approximately 3 miles southwest of Lincoln, California (Jones & Stokes 2002).

Population Status and Trends

California

Occurrences in California are located on private property as well as on land owned by the Sacramento Municipal Utility District; Sacramento County Parks and Recreation District (South Florin County Park); Sacramento County (the Sacramento Regional Wastewater Treatment Plant Bufferlands and the former Mather Field Air Base); Solano County Farmlands and Open Space Foundation (Jepson Prairie); the California Department of Fish and Game (the Calhoun Cut Ecological Reserve in Solano County and the Dales Lake Ecological Reserve in Tehama County); and the Department of Defense (Beale Air Force Base) (California Natural Diversity Database 2003). The current status and trends of legenere populations in California are unknown; however, a number of the known occurrences have been extirpated (California Natural Diversity Database 2003). Legenere populations, like those of many vernal pool species, fluctuate in abundance from year to year depending on the

Legenere Photo by John Game



amount of rainfall. Estimates of some populations have fluctuated from no plants observed in consecutive dry years to thousands of plants in a year of normal or high rainfall, indicating that populations persist by means a dormant seed bank. (Holland 1983; California Natural Diversity Database 2003.)

Placer County Phase I Planning Area

The two known occurrences in the Phase I Planning Area are located on private lands. One of these is in the Orchard Creek Conservation Bank and is protected in perpetuity by a conservation easement. The second occurrence is located within a developing area and has not been observed since 1984. Although the population is presumed to be extant, the CNDDB indicates that a portion of the vernal pools in this area have been destroyed; consequently, the current status of this population is unknown (California Natural Diversity Database 2003).

Natural History

Life History

Legenere is an inconspicuous glabrous annual plant that grows to approximately 10–15 centimeters (4–6 inches) tall, but that can attain heights of up to 30 centimeters (12 inches) (Morin 1993). Legenere produces early deciduous submerged leaves that are linear and 1.3–3.0 centimeters (0.5–1.2 inches) long. Emergent stems produce elliptical, leaf-like bracts. The plant has erect lateral branches that are stiff and sometimes fleshy. The corolla, when present, is bilaterally symmetrical with five lobes and is greenish-white or yellowish in color. Cylindrical capsules are borne on slender, elongate pedicels and contain up to 20 smooth brown seeds. The flowering period for legenere is generally from April through June depending on the depth of the vernal pool or the duration of ponding (California Native Plant Society 2001). Legenere can occur within matted vegetation at the bottom of drying vernal pools. It can be

difficult to locate in the field because of its relatively inconspicuous flowers and sprawling habit within matted vegetation composed of other, superficially similar species.

Habitat Requirements

Legenere is found in vernal pools and swales, seasonal marshes, artificial ponds, floodplains of intermittent streams, and other seasonally inundated habitats (Holland 1983; Morin 1993; California Natural Diversity Database 2003). Wetlands that support legenere are typically inundated for long periods and range in size from slightly more than 3.7 square meters (40 square feet) to 40 hectares (100 acres) (Holland 1983; California Natural Diversity Database 2003). In larger habitats, legenere grows only in areas inundated to less than 8 inches (Holland 1983). This species occurs in northern basalt flow, northern claypan, northern hardpan, northern volcanic ashflow, and northern volcanic mudflow vernal pool types and could occur on any of the geomorphic types within the Phase I Planning Area boundaries (Sawyer and Keeler-Wolf 1995). Soils are generally shallow, acidic clays over a hardpan or volcanic substrate (Holland 1983). Surrounding plant communities are typically grasslands. At one recorded occurrence in Solano County, legenere was found in a vernal pool as well as the adjacent grassland. Elevations of occurrences range from 3 meters (10 feet) in Solano County to 884 meters (2,900 feet) in Lake County. The latter occurrence (and another at 732 meters [2,400 feet]) were recorded in the 1950s; subsequent surveys in the 1980s found the habitat but not the legenere. (California Natural Diversity Database 2003.)

Reproduction and Dispersal

Legenere begins growth underwater and flowers in April or early May. The seeds germinate between late February and April while water is still present, with flowering completed by the time the vernal pool is dry in June. Field surveys indicate that the optimal identification period for this species is limited to a 6- to 8-week period. Each plant produces six to ten capsules, which produce several hundred seeds. The timing and mechanisms of seed dispersal are unknown, but dispersal agents could include gravity, water, and possibly waterfowl. Intact capsules may hold seed, if undisturbed, and float to other hydrologically connected areas in the following winter. (Holland 1983.) The chromosome number has not been determined for legenere.

Ecological Relationships

Legenere often has apetalous flowers (i.e., flowers without petals) suggesting that legenere may self pollinate (Holland 1983); however, pollination studies have not been completed. Very little information is known regarding the intra- and interspecific interactions among vernal pool plants such as legenere. It has been found in both natural and artificial vernal pools (California Natural Diversity Database 2003). Common associates found with legenere include smooth goldfields (*Lasthenia glaberrima*), creeping spikerush (*Eleocharis macrostachya*), three-sepaled buttercup (*Ranunculus bonariensis* var. *trisepalus*), and hairgrass (*Deschampsia danthonioides*). It can also co-occur with several other rare plants, including Boggs Lake hedge-hyssop (*Gratiola heterosepala*), dwarf downingia (*Downingia pusilla*), Sebastopol meadowfoam (*Limnanthes vinculans*), many-flowered navarretia (*Navarretia leucocephala* ssp. *plieantha*), Sacramento orcutt grass (*Orcuttia viscida*), slender orcutt grass (*Orcuttia tenuis*), and Sanford's arrowhead (*Sagittaria sanfordii*) (California Natural Diversity Database 2003). In addition to other rare plants, legenere is also thought to co-occur with rare invertebrate species including vernal pool fairy shrimp (*Branchinecta lynchi*) and vernal pool tadpole shrimp (*Lepidurus packardi*), although this has not been well documented.

Population Threats

The number of known extant populations of legenere has increased in the last 20 years, but the quality and quantity of vernal pool habitat have declined over the same period. Habitat of legenere populations has been lost or degraded through agricultural and urban development, overgrazing, agricultural practices, changes in hydrology, recreational use, trash dumping, and competition from nonnative plants (Holland 1983; California Department of Fish and Game 1992a, 1998; California Native Plant Society 2001; California Natural Diversity Database 2003). Only two populations of legenere are known from the Phase I Planning Area, and both the quality and quantity of available habitat there are in decline. Land uses adjacent to occupied vernal pool habitat can affect vernal pool inundation periods and water quality, introduce exotic species, encourage recreational use, and encourage trash dumping. The full effects of these factors are unknown.

Due to the unique hydrology of vernal pools (i.e., legenere habitat), they typically support species adapted to a specific hydrologic regime, and are therefore somewhat resistant to the effects of exotic species that cannot tolerate prolonged periods of inundation. Several species, including mannagrass (*Glyceria*) and common frog-fruit, are reported to compete with legenere in its habitat (Holland 1983; California Natural Diversity Database 2003).

Due to the short survey window for this species, surveys timed correctly over the identification window (approximately 8 weeks as described above) are essential to obtain an adequate inventory. Inadequate or incorrectly timed surveys leading to the loss of undocumented or undiscovered populations is a threat to legenere.

Dwarf Downingia (*Downingia pusilla*)

Status

FEDERAL: None.

STATE: None.

OTHER: California Native Plant Society List 2.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

Endemic to California, dwarf downingia is found in the inner Coast Ranges, southern Sacramento Valley, and northern and central San Joaquin Valley (California Native Plant Society 2001). Current information indicates 104 extant occurrences of dwarf downingia, six extirpated occurrences, and two possibly extirpated occurrences (California Natural Diversity Database 2003; Jones & Stokes 2002; Vollmar 2002).

Placer County Phase I Planning Area

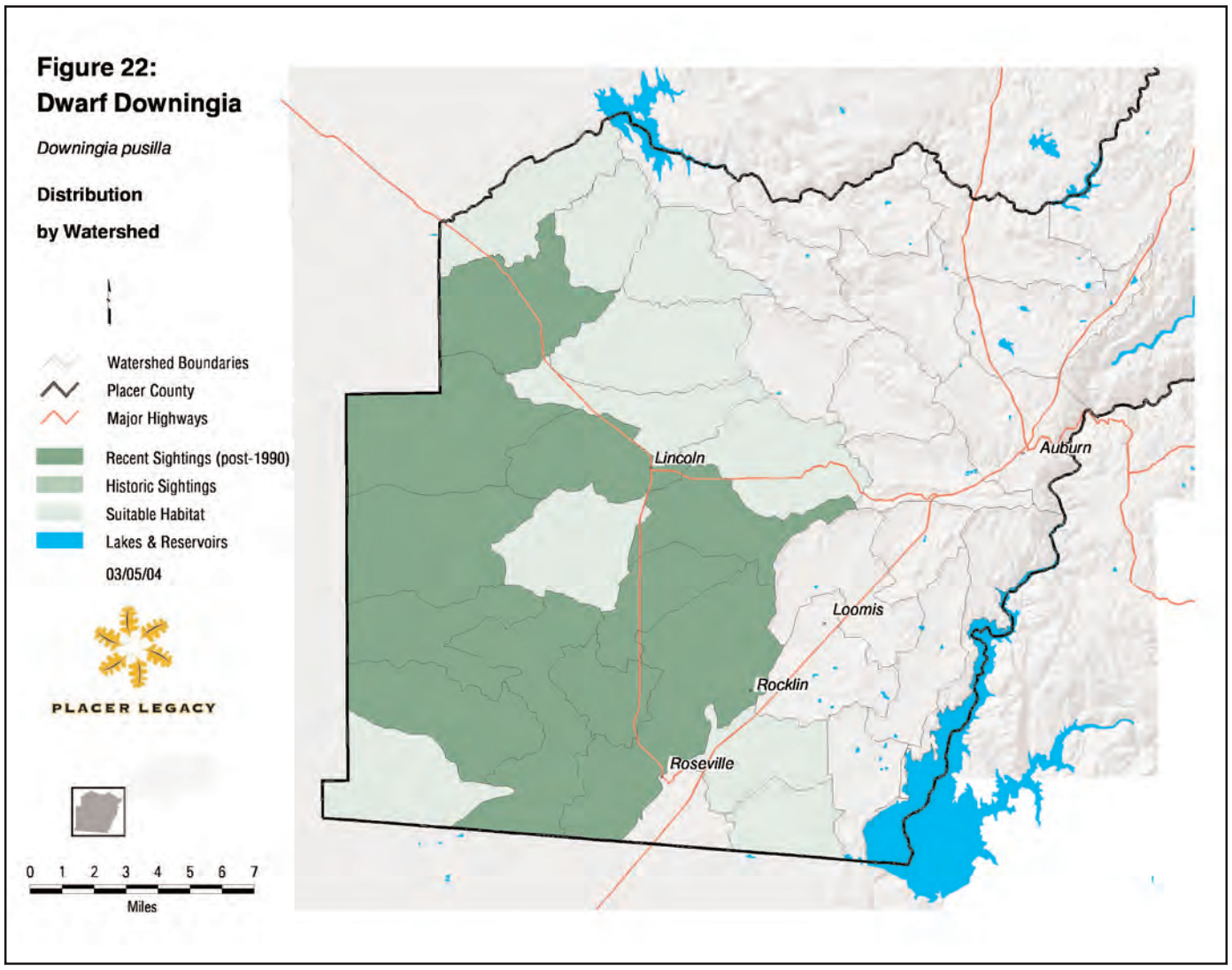
Historical

Very little information exists on the historical distribution of dwarf downingia prior to urbanization within the Phase I Planning Area.

Current

The CNDDB (2003) lists 13 extant occurrences of dwarf downingia in the Phase I Planning Area. Most of these occurrences are located just west of the city of Roseville. Surveys conducted by Jones & Stokes in 2002

Dwarf Downingia Photo by Dean Taylor (Jepson Herbarium)



located three additional occurrences: two on private lands near Roseville and one on a private conservation bank owned by Wildlands, Inc. (Jones & Stokes 2002; Figure 22). These three occurrences had not been recorded in the CNDDB as of preparation of this account. No other occurrences are known from the Phase I Planning Area.

Population Status and Trends

California

Occurrences in California are located on private property as well as on land owned by the Department of Defense (Beale Air Force Base); Sacramento County Parks and Recreation District; Solano County Farmlands and Open Space Foundation (Jepson Prairie); Sonoma County (Sonoma Valley Regional Park); the California Department of Fish and Game (Calhoun Cut Ecological Reserve and Thomes Creek Ecological Reserve); the Bureau of Land Management; the California Department of Transportation; and the Cities of Vallejo, Napa, and Red Bluff. The current status and trends of many of these populations are unknown, and many of them have not been observed for at least 15–20 years. Dwarf downingia populations, like those of many vernal pool species, fluctuate in abundance from year to year depending on the amount and timing of rainfall. Variation in the number of plants present has been observed at four locations, although it is thought that dwarf downingia is less likely to fluctuate to the same degree as vernal pool species that require long periods of inundation to stimulate germination. (California Natural Diversity Database 2003.)

Placer County Phase I Planning Area

There are currently 16 extant occurrences of dwarf downingia in the Phase I Planning Area (Jones & Stokes 2002; California Natural Diversity Database 2003). One of these occurrences is located on land owned by the

Department of Defense at the Lincoln Communication Annex. Another is located at a private conservation bank owned by Wildlands, Inc. The remaining occurrences are located on private land and currently receive no protection.

Natural History

Life History

Dwarf downingia is a small herbaceous annual plant that grows to approximately 2–15 centimeters (0.8–5.9 inches) tall. It is slightly succulent and may have several leaves when flowering. The flowers are 2–4 millimeters (0.08–0.16 inch) across and vary from white to blue. It is distinguished from other downingia species by its small corolla size and short, untwisted ovary. The plants flower from March through May (California Native Plant Society 2001).

Habitat Requirements

Dwarf downingia is primarily associated with northern claypan vernal pools in the central Sacramento Valley, northern hardpan vernal pools in the Sierra Nevada foothills, and vernal pools of the interior Coast Range valleys in Napa and Sonoma Counties (California Natural Diversity Database 2003). It occurs in vernal pools and adjacent vernal swales and in artificial features within the vernal pool landscape, such as stock ponds, roadside ditches, gravel pits, tire ruts, and scraped depressions (Witham 1991; Ayers 1993; California Natural Diversity Database 2003). The vernal pools in which this species occurs have been described as having a short hydroperiod (i.e., "flashy" vernal pools), although it also occurs at the margins of wetlands with longer periods of inundation, such as sloughs and seasonal marsh. At Beale Air Force Base in Yuba County, dwarf downingia occurs in vernal pools of average area and depth. The mean area of occupied vernal pools was 299 square meters (m^2) ($\pm$ 89 m^2) [3,218 square feet (ft^2) ($\pm$ 958 ft^2)], and the mean depth of the occupied pools was 20.8 cm ($\pm$ 3.7 cm) [8.2 inches ($\pm$ 1.5 inches)]. (Platenkamp 1998.)

Reproduction and Dispersal

Seed production in dwarf downingia occurs in late May or early June. Due to the small size of this plant and its seeds, dispersal likely occurs within the vernal pool occupied by the parent plant.

Ecological Relationships

While most downingias in California are primarily pollinated by small *Panurginus* bees and some flies, the flowers of dwarf downingia are self-pollinated and bloom between March and May when pools are dry (Thorp 1976, 1990; Witham undated). Dwarf downingia is often found in areas of low competition from other species, such as sparsely vegetated portions of wetlands and compacted soils along trails created by grazing animals. Associated species include various vernal pool plants, including other downingias (*Downingia bicornuta* and *D. ornatissima*), popcornflowers (*Plagiobothrys* spp.), goldfields (*Lasthenia fremontii* and *L. glaberrima*), hedge-hyssop (*Gratiola ebracteata*), and woolly-marbles (*Psilocarphus brevissimus*) (California Natural Diversity Database 2003). In Placer County, this species co-occurs with another rare plant, legenere (*Legenere limosa*), at one known occurrence (Jones & Stokes 2002).

Population Threats

The number of known populations of dwarf downingia has increased in the last 20 years, but the quality and quantity of vernal pool habitat have declined over the same period. Besides being threatened by a general loss of vernal pool habitat, the species is threatened by overgrazing, agricultural conversion, competition from nonnative species, and off-road vehicles (California Native Plant Society 2001; California Natural Diversity Database 2003).

The unique hydrology of vernal pools (dwarf downingia's habitat) typically supports species adapted to a specific hydrologic regime, and the habitat is therefore somewhat resistant to the invasion of exotic species that cannot tolerate prolonged periods of inundation. It has been suggested, however, that elimination of grazing may allow other seasonal wetland plants to dominate the habitat, thereby making the habitat unsuitable for dwarf downingia, because it is thought to require areas of low competition (California Natural Diversity Database 2003).

Bogg's Lake Hedge-Hyssop (*Gratiola heterosepala*)

Status

FEDERAL: None.

STATE: Endangered (November 1978).

OTHER: California Native Plant Society List 1B.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

California

Bogg's Lake hedge-hyssop was first collected in Lake County, California, in 1923 (Mason and Bacigalupi 1954). Until the 1980s, it was known from only a few sites in Lake, Madera, and Sacramento Counties. During the 1980s, an additional 20 occurrences were recorded in Shasta, Fresno, Placer, Sacramento, Lake, and Modoc Counties, and one occurrence was recorded in Oregon. Many more records were added during the 1990s, and the CNDDB now lists 85 extant occurrences of this species, including one that is possibly extirpated (California Natural Diversity Database 2003). There are two primary concentrations of the species: one on the Modoc Plateau and one in the eastern Sacramento Valley (California Natural Diversity Database 2003).

The distribution of Bogg's Lake hedge-hyssop populations is patchy throughout its range, even in areas of suitable habitat. Uneven distribution and abundance may be due to artificial or natural factors, including historic land management practices (e.g., disking or land leveling) and site characteristics (i.e., species range, soil types, and landforms).

Placer County Phase I Planning Area

Historical

Bogg's Lake hedge-hyssop was not recorded in Placer County until the mid-1980s, when three populations were located.

Current

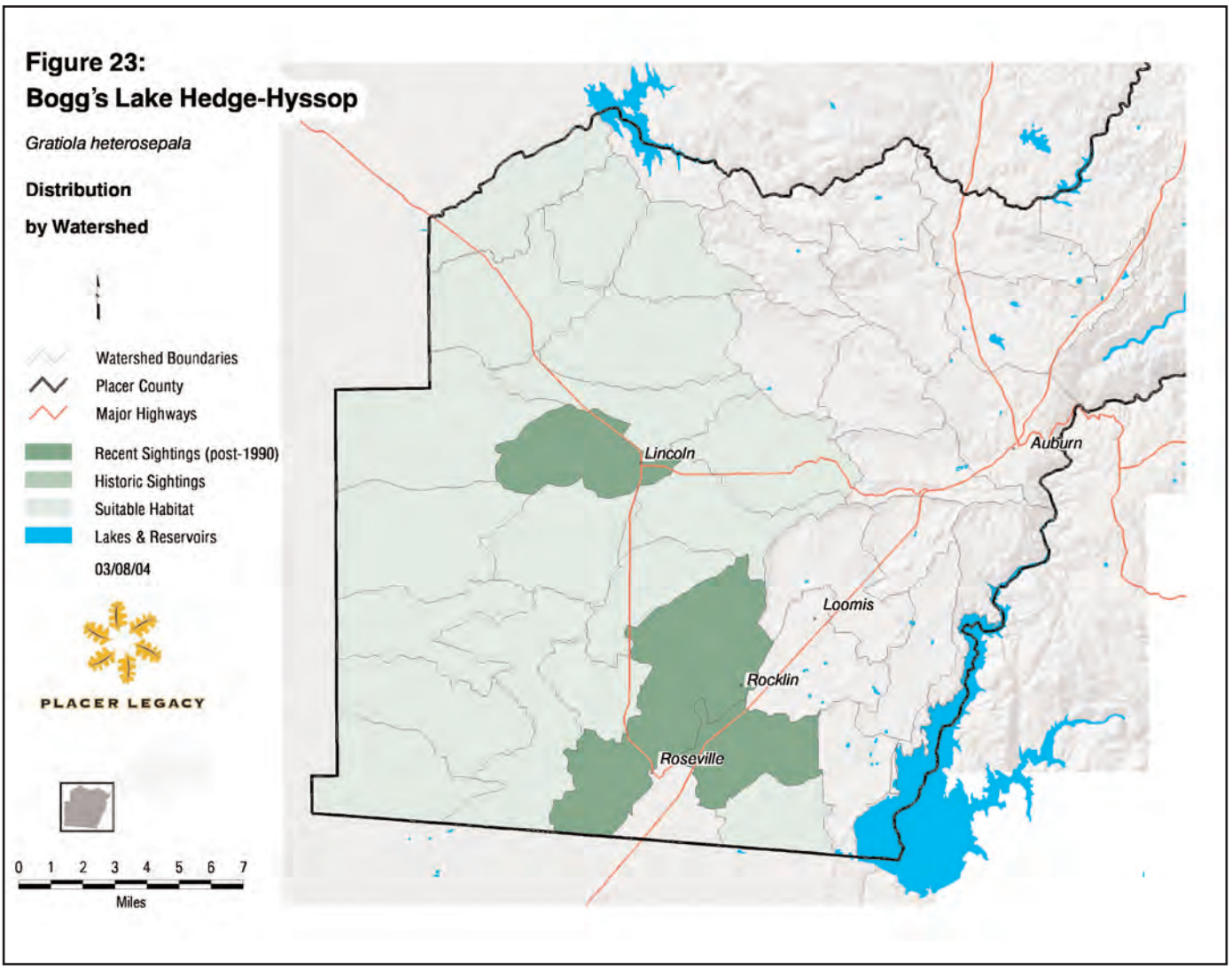
Bogg's Lake hedge-hyssop is currently known from only three occurrences in the Phase I Planning Area. Two of these occurrences are located between Rocklin and Roseville; the third is located just north of Lincoln (Figure 23).

Population Status and Trends

California

Populations of Bogg's Lake hedge-hyssop, like those of many vernal pool species, fluctuate in abundance from year to year depending on the amount of rainfall. When

Bogg's Lake Hedge-Hyssop Photo by Rob Preston



a vernal pool containing Bogg's Lake hedge-hyssop seed does not fill sufficiently, the seeds may not germinate. Estimates of some populations have fluctuated from no plants in a dry year to thousands in a wet year. The most abundant populations recorded are at Jepson Prairie in Solano County; Jepson Prairie supported an estimated million plants in 1993 and 1995, both of which were wet years (California Natural Diversity Database 2003). Population size at the type locality at Bogg's Lake has varied from 1,000 plants in 1981, to none from 1989 through 1997, to five in 1997 (California Natural Diversity Database 2003).

Placer County Phase I Planning Area

Two of the three populations in Placer County were reported to be threatened by proposed urban development in 1987 and 1989. The current status of these populations is unknown; however, they are presumed to be extant. The third population was observed in 1986 on private land and has not been observed since. The current status of this population is also unknown; however, it is also presumed to be extant. (California Natural Diversity Database 2003.)

Natural History

Life History

Bogg's Lake hedge-hyssop is a small, semiaquatic, herbaceous annual that grows to approximately 2–10 centimeters (0.8–3.9 inches) tall (Wetherwax 1993). Each plant generally produces only one or two flowers. The tubular bilateral corolla has two yellow upper lobes and three white lower lobes. The flowers appear yellow from a distance. The blooming period for Bogg's Lake hedge-hyssop is between April and June, while the vernal pools are still inundated with less than 5 centimeters (2 inches) of water (California Native Plant Society 2001).

Habitat Requirements

Bogg's Lake hedge-hyssop occurs in vernal pools primarily on saturated clay (adobe) soils and, at the Bogg's Lake Preserve, on shallow lake margins. Less frequently, this species has been found on loam and loamy sand soils. In smaller vernal pools, it inhabits barren, muddy areas on extremely shallow soils (California Department of Fish and Game 1996, 1998). Some northern California occurrences are on slightly acidic soils (Corbin et al. 1994). The species has also been found in deep vernal pools and has occurred as a volunteer species in created vernal pools in Sacramento County and in an artificial clay-lined habitat in Siskiyou County (California Natural Diversity Database 2003).

Reproduction and Dispersal

Bogg's Lake hedge-hyssop seeds germinate when pools become inundated, and growth begins underwater. The plants complete a rapid life cycle during the period when vernal pools have begun to dry but still contain shallow water up to 5 centimeters (2 inches) deep (Kaye et al. 1990; Corbin et al. 1994). Bogg's Lake hedge-hyssop, which flowers between April and June, is thought to be self-pollinated (Kaye et al. 1990). Studies in Oregon found that seed production was unchanged whether or not insects were excluded during flowering. Insects were not observed visiting flowers under natural conditions (Kaye et al. 1990). Fruits mature within 1–2 weeks after flowering begins. Soon after seed set, the plants die and the seeds disperse (Corbin et al. 1994). Seed dispersal agents have not been studied, although CNDDB records for two Tehama County occurrences note that seeds may have been carried to the site (by birds or humans) from nearby occurrences (California Natural Diversity Database 2003). Seed longevity in nature has not been studied, although an interval of 3 years between observations of growing plants on the Lassen National Forest suggests that dormancy can persist for at least that long. Flowering at higher elevations occurs as late as August. (Corbin et al. 1994.) Dead plants disintegrate rapidly, leaving no identifiable remains in the dry pools.

Ecological Relationships

Populations are usually composed of scattered individuals, but plants have been observed aggregated in groups in cattle hoof prints (Mason and Bacigalupi 1954). Bogg's Lake hedge-hyssop often grows in association with bractless hedge-hyssop (*Gratiola ebracteata*) and Orcutt's quillwort (*Isoetes orcuttii*) in sparsely vegetated areas. Other vernal pool associates include hairy clover-fern (*Marsilea vestita* subsp. *vestita*), popcornflowers (*Plagiobothrys* spp.), downingias (*Downingia* spp.) Howell's quillwort (*Isoetes howellii*), Nuttall's quillwort (*Isoetes nuttallii*), coyote thistle (*Eryngium* spp.), woolly-heads (*Psilocarphus* spp.), creeping spikerush (*Eleocharis macrostachya*), and smooth goldfields (*Lasthenia glaberrima*).

Bogg's Lake hedge-hyssop also co-occurs with several other rare plants, including slender orcutt grass (*Orcuttia tenuis*), Greene's tuctoria (*Tuctoria greenei*), succulent owl's-clover (*Castilleja campestris* subsp. *succulenta*), hairy orcutt grass (*Orcuttia pilosa*), Hoover's spurge (*Chamaesyce hooveri*), legenere (*Legenere limosa*), many-flowered navarretia (*Navarretia leucocephala* subsp. *plieantha*), Sacramento orcutt grass (*Orcuttia viscida*), and San Joaquin Valley orcutt grass (*Orcuttia inaequalis*). Within the Phase I Planning Area, Bogg's Lake hedge-hyssop occurs with dwarf downingia (*Downingia pusilla*). (California Natural Diversity Database 2002.)

Population Threats

Although many new populations of Bogg's Lake hedge-hyssop have been discovered in recent years, both the quality and quantity of available habitat have declined during the same time period as vernal pools have been removed for agricultural and urban development and damaged by overgrazing and off-road vehicle traffic (California Department of Fish and Game 1992a, 1998; Corbin et al. 1994; California Natural Diversity Database 2003). Populations have also been disturbed or extirpated by hydrologic alteration and by disking and grading (Kaye et al. 1990; California Natural Diversity Database 2003).

Several populations in Modoc County are threatened by competition from Medusa-head (*Taeniatherum caput-medusae*) (Corbin et al. 1994). Several of the small occurrences (fewer than 100 plants observed) are undergoing rapid declines and are in danger of extirpation from chance events (Menges 1991; California Natural Diversity Database 2003). The 47 occurrences of Bogg's Lake hedge-hyssop located on U.S. Forest Service or U.S. Bureau of Land Management lands may be threatened by grazing, logging activities, recreational use, hydrologic alteration, road construction, fire suppression activities, competition from annuals, and herbicide drift (Corbin et al. 1994; California Natural Diversity Database 2003).

Grazing and trampling may threaten several of the approximately 48 occurrences that are known to be grazed (California Natural Diversity Database 2003). Grazing is detrimental if livestock use is concentrated in a small area or if it occurs before seed set. Grazing may be a compatible land use if it occurs after seed set (Mason and Bacigalupi 1954; California Department of Fish and Game 1987a). Additional studies are needed to establish compatible grazing standards.

Ahart's Dwarf Rush (*Juncus leiospermus var. ahartii*)

Status

FEDERAL: None

STATE: None

OTHER: California Native Plant Society List 1B

RECOVERY PLAN: None

PLACER NCCP/HCP CATEGORY: Class 3

Distribution

California

Ahart's dwarf rush is endemic to northern California where it occurs in Butte, Calaveras, Placer, Sacramento, and Yuba Counties. Current information indicates that there are seven extant occurrences and two possibly extirpated occurrences throughout its range (California Natural Diversity Database 2002; Witham pers. comm.). Half the known occurrences are located in Butte County; most of these are on the Peter Ahart ranch (California Natural Diversity Database 2002).

Placer County Phase I Planning Area

Historical

Virtually no information exists on the historical distribution of Ahart's dwarf rush prior to urbanization and agricultural development in Placer County. Ahart's dwarf rush was described in 1986 (Ertter 1986), and the earliest known collections are from 1973.

Current

Ahart's dwarf rush is currently reported in Placer County from a single occurrence east of the Lincoln Airport (Figure 24). However, communications with a knowledgeable individual indicate that the population has been extirpated (Witham pers. comm). No other occurrences are known from the Phase I Planning Area.

Population Status and Trends

California

Occurrences in California are located mostly on private property; however, one population is found on Sacramento County property, and another is found on City of Oroville property (California Natural Diversity Database 2002). Most of the occurrences have not been observed in at least 15–20 years; accordingly, the current status and trend of populations in California are unknown.

Placer County Phase I Planning Area

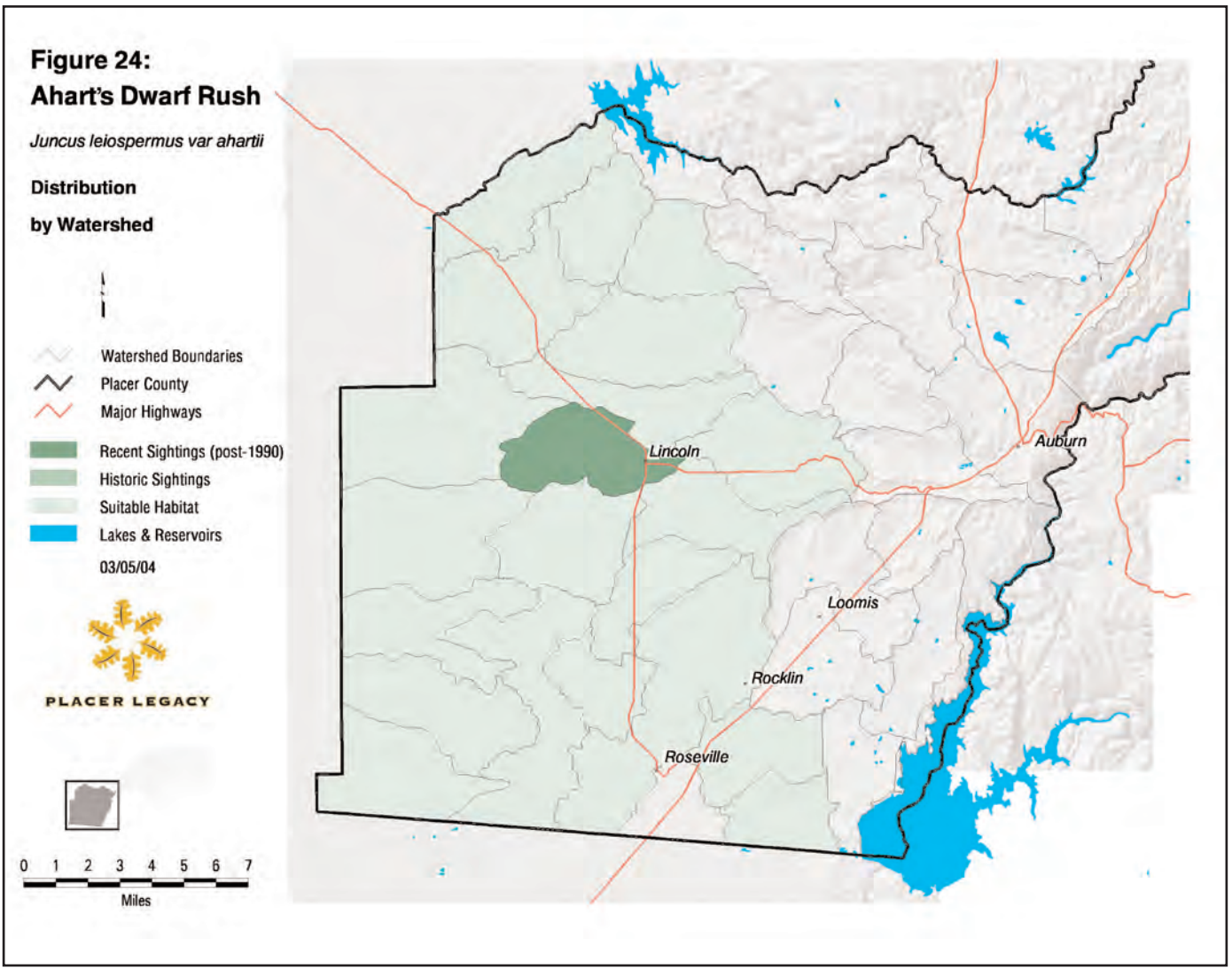
One occurrence of Ahart's dwarf rush is known from Placer County approximately 1 mile east of the Lincoln Airport (California Natural Diversity Database 2002). The population was reported in 1990 and has not been observed since. Communications with a knowledgeable individual suggest that this population has been extirpated from this location because surveys failed to refind the population and the site was planned for development (Witham pers. comm.).

Natural History

Life History

Ahart's dwarf rush is a small, reddish, grass-like annual in the rush family (Juncaceae). Plants range in height from less than an inch to 2.5 inches (Swab 1993). Each plant produces as many as 100 slender stems from its base, but the individual stems do not branch. The grass-like leaves arise from the base and are approximately

Ahart's Dwarf Rush Photo by Rob Schlising



half as long as the stems. The flowering period for Ahart's dwarf rush is April–May; each stem produces a single flower at its tip (California Native Plant Society 2001).

Habitat Requirements

Ahart's dwarf rush occurs on vernal pool margins and mesic valley and foothill grassland areas at elevations of 30–100 meters (Swab 1993; California Native Plant Society 2001). Very little information exists on the specific habitat requirements of Ahart's dwarf rush; however, information in the CNDDB indicates that the species prefers vernal pools with short inundation durations (i.e., "flashy" vernal pools) and/or the upper margins of deeper vernal pools. Within these habitats, it typically occurs in disturbed areas such as farmed fields or on gopher turnings (California Natural Diversity Database 2002; Witham pers. comm.).

Reproduction and Dispersal

Ahart's dwarf rush flowers April–May, then produces a spherical or egg-shaped capsule containing many smooth seeds. The species is wind pollinated. Due to the small size of this plant and its seeds, dispersal likely occurs within the habitat occupied by the parent plant.

Ecological Relationships

Very little information exists on the ecology of Ahart's dwarf rush. Reports in the CNDDB (2002) and communication with a knowledgeable individual indicate that the species either prefers some level of surface disturbance or does well in areas where the competition has been dramatically reduced (Witham pers. comm.).

Most plants within a population are often found in these disturbed areas. Common associates found with Ahart's dwarf rush include Fremont's goldfields (*Lasthenia fremontii*), annual hairgrass (*Deschampsia danthonioides*), and Vasey's coyote thistle (*Eryngium vaseyi*), as well as other superficially similar species of *Juncus* (*Juncus uncialis*, *J. bufonius*, and *J. capitatus*) (California Natural Diversity Database 2002).

Population Threats

Because many of the known populations of Ahart's dwarf rush in California have not been observed in at least 15–20 years, very little information exists on the current threats to the species. Many of the known populations are located on private land and have not been accessible for study. Development is the only threat published in the literature (California Native Plant Society 2001); however, it has been suggested that off-road vehicle use, road construction, livestock grazing, and other disturbances from adjacent development can also affect the species (U.S. Fish and Wildlife Service 2002).

Red Bluff Dwarf Rush (*Juncus leiospermus var. leiospermus*)

Status

FEDERAL: None.

STATE: None.

OTHER: California Native Plant Society List 1B.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

Red Bluff dwarf rush is endemic to northern California where it occurs in Butte, Placer, Shasta, and Tehama counties. Current information indicates that there are 36 extant occurrences and two possibly extirpated occurrences throughout its range (California Natural Diversity Database 2003).

Placer County Phase I Planning Area

Historical

Virtually no information exists on the historical distribution of Red Bluff dwarf rush prior to urbanization and agricultural development. *Juncus leiospermus* was described in 1948 (Hermann 1948). Prior to that, it had been collected only three times: in 1894, 1916, and 1937 (CalFlora 2002). It was not observed again until 1978 (Ahart 1980).

Current

Only one occurrence is currently known in the Phase I Planning Area. The population, just north of Roseville (Figure 25), was observed in 1982 but was not relocated in 1997 (California Natural Diversity Database 2003). This population represents the southernmost occurrence of Red Bluff dwarf rush discovered to date. The nearest population occurs in Butte County approximately 60 miles to the north (California Natural Diversity Database 2003).

Population Status and Trends

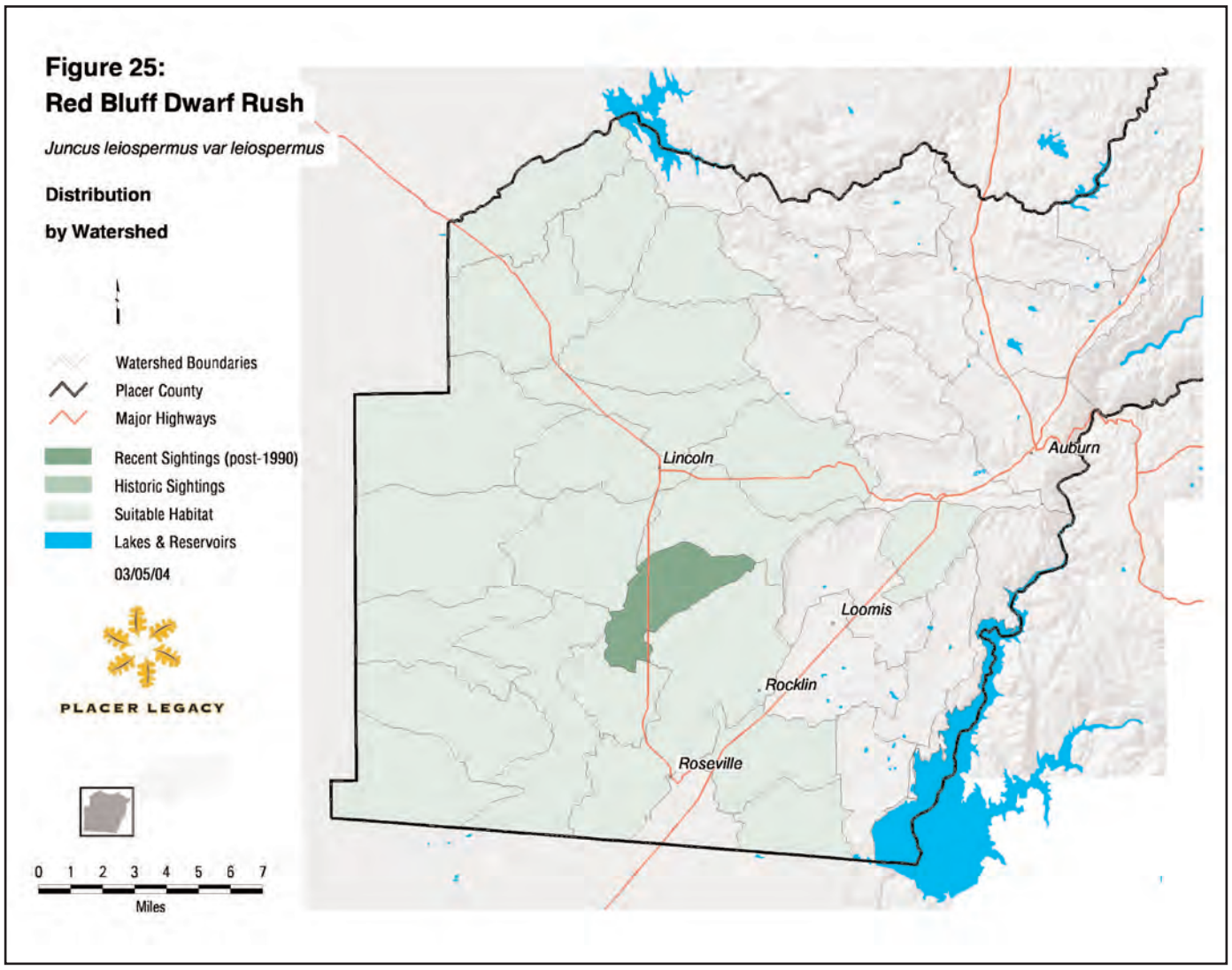
California

Very little information is available on the status and trends of Red Bluff dwarf rush in California. Many populations are represented by historic records and have not been seen in a number of years. Occurrences in California are located mostly on private property; however, the species is also found on lands owned or managed by the U.S. Forest Service (Lassen National Forest), the Bureau of Land Management, and the City of Redding.

Placer County Phase I Planning Area

Only one population of Red Bluff dwarf rush, located near Roseville, has been recorded in Placer County. The population was last seen in 1982; however, a habitat survey conducted in 1997 indicates that the habitat is still present. Because of extensive recent development that has occurred in and around Roseville, this population may have been extirpated; accordingly, confirmation is needed (see *Data Gaps and Conservation Implications* below).

Red Bluff Dwarf Rush Photo by Dean Taylor (Jepson Herbarium)



Natural History

Life History

Red Bluff dwarf rush is a small reddish grass-like annual that grows to approximately 6.4 centimeters (2.5 inches) tall (Swab 1993). Each plant can produce as many as 100 slender stems from its base, but the individual stems do not branch. The grass-like leaves arise from the base and are approximately half as long as the stems. Each stem generally produces two to seven flowers from March through May (California Native Plant Society 2001).

Habitat Requirements

Red Bluff dwarf rush is known to occur in a variety of habitats, including meadows and seeps, vernal pools, and vernally mesic areas in chaparral, cismontane woodland, and valley and foothill grassland from 35 to 1,020 meters (115 to 3,350 feet) (California Native Plant Society 2001). Very little information exists on the specific habitat requirements of Red Bluff dwarf rush; however, information in the CNDDB (2003) indicates that the species prefers areas that have saturated soils during the rainy season (November–April), such as vernal pools and vernally mesic areas.

Reproduction and Dispersal

Red Bluff dwarf rush has an annual lifecycle, generally flowering from March through May. The species is wind pollinated. Due to the small size of this plant and its seeds, dispersal likely occurs within the habitat occupied by the parent plant.

Ecological Relationships

Very little information exists on the ecology of Red Bluff dwarf rush. Reports in the CNDDB (2003) and correspondence with a knowledgeable individual indicate that the species prefers areas with significant surface disturbance or where the competition has been dramatically reduced (Witham pers. comm.). Often, most plants within a population are found in these disturbed areas. Occurrence records indicate the species is found in association with other species requiring vernally mesic conditions; these include other species of *Juncus* (*Juncus bufonius, J. capitatus, J. phaeocephalis*); popcornflowers (*Plagiobothrys* spp.); and coyote thistle (*Eryngium castrense*).

Population Threats

Because many of the known populations of Red Bluff dwarf rush have not been observed in at least 15–20 years, very little information exists on current threats to the species. Many of the known populations are located on private land. Published threats to the species include urban development, industrial forestry, and agricultural conversion (California Native Plant Society 2001). Land uses adjacent to occupied habitat are thought to affect the hydrological regime, introduce exotic species, encourage recreational use, and encourage trash dumping. The full effects of these factors are unknown.

Vernal Pool Fairy Shrimp (*Branchinecta lynchi*)

Status

FEDERAL: Threatened (59 FR 48136, September 19, 1994).

STATE: None.

OTHER: G2G3/S2S3.

RECOVERY PLAN: *Vernal Pools of Northern California* (under development).

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

California

Vernal pool fairy shrimp (*Branchinecta lynchi*) (Eng, Belk, and Eriksen 1990) is endemic to California. The historical range encompasses annual grasslands of the Great Central Valley. Currently, the species ranges from Red Bluff in Shasta County south to Tulare County. Disjunct populations also occur on the Santa Rosa Plateau in Riverside County and in the South Coast Ranges (Eriksen and Belk 1999).

Vernal pool fairy shrimp have been reported from the following California Vernal Pool Regions: Northwest Sacramento Valley, Northeast Sacramento Valley, Southeast Sacramento Valley, Solano-Colusa, Livermore, Central Coast, Carrizo, San Joaquin Valley, South Sierra Foothills, and Western Riverside County (California Department of Fish and Game 1998b). The CNDDB (2003) lists 329 occurrences of vernal pool fairy shrimp statewide.

Placer County Phase I Planning Area

Historical

The historical distribution of vernal pool fairy shrimp can only be inferred from the historical distribution of its habitat. Annual grasslands of western Placer County, particularly within the Great Valley ecoregion, probably supported a patchy distribution of vernal pool fairy shrimp.

Current

There are 37 known populations of vernal pool fairy shrimp in the Phase I Planning Area, which is within the Southeastern Sacramento Valley Vernal Pool Region (California Department of Fish and Game 1998b; California Natural Diversity Database 2003).

The majority of extant populations in the Phase I Planning Area occur in vernal pools of the northern hardpan and north volcanic mudflow types. These vernal pool types are common to the areas surrounding the Placer County cities of Roseville, Lincoln, and Rocklin (Figure 26).

Population Status and Trends

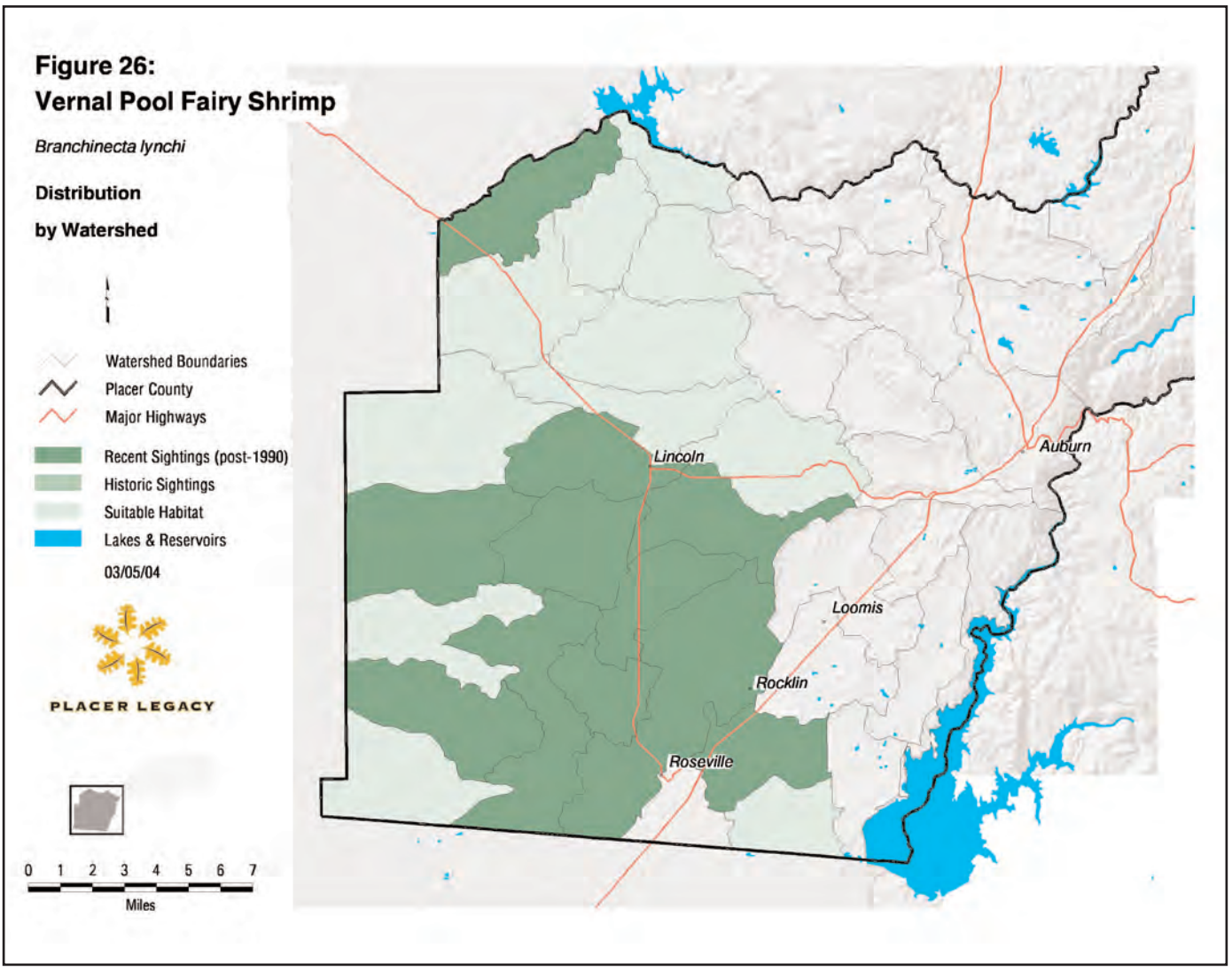
California

The CNDDB (2003) lists 329 occurrences of vernal pool fairy shrimp in California. Although vernal pool fairy shrimp are widely distributed, they are now locally uncommon throughout their historical range. Where vernal pool fairy shrimp co-occur with other shrimp species, they are always outnumbered by the other species (Eriksen and Belk 1999).

Placer County Phase I Planning Area

The CNDDB (2003) lists 37 occurrences of vernal pool fairy shrimp in the Phase I Planning Area. Several nature preserves and mitigation banks have been established in the Phase I Planning Area with the partial goal of preserving habitat for vernal pool fairy shrimp.

Vernal Pool Fairy Shrimp Jones & Stokes file photo



These preservation areas include Aitken Ranch Mitigation Bank, Wildlands Mitigation Bank, Orchard Creek Preservation Area, Eastridge Southern Wetland Preserve, Sterling Pacific Assets' Lincoln Crossing Mitigation Site, and the City of Roseville's Woodcreek Compensation Area (Jones & Stokes file data, California Natural Diversity Database 2003).

Natural History

Habitat Requirements

Vernal pool fairy shrimp inhabit rain-filled ephemeral pools (i.e., vernal pools) that form in depressions, usually in grassland habitats (Eng et al. 1990). Pools must fill frequently and persist long enough for the species to complete its lifecycle, which takes place entirely within vernal pools. Pools occupied by vernal pool fairy shrimp often have grass or mud bottoms and clear to tea-colored water; they are often in basalt flow depression pools in unplowed grasslands. Water chemistry is key in determining fairy shrimp occurrence; alkalinity, total dissolved solids (TDS), and pH are some of the most important factors (Eriksen and Belk 1999). Vernal pool fairy shrimp inhabit alkaline pools, ephemeral drainages, rock outcrop pools, ditches, stream oxbows, stockponds, vernal pools, vernal swales, and other seasonal wetlands. Occupied habitats range in size from rock outcrop pools as small as 0.8 square meter, (1 square yard) to large vernal pools up to 4.5 hectares (11 acres). The maximum potential water depth of occupied habitat ranges from 3 to 122 centimeters (1.2 to 48 inches) (Helm 1998; Eriksen and Belk 1999; U.S. Fish and Wildlife Service 2001).

Vernal pools are characterized by a specific flora endemic to the hydrology and soil composition of the habitat. Vernal pool fairy shrimp and other fairy shrimp species have been observed in depressions filled with water that do not meet the definition of vernal pools (Helm 1998; Stone pers. comm.). Examples of non–vernal pool habitats are roadside ditches, wheel-ruts left by off-highway vehicles or other heavy equipment, and railroad toe-drains (Helm 1998). Vernal pool fairy shrimp are not found in riverine, estuarine, or other permanent waters that support fish (50 FR 48136–48153, September 16, 1994).

Reproduction

Male vernal pool fairy shrimp seek out females visually. Using specialized second antennae, the male grasps the female between the last pair of phyllopods and the brood pouch. Sperm are released directly into the female's brood pouch during copulation. Following insemination, the female releases eggs from lateral pouches into the ovisac, where the eggs are fertilized (Eriksen and Belk 1999).

Following fertilization, embryonic and cyst development begin. Embryonic development ceases when the late gastrula stage is reached. At that point, metabolism slows and a halted embryo is isolated from the environment by development of a many-layered membranous shell. The embryo and the shell comprise the cyst, or resting egg. Females carry cysts in a brood sac. Cysts are dropped to the pool bottom or remain in the female's brood sac until the female dies. Cysts are capable of withstanding heat, cold, and prolonged desiccation. When occupied pools fill with water in the same or subsequent seasons, some, but not all, of the deposited cysts may hatch. The egg bank in the soil may comprise cysts from several years of breeding. When the vernal pools fill with rainwater and the water temperature drops below 10°C (50°F), the resting eggs hatch into small nauplii. The early stages of vernal pool fairy shrimp develop rapidly into adults, reaching maturity in as little as 18 days (Eriksen and Belk 1999).

Dispersal Patterns

Vernal pool fairy shrimp disperse locally during extremely wet years when individual pools in a complex spill into or are connected with adjacent pools. Long-distance dispersal can result from cysts being carried on the wind and on the bodies or in the guts of larger animals. Cysts, including those still in brood sacs, can pass undamaged and undigested through the digestive tracts of birds (Proctor et al. 1967 cited in Eriksen and Belk 1999); subsequent deposition of fecal matter can result in the inoculation of a new site. Cysts trapped in mud can adhere to the feet and feathers of waterfowl and the hooves and fur of grazing mammals and be transported to the dried mud of different vernal pool complexes (Eriksen and Belk 1999). Cysts may also be transported between pools in the digestive tracts of amphibian predators such as frogs and salamanders (Rogers pers. comm.).

Longevity

Vernal pool fairy shrimp can reach maturity as soon as 18 days after hatching. In lower water temperatures (less than 14°C [57°F]), individuals have been observed to require 41 days to mature (Helm 1998). Vernal pool fairy shrimp has the shortest maximum longevity of any fairy shrimp species observed in California: 139 days. Based on laboratory observations, Helm (1998) determined the mean longevity to be 90 days. Field observations indicate that vernal pool fairy shrimp typically persist only 10–12 weeks (Eriksen and Belk 1999; Stone pers. comm.). Populations of vernal pool fairy shrimp often disappear long before the vernal pools dry up (U.S. Fish and Wildlife Service 2001).

Sources of Mortality

The greatest sources of mortality to vernal pool fairy shrimp are predation and heatstroke. Vernal pool fairy shrimp are unable to filter oxygen from their aquatic habitat when water temperatures remain above 70°F (21°C). In addition, both adult fairy shrimp and diapausing cysts can be crushed by foot traffic and off-highway vehicles. (Hathaway et al. 1996.)

Behavior

Vernal pool fairy shrimp are omnivorous filter feeders that indiscriminately filter particles of the appropriate size from their surroundings. The diet consists of bacteria and plant and animal particles, including suspended unicellular algae and metazoans (Eriksen and Belk 1999).

Adults use eleven pairs of legs, or phyllopods, for locomotion, to filter suspended food particles from the environment, and for respiration. Vernal pool fairy shrimp typically swim in a 'zig-zag' or 'figure-eight' pattern with the phyllopods oriented toward the water surface (i.e., they swim on their backs).

Movement and Migratory Patterns

The presence of vernal pool fairy shrimp adults coincides with the filling and drying pattern of the vernal pool habitats. Adult populations are typically present from mid-December through mid-March (Eriksen and Belk 1999). Resting cysts are always present in an occupied pool basin.

Ecological Relationships

Fairy shrimp are preyed upon by migratory waterfowl, amphibians, predatory diving beetles (Coleoptera: Dytiscidae), water boatmen (Hemiptera: Corixidae), and vernal pool tadpole shrimp. Large freshwater branchiopods in California serve as an important source of protein and energy for migratory waterfowl (Eriksen and Belk 1999). Many vernal pools occur along the Pacific flyway; the use of these pools as resting and feeding grounds by migratory birds is well documented (Silveria 1998; Sterling pers. comm.).

Vernal pool fairy shrimp rarely co-occur with other invertebrate species, but when they are found in mixed assemblages they are never the most abundant species (50 FR 48136–48153, September 16, 1994). The two species most likely to co-occur with vernal pool fairy shrimp are California linderiella (*Linderiella occidentalis*) and vernal pool tadpole shrimp (*Lepidurus packardi*). Only very rarely do vernal pool fairy shrimp co-occur with other *Branchinecta* species (Eriksen and Belk 1999).

Population Threats

The greatest threats to the persistence of vernal pool fairy shrimp are habitat loss and degradation resulting from urban development and agriculture. Vernal pools occur in large, flat, open grasslands that are ideal for a number of economic uses, including airports, military bases, rice and grain fields, cattle grazing, aggregate mining, and urban development.

Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)

Status

FEDERAL: Endangered (59 FR 48136, September 19, 1994).

STATE: None.

OTHER: None.

RECOVERY PLAN: Vernal Pools of Northern California (under development).

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

California

Vernal pool tadpole shrimp (*Lepidurus packardi* Simon, 1886) is reported as endemic to the Central Valley of California (59 FR 48136, September 19, 1994; Helm 1998; Rogers 2001;). Rogers (2001) determined that specimens from southern Oregon and the California Great Basin that were originally described as *L. packardi* were, in fact, *Lepidurus cryptus*, a recently described species of tadpole shrimp.

The historical range of vernal pool tadpole shrimp in California includes annual grasslands of the Great Central Valley. Today the species ranges from Shasta County in the north to Merced County in the south, with one disjunct population occurring in western Alameda County (Rogers 2001).

Vernal pool tadpole shrimp have been reported from the following California Vernal Pool Regions: Northwest Sacramento, Northeast Sacramento, Southeast Sacramento, Solano, San Joaquin, and South Sierra Foothill (California Department of Fish and Game 1998b). The CNDDB (2003) lists 167 occurrences of vernal pool tadpole fairy shrimp in California.

Placer County Phase I Planning Area

Historical

The historical distribution of vernal pool tadpole shrimp can only be inferred from the historical distribution of its habitat. Annual grasslands of western Placer County, particularly within the Great Valley ecoregion, probably supported a patchy distribution of vernal pool tadpole shrimp (Rogers pers. comm.).

Current

There are two known populations of vernal pool tadpole shrimp in western Placer County (Figure 27). The first is located on the U.S. Air Force Lincoln Communications Facility where, in 1996, at least five vernal pools of a 236-pool complex were observed to support vernal pool tadpole shrimp (36 pools were surveyed). The second is on the Woodcreek Oaks Mitigation Site between Kasenburg Creek and the south branch of Pleasant Grove Creek; vernal pool tadpole shrimp adults were observed in one pool on this site in 1995. An additional population was observed in 1993 west of the Phase I Planning Area in Sutter County (California Natural Diversity Database 2003).

Population Status and Trends

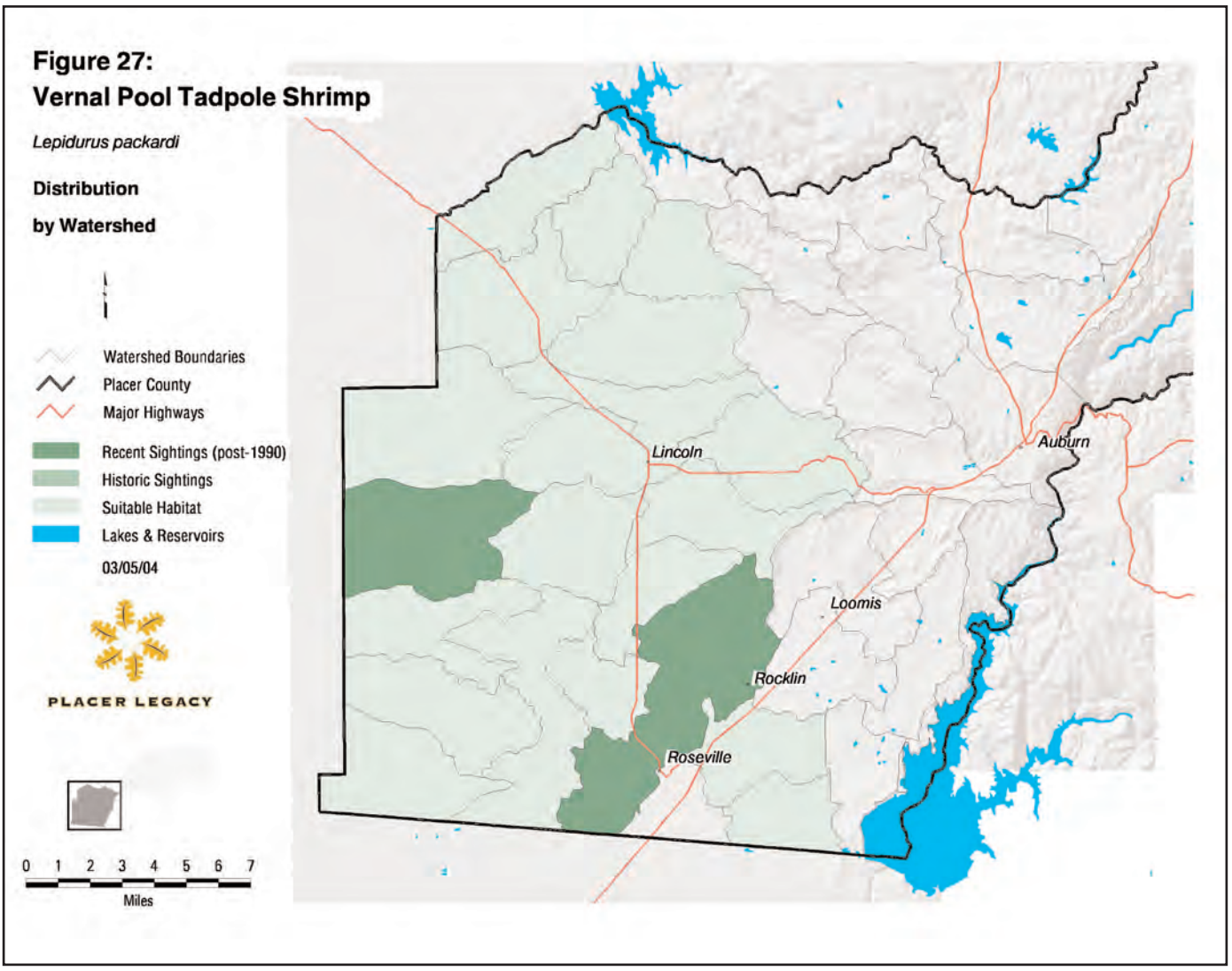
California

The CNDDB (2003) lists 167 occurrences of vernal pool tadpole shrimp in California. Although vernal pool tadpole shrimp is widely distributed in California, it is now locally uncommon throughout the historical range (Helm 1998; Eriksen and Belk 1999).

Placer County Phase I Planning Area

The CNDDB (2003) lists two occurrences of vernal pool tadpole shrimp within the Phase I Planning Area. Several nature preserves and mitigation banks have been established in the Phase I Planning Area with the partial goal of preserving habitat for vernal pool tadpole shrimp. These preservation areas include Aitken Ranch Mitigation Bank, Wildlands Mitigation Bank, Orchard Creek Preservation Area, Eastridge Southern Wetland Preserve, Sterling Pacific Assets' Lincoln Crossing Mitigation Site, and the City of Roseville's

Vernal Pool Tadpole Shrimp Jones & Stokes file photo



Woodcreek Compensation Area (California Natural Diversity Database 2003; Jones & Stokes file data).

Natural History

Habitat Requirements

Vernal pool tadpole shrimp occur in a variety of natural and artificial seasonally inundated habitats. They require seasonally aquatic habitats that are wet for at least 7 weeks and dry in summer (Gallagher 1996). Helm (1998) observed vernal pool tadpole shrimp occurring in vernal pools (natural, artificial, and constructed); seasonal wetlands (natural and artificial); alkaline pools; clay flats; vernal swales; stockponds; railroad right-of-way pools; roadside ditches; and road rut pools resulting from vehicular activity. Occupied pools and wetlands typically have highly turbid waters or aquatic vegetation that may provide shelter from predators (59 FR 48136, September 19, 1994; Stone pers. comm.).

Reproduction

Vernal pool tadpole shrimp are bisexual. Diapausing cysts (eggs) occurring in the dry pool bottom hatch within 3 weeks of inundation (Ahl 1991). The hatched neonate is a metanauplius that undergoes several molts, each gaining additional phyllopod appendages until reaching sexual maturity. This process takes approximately 6–7 weeks depending on temperature and food availability (Ahl 1991; Gallagher 1996; Helm 1998). Reproduction occurs throughout the ponding season, when females average 10–12 millimeters (0.39–0.47 inch) in carapace length (Ahl 1991). Laboratory studies conducted by Ahl (1991) revealed that eggs can hatch during the same ponding event in which they were laid without intervening dehydration. The remaining unhatched cysts settle to the pool substrate and contribute to the cyst bank for subsequent wet seasons.

Dispersal Patterns

Vernal pool tadpole shrimp disperse locally during extremely wet years when individual pools in a complex spill into or are connected with adjacent pools. Long-distance dispersal can result from cysts being carried on the wind and on the bodies or in the guts of larger animals. Cysts, including those still in brood sacs, can pass undamaged and undigested through the digestive tracts of birds (Proctor et al. 1967 cited in Eriksen and Belk 1999); subsequent deposition of fecal matter can result in the inoculation of a new site. Cysts trapped in mud can adhere to the feet and feathers of waterfowl and the hooves and fur of grazing mammals and be transported to the dried mud of different vernal pool complexes (Eriksen and Belk 1999). Cysts may also be transported between pools in the digestive tracts of amphibian predators such as frogs and salamanders (Rogers pers. comm.).

Longevity

Vernal pool tadpole shrimp is considered a long-lived species. Adults are often present and reproductive until the pools dry up in the spring (59 FR 48136, September 19, 1994).

Sources of Mortality

The greatest sources of mortality to vernal pool tadpole shrimp are predation and desiccation. Tadpole shrimp are left exposed when their habitat dries up. In addition, both adult shrimp and diapausing cysts can be crushed by foot traffic and off-highway vehicles (Hathaway 1996).

Movement and Migratory Patterns

The presence of vernal pool tadpole shrimp adults coincides with the filling and drying pattern of the vernal pool habitats. Populations of adults are typically present from December through early April. Resting cysts are always present in an occupied pool basin.

Behavior

Vernal pool tadpole shrimp are filter feeders and opportunistic predators on aquatic insect larvae, segmented worms (Oligochaeta), water fleas (Cladocera), seed shrimp (Ostracoda), copepods (Copepoda), fairy shrimp (Anostraca), and other vernal pool tadpole shrimp. This species hunts by moving along the pool bottom or aquatic vegetation, stirring up the muddy substrate, and capturing prey items with its phyllopods to direct them into the feeding groove or mouth (Rogers pers. comm.). This feeding behavior and predator avoidance leads to vernal pool tadpole shrimp being most often observed at the pool bottom.

Ecological Relationships

Vernal pool tadpole shrimp are preyed on by migratory waterfowl, amphibians, predatory diving beetles (Coleoptera: Dytiscidae), water boatmen (Hemiptera: Corixidae), and other vernal pool tadpole shrimp. Large freshwater branchiopods in California serve as an important source of protein and energy for migratory waterfowl (Eriksen and Belk 1999). Many vernal pools occur along the Pacific flyway; the use of these pools as resting and feeding grounds by migratory birds is well documented (Silveria 1998; Sterling pers. comm.).

Vernal pool tadpole shrimp commonly co-occur with vernal pool fairy shrimp (*Branchinecta lynchi*), Conservancy fairy shrimp (*Branchinecta conservatio*), and California linderiella (*Linderiella occidentalis*) (Helm 1998; Stone pers. comm.).

Population Threats

The greatest threats to the persistence of vernal pool tadpole shrimp are habitat loss and degradation resulting from urban development and agriculture. Vernal pools occur in large, flat, open grasslands that are ideal for a number of economic uses including airports, military bases, rice and grain fields, cattle grazing, aggregate mining, and urban development.

California Linderiella (*Linderiella occidentalis*)

Status

FEDERAL: None.

STATE: None.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

California linderiella (*Linderiella occidentalis*) (Dodds1923) is a fairy shrimp species endemic to California. In the Central Valley, it ranges from near Redding in the north to as far south as Fresno County, with populations more concentrated east of the Sacramento and San Joaquin Rivers. More scattered populations occur through the coastal region from Mendocino to Ventura Counties. (Eriksen and Belk 1999.)

Placer County Phase I Planning Area

Historical

The historical distribution of California linderiella can only be inferred from the historical distribution of its habitat. Annual grasslands of western Placer County, particularly in the Great Valley ecoregion, probably supported a patchy distribution of the species.

Current

The CNDDB (2003) lists 25 occurrences of California linderiella within the Phase I Planning Area. The species is distributed throughout the vernal pool grasslands surrounding the cities of Roseville, Rocklin, and Lincoln (Figure 28).

Population Status and Trends

California

The CNDDB (2003) lists 207 occurrences of California linderiella in California. This species of fairy shrimp has been reported from the following Vernal Pool Regions: Northeast Sacramento Valley, Southeast Sacramento Valley, Santa Rosa, Central Coast, San Joaquin Valley, South Sierra Foothills, and Santa Barbara (California Department of Fish and Game 1998b).

California linderiella is the most common anostracan inhabitant of cool, soft-water pools in California's Central Valley grasslands. There have been significant historic losses of vernal pool habitat, and vernal pool habitat continues to be threatened by development (Eriksen and Belk 1999).

Placer County Phase I Planning Area

The CNDDB (2003) lists 25 occurrences of California linderiella within the Phase I Planning Area. Urban development and agricultural conversion threaten the majority of natural vernal pools in the Phase I Planning Area.

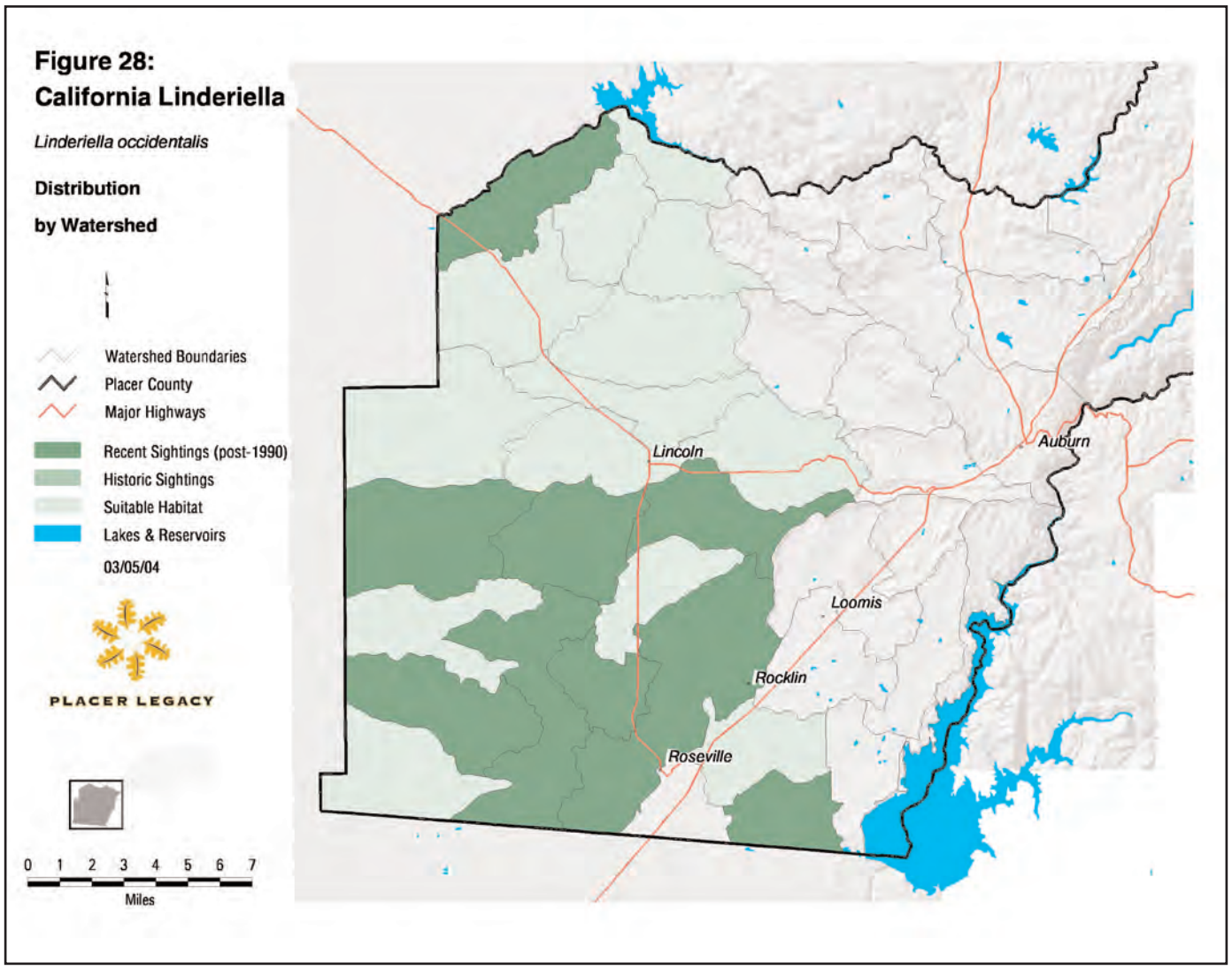
Significant public, government, and preserved lands that are known to support populations of California linderiella include the Wildlands Mitigation Bank, U.S. Air Force Lincoln Communications Facility, Orchard Creek Mitigation Bank, Woodcreek Wetland Compensation Area, Aitken Ranch Mitigation Bank, Diamond Oaks Mitigation Site, and The Highland Reserve (Jones & Stokes file data; California Natural Diversity Database 2003).

Natural History

Habitat Requirements

California linderiella inhabit rain-filled, ephemeral pools (i.e., vernal pools) that form in depressions, usually in grassland habitats (Eng et al. 1990). Pools must fill frequently and persist long enough for the species to complete its lifecycle, which takes place entirely within

California Linderiella Photo by Aundrea McCombs



vernal pools. Compared to other endemic Central Valley fairy shrimp species, California linderiella is the most tolerant of warm water and consequent low dissolved oxygen (Helm 1998). Most pools occupied by California linderiella are vegetated and contain clear water. However, it is not uncommon to observe California linderiella in mud-bottomed pools with slightly turbid water.

Water chemistry is key in determining fairy shrimp occurrence; alkalinity, total dissolved solids (TDS), and pH are some of the most important factors. California linderiella occurs in poorly buffered waters with low alkalinity (13–170 ppm) and low TDS (33–273 ppm) that can regularly fluctuate in pH from 6.2 to 8.5. California linderiella typically occupy reasonably large pools and may occur in roadside ditches in the Central Valley. (Eriksen and Belk 1999.)

Reproduction

Male fairy shrimp visually seek females. Using specialized second antennae, the male grasps the female between the last pair of phyllopods and the brood pouch. Sperm is released directly into the female's brood pouch during copulation. Following insemination, the female shrimp releases eggs from lateral pouches into the ovisac, where the eggs are fertilized. (Eriksen and Belk 1999.)

Following fertilization, embryo and cyst development begins. Embryonic development ceases when the late gastrula stage is reached. At this point metabolism slows, and the halted embryo is isolated from the environment by development of a many-layered membranous shell. The embryo and the shell comprise the cyst, or resting egg. Females carry the cysts in a brood sac. Cysts are dropped to the pool bottom or remain in the

female's brood sac until the female dies; they are capable of withstanding heat, cold, and prolonged desiccation. When the pools fill in the same or subsequent seasons, some, but not all, of the cysts may hatch. The egg bank in the soil may consist of cysts from several years of breeding. (Eriksen and Belk 1999.)

Dispersal Patterns

California linderiella disperse locally during extremely wet years when individual pools in a complex spill into or are connected with adjacent pools. Long-distance dispersal can result from cysts being carried on the wind and on the bodies or in the guts of larger animals. Cysts, including those still in brood sacs, can pass undamaged and undigested through the digestive tracts of birds (Proctor et al. 1967 cited in Eriksen and Belk 1999); subsequent deposition of fecal matter can result in the inoculation of a new site. Cysts trapped in mud can adhere to the feet and feathers of waterfowl and the hooves and fur of grazing mammals and be transported to the dried mud of different vernal pool complexes (Eriksen and Belk 1999). Cysts may also be transported between pools in the digestive tracts of amphibian predators such as frogs and salamanders (Rogers pers. comm.).

Longevity

California linderiella can achieve sexual maturity in as little as 31 days after hatching, although the average is 45 days. California linderiella has the longest maximum longevity of the fairy shrimp species endemic to the Central Valley: 168 days (6 months). (Helm 1998; Eriksen and Belk 1999). Helm (1998) determined the average longevity to be 139 days.

Sources of Mortality

The primary sources of mortality to California linderiella are predation and desiccation. California linderiella often persist until their habitat dries up and the shrimp are left exposed. In addition, both adults and diapausing cysts can be crushed by foot traffic and off-highway vehicles. (Hathaway et al. 1996.)

Behavior

California linderiella are omnivorous filter feeders that indiscriminately filter particles of the appropriate size from their surroundings. Diet consists of bacteria and plant and animal particles, including suspended unicellular algae and metazoans (Eriksen and Belk 1999). Adults use eleven pairs of legs, or phyllopods, for locomotion, to filter suspended food particles from their environment, and for respiration.

Movement and Migratory Patterns

The presence of California linderiella adults coincides with the filling and drying pattern of vernal pool habitats. Typically, populations of adults are present from late December to early May. (Eriksen and Belk 1999). Resting cysts are always present in an occupied pool basin.

Ecological Relationships

Fairy shrimp are preyed on by migratory waterfowl, amphibians, predatory diving beetles (Coleoptera: Dytiscidae), water boatmen (Hemiptera: Corixidae), and vernal pool tadpole shrimp (*Lepidurus packardi*). Large freshwater branchiopods in California serve as an important source of protein and energy for migratory waterfowl (Eriksen and Belk 1999). Many vernal pools occur along the Pacific flyway; the use of these pools as resting and feeding grounds by migratory birds is well documented (Silveria 1998; Sterling pers. comm.).

California linderiella commonly co-occur with vernal pool fairy shrimp (*Branchinecta lynchi*) and vernal pool tadpole shrimp (*Lepidurus packardi*). California linderiella has been infrequently collected with Conservancy fairy shrimp (*Branchinecta conservatio*) (Eriksen and Belk 1999).

Population Threats

The greatest threats to the persistence of California linderiella are habitat loss and degradation resulting from urban development and agriculture. Vernal pools occur in large, flat, open grasslands that are ideal for a number of economic uses including airports, military bases, rice and grain fields, cattle grazing, aggregate mining, and urban development. Many of the suitable vernal pools for this species in the Phase I Planning Area have been lost to development.

Valley Elderberry Longhorn Beetle (*Desmocerus californicus dimorphus*)

Status

FEDERAL: Threatened.

STATE: None.

OTHER: G3T2/ S2.

RECOVERY PLAN: *Recovery Plan for the Valley Elderberry Longhorn Beetle* (1984).

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

California

Valley elderberry longhorn beetle (VELB) (*Desmocerus californicus dimorphus*) (Fisher 1921) is endemic to the riparian areas of the Central Valley of California (Linsley and Chemsak 1972). Neither subspecies of *Desmocerus californicus* Horn, 1881, has been observed outside of California. Three species of the genus *Desmocerus* (Audinet-Serville) occur in North America.

USFWS recognizes the range of VELB to include the watersheds of the American, San Joaquin, and Sacramento Rivers and tributaries of these watersheds at elevations below about 900 meters (3,000 feet). VELB primarily occurs in the greater Sacramento and northern San Joaquin Valleys. Barr (1991) reported the range of VELB to include all of the Central Valley from Shasta County in the north to Kern County in the south.

The range of VELB may overlap with that of *D. c. californicus* Horn along the eastern edge of the Coast Ranges and in the southern San Joaquin Valley (Halstead and Oldham 2000). Delineating the ranges of these two taxa will require focused distribution studies because of the reclusive nature, short-lived adult forms, and sexual dimorphism of VELB.

Placer County Phase I Planning Area

Historical

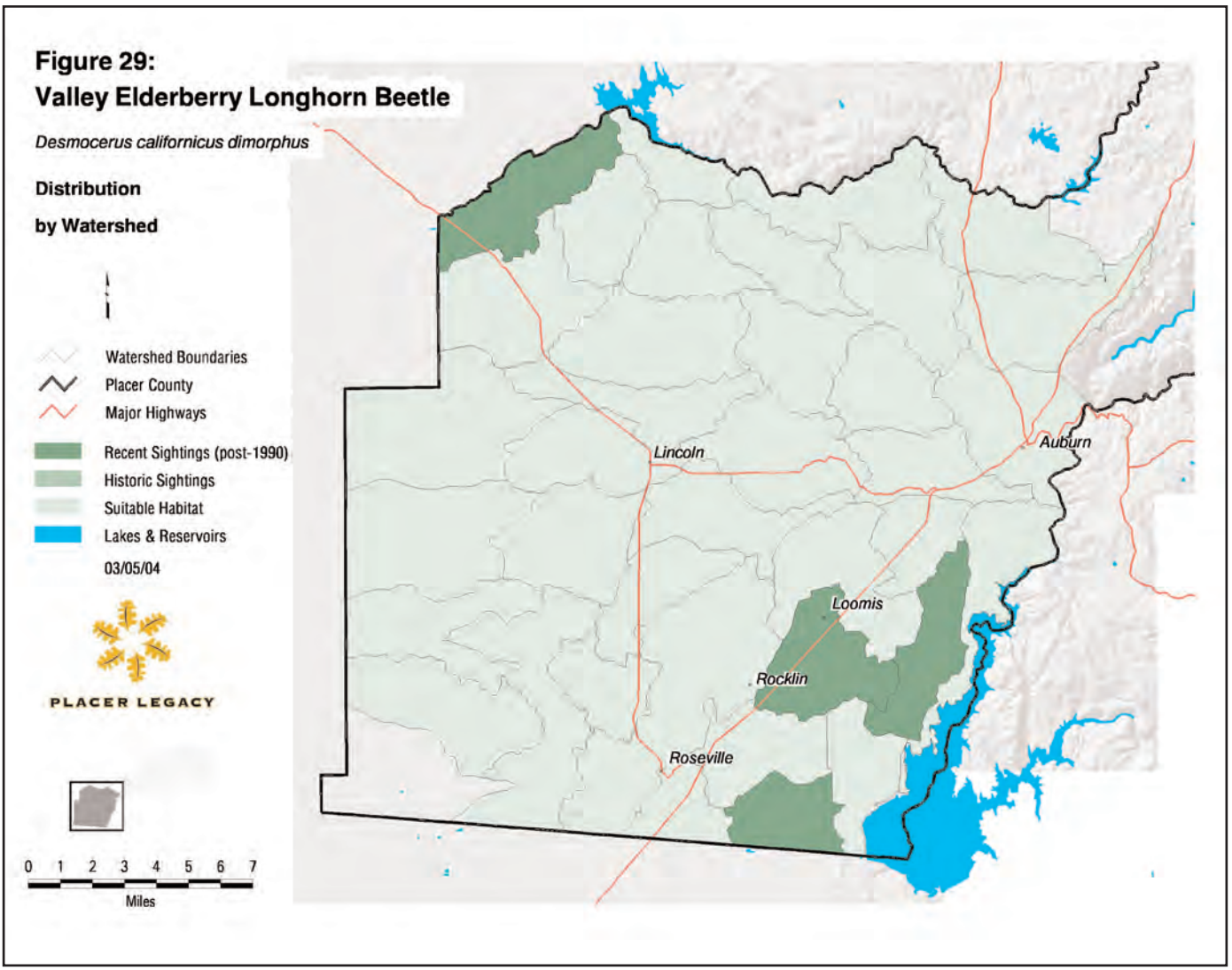
Riparian habitat occurred historically along low-elevation creeks, streams, and rivers throughout western Placer County. VELB is likely to have occurred in a patchy distribution along Bear River, Coon Creek, Markham Ravine, Auburn Ravine, Pleasant Grove Creek, Dry Creek, the American River, and associated tributaries that supported *Sambucus* sp. and associated riparian vegetation.

Current

VELB is known to occur in the American River watershed below Auburn in the vicinity of Folsom Lake; the Dry Creek watershed along Secret Ravine and Miners Ravine; Wildlands Mitigation Bank; and the Bear River watershed near Wheatland (Sutter County) (Figure 29). The taxon has not been observed in Placer County higher than 200 meters (650 feet) above sea level. (California Natural Diversity Database 2003).

Barr (1991) observed recent exit holes on red elderberry shrubs at two sites along Miners Ravine at elevations of 125 meters (410 feet) and 146 meters (480 feet); she also found old exit holes along Secret Ravine at 98 meters (320 feet).

Valley Elderberry Longhorn Beetle Photo by Greg Sutter



Population Status and Trends

California

The population of VELB is steadily declining with the elimination of riparian habitat throughout its historical range. Less than 1% of the original riparian habitat remains, mostly distributed in small, isolated fragments (Collinge et al. 2001). The CNDDB (2003) lists 187 occurrences statewide.

In a statewide distribution study, Barr (1991) found 64 (27.8%) of the 230 sites she surveyed to have been recently occupied by VELB. In the Sacramento Valley region, Barr surveyed 79 sites and observed exit holes at 29 (36.7%) (Collinge et. al. 2001). In 1997, Collinge et. al. (2001) repeated Barr's methods at 65 sites in 14 watersheds; they found evidence of VELB occupancy at 30 (46.2%) of the 65 sites. Collinge et. al. (2001) observed turnover to occur only within occupied drainages, meaning that although adult VELB may colonize elderberry shrubs located within the drainage or watershed of the host plant, between-drainage dispersal is rare. The Holyoak Laboratory of the University of California, Davis, is currently collecting data from many of the same sites as those used by Barr (1991) and Collinge et al. (2001) in order to further study population trends of VELB in the Sacramento Valley (Talley pers. comm.).

Placer County Phase I Planning Area

The CNDDB (2003) lists seven VELB occurrences within the Phase I Planning Area.

Natural History

Habitat Requirements

Habitat for VELB consists of elderberry shrubs (*Sambucus* sp.) with a basal diameter greater than 2.54 centimeters (1 inch) occurring in riparian forests or elderberry savannas adjacent to riparian vegetation.

VELB utilizes two species of elderberry plants: blue elderberry (*Sambucus mexicana* Presl: Caprifoliaceaea) and red elderberry (*Sambucus racemosa* L. var. *microbotrys* [Rydb.] Kearney and Peebles: Caprifoliaceaea). VELB does not seem to select one species over the other, but colonizes whichever elderberry species is available (Barr 1991). Collinge et al. (2001) found that isolated elderberry plants and isolated drainages are less likely to support VELB populations than plants with connectivity to other habitat.

Individual VELB rely on the same elderberry plant (or clump of plants) throughout the life cycle. Adults feed on the elderberry leaves and flowers, mating pairs are typically observed on an elderberry shrub, eggs are laid on the stem or leaves of an elderberry plant, and the larval and pupal stages develop within the elderberry stem pith (Barr 1991; Talley pers. comm.).

Elderberry usually co-occurs with other woody riparian plants, including Fremont cottonwood (*Populus fremontii*), California sycamore (*Platanus racemosa*), various willows (*Salix* spp.), wild grape (*Vitis californica*), blackberry (*Rubus* spp.), and poison-oak (*Toxicodendron diversilobum*) (U.S. Fish and Wildlife Service 1984; Collinge et al 2001).

Reproduction

VELB adults are active during the flowering period of the host elderberry plant, usually from mid-March to mid-May (U.S. Fish and Wildlife Service 1984). The adults feed on the plants' leaves and flowers, and the females lay hundreds of eggs on the plant stems and leaves. Larvae emerge within a few days and burrow into the plant stem. The larva feeds downward through the stem pith, excavating a distinct feeding chamber filled with frass and shredded wood (Barr 1991). After 1–2 years, the larva chews a hole to the stem surface and returns to the chamber to pupate (Halstead and Oldham 1990). When the host plant begins to flower, the adult emerges and exits the chamber through a characteristic exit hole 0.4–1.0 centimeter (0.15–0.4 inch) in diameter (Barr 1991).

Dispersal Patterns

Hanks (1999) sorted longhorn beetles (Cerambycidae) into four categories based on the condition of the host plant at the time of colonization; these categories are *healthy host*, *weakened host*, *stressed host*, and *dead host*. According to Hanks's system, VELB appears to be a *weakened host* species. The implication of this designation is that emergent adults, which feed exclusively on elderberry foliage, have the option of completing the entire lifecycle on an individual host plant. The close proximity of the adult feeding/mating sites to the available oviposition sites results in a sedentary dispersal pattern.

Collinge et al. (2001) found that colonization of new elderberry sites within occupied drainages is rare and that dispersal between drainages probably does not occur at all. This pattern implies that even when an individual VELB disperses from its host plant to colonize new habitat, it will only travel along the riparian corridor within its home drainage. Most remaining elderberry habitat and riparian vegetation exist in small isolated patches; consequently, the distances between VELB populations and unoccupied VELB habitat limits the taxon's ability to successfully colonize new sites.

Longevity

VELB eggs hatch in approximately 3 days. The larval and pupal stages combined span 1or 2 years. Adult males live only for a few days, and adult females persist approximately 3–4 weeks. The great majority of a VELB's life span is spent within the stem of the host plant (Barr 1991; Collinge et al. 2001; Talley pers. comm.).

Sources of Mortality

Important sources of mortality include any action that might be detrimental to the health of the larval host plant. While feeding in the host plant, VELB larvae are vulnerable to such actions as pesticide application, trimming, dewatering, flooding, and Argentine ant invasion (Huxel 2000; Collinge et al. 2001; Talley pers. comm.).

Behavior

VELB larvae feed on the soft tissues of the elderberry plant's central pith (the vascular cells at the center of the plant stem). Larvae leave shredded wood and frass behind as they create feeding chambers in the stem. Pupae do not feed. The pupae undergo metamorphosis within an enlarged pupal chamber. Adult beetles feed on the nectaries, flowers, and leaves of the host plant or those of another elderberry plant close to the host plant. The emergence of the adult beetle from the elderberry stem creates a characteristic sub-oval exit hole 0.4 to 1.0 centimeter (0.15 to 0.4 inch) in diameter (Barr 1991; Colinge et al. 2001; Talley pers. comm.).

Ecological Relationships

VELB is a specialized herbivore that feeds exclusively on elderberry shrubs. The larval form is a nonlethal parasite on red and blue elderberry shrubs. The adult form is also a pollinator of red and blue elderberry shrubs.

Elderberry shrubs may be affected (directly or indirectly) by the stem-boring activity of VELB larvae. Arnold (1990) reported that 20% of elderberry shrubs examined that had more than two exit holes died from a fungal disease. Although this ecological relationship is not well documented for VELB, other longhorn beetles (Cerambycidae) have been shown to indirectly transport disease-causing fungi and bacteria between host plants (Hanks 1999).

Population Threats

The greatest threats to the persistence of VELB are habitat loss and fragmentation, flood management, pesticide and herbicide use, and exotic species invasion (U.S. Fish and Wildlife Service 1984; Huxel 2000; Collinge et al. 2001).

Urban and agricultural development, aggregate mining, and flood control practices (e.g., damming and channel maintenance) have damaged or eliminated a large percentage of the riparian forests that once occurred in California, reducing and fragmenting the available habitat for VELB (Barr 1991).

Invasion of the exotic Argentine ant (Linepithema humile Mayr) into riparian habitats may present a threat to the distribution and survival of VELB. Huxel (2000) surveyed 15 sites in the Putah Creek watershed and 15 sites in the American River watershed for presence of L. humile, native ant species, and VELB. Results of the Putah Creek survey showed the presence of Argentine ant to have a negative relationship with VELB presence and showed native ant species to have a positive relationship with VELB presence. Although results of the American River survey showed no significant relationships, Huxel observed that the invasion of Argentine ant into the American River watershed was relatively recent (<5 years).

Central Valley Fall-Run Chinook Salmon (*Oncorhynchus tshawytscha*)

Status

FEDERAL: Candidate (64 FR 50394, September 16, 1999).

STATE: None.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

North America

There are probably more than a thousand spawning populations of chinook salmon on the North American coast from southeastern Alaska to California (Healey 1991). Chinook salmon is one of the most abundant salmon species in North America.

California

The Central Valley fall-run evolutionarily significant unit (ESU) includes fall-run and late–fall-run chinook salmon in the Sacramento and San Joaquin Rivers and their tributaries.

Placer County Phase I Planning Area

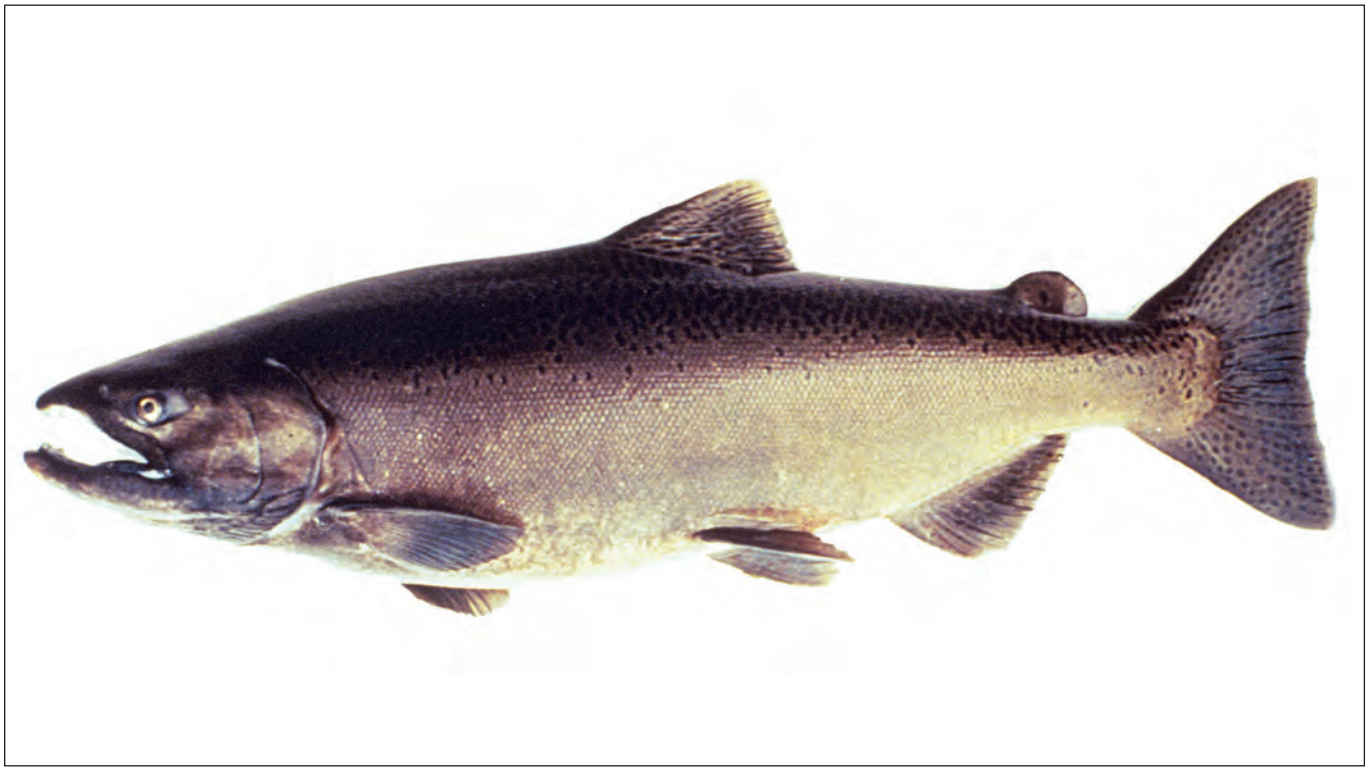
Historical

The Bear River supported a fall run that ascended as far as present day Camp Far West Reservoir, where a waterfall probably hindered further passage (Yoshiyama et al. 1996). In the 1950s, as many as a thousand salmon spawned in the Dry Creek system, about 10% of which used Miners Ravine (Department of Water Resources 2002).

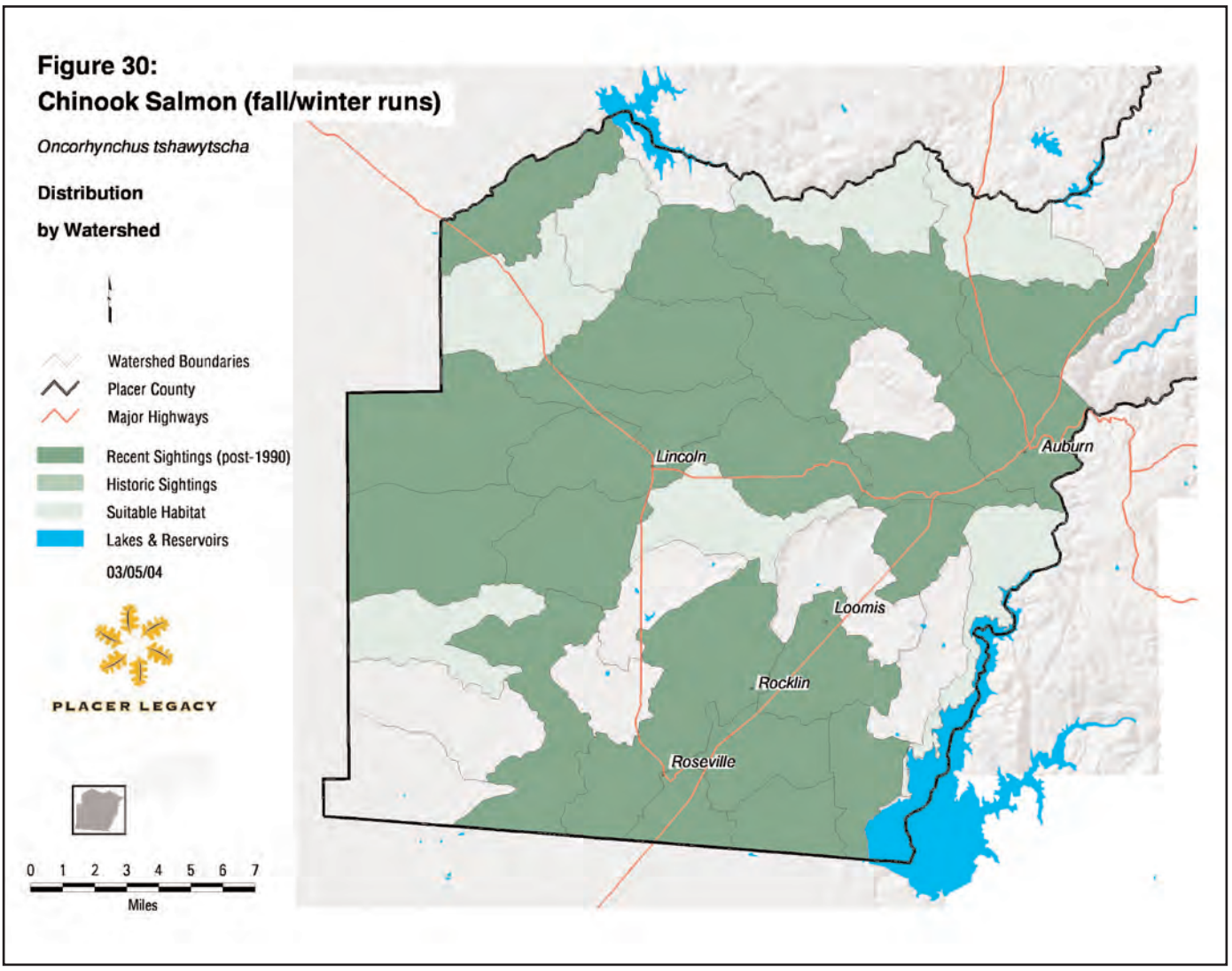
Current

The Bear River supports an occasional run of adult fall-run chinook salmon in years when flows are sufficient to provide passage (Yoshiyama et al. 1996).

Adult salmon carcasses have been observed in Antelope Creek, Miners Ravine, and Secret Ravine in late fall (CALFED Bay-Delta Program 2000). Chinook carcasses, redds, and live fish were identified each year in adult salmonid spawning surveys conducted during fall/winter 2000–2002 in Linda and Cirby Creeks in Roseville (Garcia and Associates 2002). During the 2001/2002 fall/winter, Garcia and Associates identified 30 adult fish, 19 redds, and 51 carcasses in Linda Creek. During the same period, the Dry Creek Conservancy observed 188 adult fish and 73 carcasses in the Dry Creek watershed and tributaries, excluding Linda Creek (Garcia and Associates 2002). The juvenile salmonid monitoring



Central Valley Fall-Run Chinook Salmon Photo by Richard Wydoski



conducted during February–May 2002 yielded four juvenile chinook salmon in Cirby and Linda Creeks (Garcia and Associates 2002).

In addition, wild juvenile fall-run chinook salmon have been found in small numbers in Coon Creek; Doty Creek; Auburn Ravine; and the upper creek watersheds of Dry Creek, including Secret Ravine, Antelope Creek, and Miners Ravine (CALFED Bay-Delta Program 2000; Figure 30).

It has been reported that juvenile salmon raised in the Feather and American River hatcheries have been stocked in several streams (including Coon Creek, Auburn Ravine, and tributaries of Dry Creek) since 1983, but there appear to be no records of the numbers and dates of these plants.

Population Status and Trends

California

The most abundant populations of fall-run chinook salmon occur in the Sacramento, Feather, Yuba, and American rivers (Mills and Fisher 1994). The ESU also occurs in smaller tributaries of the Sacramento River and in tributaries of the San Joaquin River. Fall-run chinook salmon have a relatively large hatchery component, averaging more than 25,000 adults. Natural spawners average about 200,000 adults for the Sacramento and San Joaquin River systems (Moyle 2002).

Placer County Phase I Planning Area

In 1963 and 1964, California Department of Fish and Game surveys indicated that 300–800 wild fall-run Chinook salmon spawned successfully in Secret Ravine. Hatchery juveniles from the Feather and American Rivers have been stocked in Coon Creek, Auburn Ravine, and tributaries of Dry Creek since 1983 (CALFED Bay-Delta Program 2000).

Natural History

Habitat Requirements

Chinook salmon are dependent on suitable water temperature and substrate for successful spawning and incubation. Although the suitability of gravel substrates for spawning depends largely on the fish size, a number of studies have determined substrate sizes that represent the most suitable conditions. Generally, chinook salmon require substrates of approximately 0.3–15 centimeters (cm) (0.1–5.9 inches), whereas steelhead prefer substrate no larger than 10 cm (Bjornn and Reiser 1991).

The quality of spawning habitat is also correlated with intra-gravel flow. Low intra-gravel flow may provide insufficient dissolved oxygen, contribute to growth of fungus and bacteria, and result in high levels of metabolic waste. High percentage of fines in gravel substrates can substantially limit intra-gravel flow, affecting the amount of spawning gravel available in the river (Healey 1991). Raleigh et al. (1986) concluded that optimal gravel conditions would include less than 5–10% fine sediments measuring 0.3 cm (0.12 inch) or less in diameter. In addition, alevins of chinook salmon, steelhead, and coho salmon have been observed in laboratory studies to have difficulty emerging when gravels exceeded 30–40% fine sediments (Bjornn 1968; Bjornn and Reiser 1991; Waters 1995).

Water depth is another factor affecting spawning gravel selection (Raleigh et al. 1986; Bjornn and Reiser 1991). Minimum water depths at redd areas vary with fish size and water velocity, because these variables affect the depth necessary for successful digging (Healey 1991). In general, water depth should be at least deep enough to cover the fish during spawning. Burner (1951 in Healey 1991 and Bjornn and Reiser 1991) observed chinook salmon spawning in water as shallow as 5 cm (0.16 foot); Vronski (1972 in Healey 1991) found chinook salmon spawning in water depths of 720 cm (23.6 feet). Thompson (1972 in Bjornn and Reiser 1991), who also studied water depth requirements for spawning, found chinook salmon spawning in depths less than 24 cm (9.4 inches).

Flow velocity also affects spawning gravel selection; however, the range in water depth and velocity is very broad (Healey 1991). Healey found water velocities of 30–189 cm/second (0.98–6.2 feet/second) reported in the literature. Studies in northern California found that chinook salmon from the Yuba and Sacramento Rivers preferred velocities of 47.2–89.9 cm/second (1.55–2.95 feet/second) and 27.4–82.3 cm/second (0.9–2.7 feet/second) respectively (California Department of Fish and Game 1991).

Survival of chinook salmon eggs and larvae during incubation declines as water temperatures increase to 12–16°C (53.6–60.8°F) (Myrick and Cech 2001).

Rearing habitat for salmonids is defined by environmental conditions such as water temperature, dissolved oxygen, turbidity, substrate, area, water velocity, water depth, and cover (Bjornn and Reiser 1991; Healey 1991; Jackson 1992). Environmental conditions and interactions among individuals, predators, competitors, and food sources determine habitat quantity and quality and the productivity of the stream (Bjornn and Reiser 1991). Rearing habitat for juvenile chinook salmon includes riffles, runs, pools, and inundated floodplain.

Use of floodplain habitat by juvenile chinook salmon has been well documented (California Department of Water Resources 1999; Sommer et al. 2001). Sommer et al. (2001) found that floodplain habitat provides better rearing and migration habitat for juvenile salmon than does the main river channel. The growth rate of chinook salmon in the Yolo bypass was generally higher than the growth rate in the main channel of the Sacramento River. The faster growth rate in the Yolo Bypass may be attributed to increased prey consumption associated with greater availability of drift invertebrates and warmer water temperatures. Invertebrate production on the floodplain may be stimulated by availability of detritus in the food web, available habitat for benthic invertebrates, and a relatively long hydraulic residence time. Long residence time reduces the rate at which nutrients and drifting invertebrates are flushed out of the system.

Survival of juvenile chinook salmon declines as water temperatures increase to 18–24°C (64.4–75.2°F). Juveniles require cooler water temperature to complete the parr-smolt transformation and to maximize their saltwater survival. Successful smolt transformation deteriorates at temperatures of 17–23°C (62.6–73.4°F).

Reproduction

Central Valley fall-run chinook salmon spawn from late September to December, with peak spawning taking place during late October and November when water temperatures decrease (Moyle 2002). Fall-run chinook salmon spawn over gravel (redds) soon after arriving at their spawning grounds. Juvenile fish remain in redds from about 32 days at 16°C (61°F) to 159 days at 3°C (37°F) (Healey 1991).

Dispersal Patterns

After emerging from gravel, juvenile chinook salmon move downstream, mostly at night. They rear in the mainstem rivers or the Delta before migrating to the ocean.

Longevity

Chinook salmon generally mature at 3–4 years and can reach 5–8 years (Healey 1991). A minority of individuals return to the river as sexually mature 2-year-olds (grilse).

Sources of Mortality

Low flows, resulting in warmer water temperatures and decreased dissolved oxygen levels, increase mortality of eggs and juvenile chinook salmon. Egg survival is reduced when elevated water temperatures reduce oxygen availability in the gravel. Another result of increased temperatures is the threat of heightened predation by nonnative fish species; sublethal temperatures reduce growth of juvenile salmon and may increase potential predators' metabolism, thus increasing the risk of predation by centrarchids and other nonnative fish species adapted to higher water temperatures. (U.S. Army Corps of Engineers 2000.)

Entrainment at diversions is another source of mortality; low flows can confuse or detain migrating juveniles, resulting in higher entrainment at diversions.

Behavior

While in streams, chinook salmon are opportunistic feeders and vary their diet according to seasonal availability. In the summer months, they feed primarily on drifting aquatic invertebrates, terrestrial insects, and active bottom invertebrates. Individual fish, however, do not usually feed on the full range of food available. Larger fish tend to eat larger prey. Feeding can occur any time of day, but most activity occurs around dusk. (Moyle 2002.)

After migrating to the ocean, chinook salmon feed on estuarine invertebrates and krill. As the juvenile salmon grow, other fish constitute an increasing component of their diet. Chinook salmon's large size and rapid growth in the ocean can be attributed to a diet of fish, squid, and crustaceans. Once they return to fresh water, adult chinook salmon stop feeding. (Moyle 2002.)

Chinook salmon occupy the freshwater system from the estuary to stream headwaters, depending on access, water temperature, and perennial flow. The distance that Central Valley fall-run chinook salmon migrate in the ocean is unknown.

Movement and Migratory Patterns

Fall-run chinook salmon migrate from the Pacific Ocean to the Central Valley rivers from approximately August to December. Peak spawning occurs during late October and November. Juvenile fall-run chinook salmon start emigrating towards the Pacific Ocean from January through June, shortly after emerging from their redds.

EcologicalRelationships

The predator/prey relationship between juvenile chinook salmon and nonnative fish species has a significant effect on mortality of young salmon. Warm water temperatures cause stress and suppress growth; both conditions increase vulnerability to predators. Moreover, because nonnative fish are adapted to warmer water temperatures, their predatory efficiency is increased by the same conditions that heighten the vulnerability of juvenile chinook salmon.

Population Threats

Degradation and loss of habitat have contributed substantially to the decline of chinook salmon. Shasta and other dams blocked access to historic spawning and rearing habitat, as they did in the case of steelhead. Other factors affecting abundance include modifications of water temperatures resulting from reservoir operations, harvest, entrainment in diversions, contaminants, predation by nonnative species, and interaction with hatchery stock (U.S. Army Corps of Engineers 2000).

Low flows limit habitat area and adversely affect water quality by elevating water temperatures and depressing dissolved oxygen; these conditions stress incubating eggs and rearing juveniles. Low flows may affect migration of juvenile and adult salmon; decreased depths can inhibit adult passage, and reduced velocity can impede the downstream movement of juveniles. Low flows in combination with diversions may result in higher entrainment losses. (U.S. Army Corps of Engineers 2000.)

In the Delta, flows drawn through the Delta Cross Channel (DCC) and Georgiana Slough transport a proportion of migrants into the central Delta. The number of juveniles entering the DCC and Georgiana Slough is assumed to be proportional to the volume of flow diverted from the Sacramento River (California Department of Fish and Game 1987b). Survival of juvenile chinook salmon drawn into the central Delta is lower than survival of juvenile chinook salmon remaining in the Sacramento River channel.

Central Valley Steelhead (*Oncorhynchus mykiss*)

Status

FEDERAL: Threatened (63 FR 13347, March 19, 1998).

STATE: Species of Special Concern.

OTHER: Critical habitat designated February 16, 2000 (65 FR 7764); withdrawn April 2002.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

North America

Current distribution of steelhead ranges from southern California to the Kuskokwim drainages near the Alaska Range. A number of known distinct populations occur from Canada to southern California.

California

The Central Valley evolutionarily significant unit (ESU) of steelhead includes the Sacramento and San Joaquin Rivers, along with all of their tributaries. The Sacramento–San Joaquin steelhead run tends primarily to use tributaries of the Sacramento, Feather, Yuba, and lower American River drainages. San Joaquin drainage populations have been reduced to a few small, isolated wild runs and some hatchery stock.

Placer County Phase I Planning Area

Historical

Historical data not are available for the Phase I Planning Area.

Current

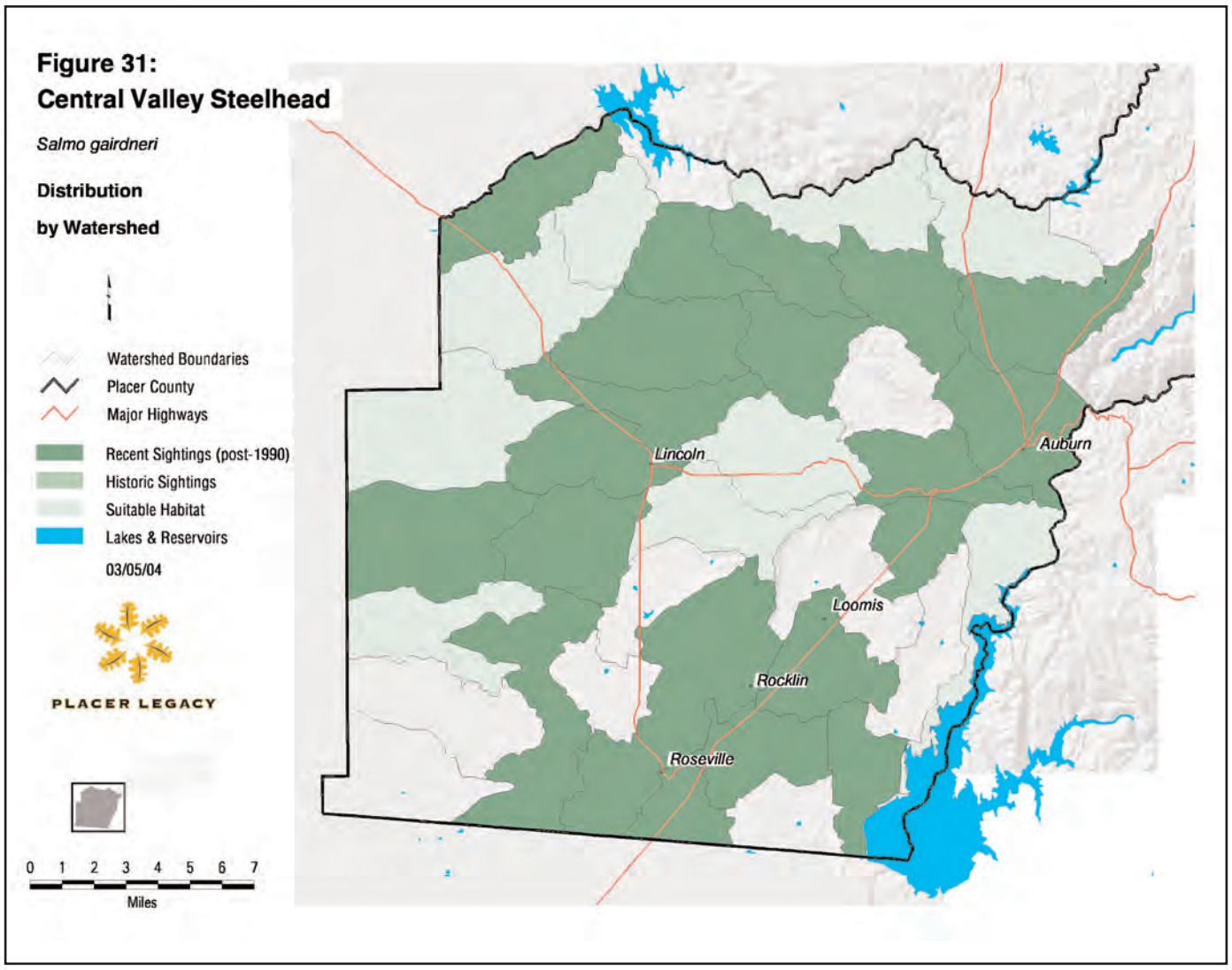
Rainbow trout/steelhead adults and fry have been seen in Coon Creek, Auburn Ravine, Dry Creek, Secret Ravine, and Miners Ravine (CALFED Bay-Delta Program 2000; Figure 31). Very little salmonid sampling has occurred in Placer County streams; however, during 1998–1999, the California Department of Fish and Game's (DFG's) Stream Evaluation Program conducted reconnaissance surveys of juvenile salmonid presence/absence in Secret Ravine Creek. This study identified hundreds of emigrating salmon and many steelhead smolts.

Population Status and Trends

California

Central Valley steelhead are now restricted to the Sacramento River downstream of Keswick Dam; the lower reaches of the Feather River, American River, and other large tributaries downstream of impassable dams; small, perennial tributaries of the Sacramento River; and the Sacramento–San Joaquin Delta (Delta). The population abundance of Central Valley steelhead, as measured by the number of adults returning to spawn, is unknown for natural spawners. Natural escapement in 1995 was estimated to be about 1,000 adults for Mill and Deer Creeks and about 1,000 for the Yuba River (Cramer and Associates 1995). Hatchery returns have

Central Valley Steelhead Photo by Jeff Kozlowski



averaged approximately 10,000 adults (Mills and Fisher 1994). Run size has averaged less than 2,500 fish (excluding hatchery stock) since 1980 (California Department of Fish and Game 1994).

Placer County Phase I Planning Area

Very few sampling efforts have been conducted in Placer County streams. Juvenile steelhead have been identified by DFG in Secret and Miners Ravines. Other monitoring studies have identified steelhead in Cirby and Linda Creeks, which are tributaries of Dry Creek. Auburn Ravine, Coon Creek, and their tributaries likely provide habitat for steelhead during some part of their life history when sufficient flows occur.

Natural History

Habitat Requirements

Steelhead are dependent on suitable water temperature and substrate for successful spawning and incubation. Although the suitability of gravel substrate for spawning depends largely on the fish size, a number of studies have determined substrate ranges that represent the most suitable conditions. Generally, steelhead prefer substrates no larger than 10 centimeters (cm) (3.9 inches) (Bjornn and Reiser 1991).

The quality of spawning habitat is also correlated with intra-gravel flow. Low intra-gravel flow may provide insufficient dissolved oxygen, contribute to growth of fungus and bacteria, and result in high levels of metabolic waste. High percentage of fines in gravel substrates can substantially limit intra-gravel flow, affecting the amount of spawning gravel available in the river (Healey 1991). Raleigh et al. (1986) concluded that

optimal gravel conditions would include less than 5–10% fine sediments measuring 0.3 cm (0.12 inch) or less in diameter. In addition, alevins of Chinook salmon, steelhead, and coho salmon have been observed in laboratory studies to have difficulty emerging when gravels exceeded 30–40% fine sediments (Bjornn 1968; Phillips et al. 1975 in Bjornn and Reiser 1991; Waters 1995).

Water depth criteria vary widely; there is little agreement among studies about the minimum and maximum values for depth (Healey 1991). Salmonids will spawn in water depths that range from a few centimeters to more than a meter. A minimum depth of 25 cm (9.8 inches) for steelhead spawning has been widely used in the literature and is within the range observed in some Central Valley rivers (California Department of Fish and Game 1991). Minimum water depth for steelhead spawning has been observed to be at least deep enough to cover the fish (Bjornn and Reiser 1991). Many fish spawn in deeper water.

Conditions supporting steelhead spawning and incubation are assumed to deteriorate as temperature warms to 11–15°C (52–59°F) (Myrick and Cech 2001). Steelhead eggs that are subjected to temperatures warmer than 15°C (59°F) are prone to increased mortality.

Rearing habitat for salmonids is defined by environmental conditions such as water temperature, dissolved oxygen, turbidity, substrate, area, water velocity, water depth, and cover (Bjornn and Reiser 1991; Healey 1991; Jackson 1992). Environmental conditions and interactions among individuals, predators, competitors, and food sources determine habitat quantity and quality and the productivity of the stream (Bjornn and Reiser 1991). Everest and Chapman (1972) found juvenile Chinook salmon and steelhead of the same size utilizing similar in-channel rearing areas. Juvenile steelhead are year-round residents; they generally use riffles and runs in the main and secondary channels along with the head and tail of pools. Shallow riffles are the most important channel type for steelhead during their first year (Barnhart 1986). Steelhead also use seasonal habitats of intermittent streams for rearing (McEwan 2001). Floodplain habitat does not appear to provide significant rearing habitat for steelhead.

Water velocity is of particular importance in determining where juvenile salmonids occur because it determines the energetic requirements for maintaining position and the amount of food delivered to a particular location. Juvenile salmonids tend to select positions that maximize energetic gain, but these positions can be altered by interactions with other fish and the presence of cover (Shirvell 1990). The water velocity preferred by chinook salmon varies with size of the fish; larger fish occupy higher velocity and deeper areas than smaller fish, potentially gaining access to abundant food and avoiding predatory birds (Bjornn and Reiser 1991; Jackson 1992). Griffith (1972 in Raleigh et al. 1984) found water velocities of 0.10–0.22 meter per second (0.32–0.72 foot per second) to be associated with occurrence of rainbow trout. Sheppard and Johnson (1985) found similar results for juvenile steelhead; they measured velocities of 0.12–0.24 meter per second (0.40–0.80 foot per second). Bovee (1978 in California Department of Fish and Game 1991) presented water velocities of 0.18–0.37 meter per second (0.6 –1.2 feet per second) as having a suitability index of 1 for juvenile rainbow trout and steelhead.

Stream substrates are important for juvenile salmonids, particularly fry; and for production of aquatic invertebrates that are food for salmonids (Bjornn and Reiser 1991). Waters (1995) and Bjornn and Reiser (1991) indicated the importance of interstitial space in riffles in influencing fry density and stream carrying capacity. The summer or winter carrying capacity of a stream declined when fine sediments filled the interstitial spaces of the substrate.

Juvenile salmonids occur over a wide variety of substrates; substrate does not appear to be a critical criterion determining rearing area selection. Baltz et al. (1987 in Jackson 1992) found that temperature was a better predictor of habitat utilization by rainbow trout and other native fish species than mean water velocity and substrate. Hampton (1988 in Jackson 1992) found that substrate was an important factor for rearing when large cobbles or boulders were used as velocity shelters in riffles and runs. This adaptation increases energetic gain or minimizes energy expenditure.

Instream and overhead cover (e.g., undercut banks, downed trees, and overhanging tree branches) are important for juvenile rearing. The addition of cover increases spatial complexity and may increase productivity. The abundance of food and the occurrence of competitors and predators determine cover value. Fine-textured instream woody material provides the hydraulic diversity necessary for selection of suitable velocities, access to drifting food, and escape refugia from predatory fish. An area of cover greater than 15% of the total habitat area may be adequate for juvenile salmonids (Raleigh et al. 1984).

Juvenile rearing success is assumed to deteriorate at water temperatures of 17–25°C (62.6–77°F) (Raleigh et al. 1984; Myrick and Cech 2001). Smolt transformation requires cooler temperatures than rearing; successful transformation occurs at temperatures of 6–10°C (42.8–50°F). Juvenile steelhead, however, are observed to migrate through the Delta at water temperatures substantially warmer than 13°C (55°F). Juvenile steelhead have been captured at Chipps Island in June and July and at water temperatures exceeding 20°C (68°F) (Nobriega and Cadrett 2001).

Successful adult migration and holding is assumed to deteriorate as water temperature warms to 14–21°C (52–69.8°F). Adult steelhead appear to be much more sensitive than juveniles to thermal extremes (McCullough 1999).

Reproduction

Spawning in the Sacramento River basin typically occurs from late December through April, with most adults spawning in January through March. Individual steelhead may spawn more than once, returning to the ocean between each spawning migration.

Steelhead spawn in relatively clean, cool (<13.9°C [57°F]) water; they build their redds and lay their eggs in clean gravel at the head of riffles. The eggs hatch 19–80 days after spawning, depending on water temperature. Larvae remain in the gravel for several weeks before emerging as fry. (National Marine Fisheries Service 1998.)

Dispersal Patterns

Adult Central Valley steelhead migrate upstream from the ocean July through May; most migrate after October and before May. Juvenile migration to the ocean generally occurs from November through May. Based on salvage data at the state and federal export facilities in the Delta, the peak months of juvenile migration appear to be March and April. After 2–3 years of ocean residence, adult steelhead return to their natal stream to spawn as 4- or 5-year-olds. (National Marine Fisheries Service 1998.)

Juvenile steelhead rear a minimum of 1 and typically 2 or more years in fresh water before migrating to the ocean following smoltification (e.g., the process of physiological change that allows ocean survival).

Longevity

Although such longevity is uncommon, steelhead may reach 9 years of age. An individual, following 2–3 years in freshwater and an additional 1–3 years in saltwater, may return to spawn several times, often missing alternate years. Many early spawning individuals do not survive beyond their first upstream migration. (National Marine Fisheries Service 1999; Moyle 2002).

Sources of Mortality

Impassable dams block access to most of the historical headwater spawning and rearing habitat of Central Valley steelhead. In addition, much of the remaining accessible spawning and rearing habitat is severely degraded by elevated water temperatures, agricultural and municipal water diversions, unscreened and poorly screened water intakes, restricted and regulated streamflows, levee and bank stabilization projects, and poor quality and quantity of riparian and shaded riverine aquatic (SRA) cover (Busby *et al.* 1996). Low flows, resulting in warmer water temperatures and decreased dissolved oxygen levels, increase mortality of eggs and juvenile steelhead. Another result of increased temperatures is the threat of heightened predation by nonnative fish species; sub-lethal temperatures reduce growth of juvenile steelhead and may increase potential predators' metabolism, thus increasing the risk of predation by centrarchids and other nonnative fish species adapted to higher water temperatures.

Reynolds et al. (1993) reported that 95% of salmonid habitat in California's Central Valley has been lost, largely as a result of mining and water development activities. They also noted that declines in Central Valley steelhead stocks are due primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment at diversions. Low flows can confuse or detain migrating juveniles, resulting in higher entrainment at diversions.

Behavior

While in streams, steelhead are opportunistic feeders and vary their diet according to seasonal availability. In the summer months, they feed primarily on drifting aquatic invertebrates, terrestrial insects, and active bottom invertebrates. Individual fish, however, do not usually feed on the full range of food available. Larger fish tend to eat larger prey. Feeding can occur any time of day, but most activity occurs around dusk. (Moyle 2002.)

After migrating to the ocean, steelhead feed on estuarine invertebrates and krill. As the juvenile steelhead grow, other fish constitute an increasing component of their diet. Steelhead's large size and rapid growth in the ocean can be attributed to a diet of fish, squid, and crustaceans. Adult steelhead in streams feed opportunistically, but it is not uncommon for them to stop eating for periods of time. (Moyle 2002.)

Steelhead occupy the freshwater system from the estuary to stream headwaters, depending on access, water temperature, and perennial flow. The distance that Central Valley steelhead migrate in the ocean is unknown.

Movement and Migratory Patterns

Steelhead migrate from the Pacific Ocean to the Central Valley rivers from approximately November through May. Peak spawning occurs January–March. Juvenile steelhead emigration varies according to how long they rear in their natal streams, but peak migration occurs in March and April. (National Marine Fisheries Service 1998.)

Ecological Relationships

The predator/prey relationship between juvenile steelhead and nonnative fish species has a significant effect on mortality of young steelhead. Warm water temperatures cause stress and suppress growth; both conditions increase vulnerability to predators. Moreover, because nonnative fish are adapted to warmer water temperatures, their predatory efficiency is increased by the same condition that heightens the vulnerability of juvenile steelhead.

Population Threats

The decline in steelhead populations is attributable to changes in habitat quality and quantity. The availability of steelhead habitat in the Central Valley has been reduced by as much as 95% or more by barriers to movement (i.e., dams). Other factors contributing to the decline of steelhead in the Central Valley include mining; agriculture; urbanization; logging; harvest; hatchery influences; and flow management, including reservoir operations, hydropower generation, and water diversion and extraction. (National Marine Fisheries Service 1996.)

In the Sacramento River and its major tributaries, the operation of the Central Valley Project and State Water Project controls the river flow. Low flows limit habitat area and adversely affect water quality by elevating water temperatures and depressing dissolved oxygen; these conditions stress incubating eggs and rearing juvenile steelhead. Low flows may affect migration of juvenile and adult steelhead; decreased depths can inhibit adult passage, and reduced velocity can impede the downstream movement of juveniles. Low flows in combination with diversions may result in higher entrainment losses (U.S. Army Corps of Engineers 2000).

In Placer County aquatic systems, sewage outfalls, lack of riparian cover, thermal pollution, and nonnative predators may all adversely affect the likelihood a healthy fishery. Below Roseville, Dry Creek steelhead must migrate past two sewage outfalls and a 10-kilometer (6.2-mile) stretch of channelized, nearly stagnant backwater in order to either spawn or outmigrate. The Coon and Auburn Creek systems pose similar difficulties for fisheries, with at least 10 kilometers (6.2 miles) of slough water to traverse. Other factors that may limit steelhead success in Placer County creeks include low fall/winter flows, excessive sediment, increased stormwater runoff, channel bank erosion, migration barriers, and elevated water temperatures.

Foothill Yellow-legged Frog (*Rana boylii*)

Status

FEDERAL: None.

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: No.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

Foothill yellow-legged frogs are endemic to California. Historically, they occurred from west of the crest of the Cascade Ranges in Oregon south to the Transverse Ranges in Los Angeles County, and in the Sierra Nevada foothills south to Kern County (Zweifel 1955; Stebbins 1985). An isolated population was reported in Sierra San Pedro Martir, Baja California (Loomis 1965). The current range excludes coastal areas south of northern San Luis Obispo County and foothill areas south of Fresno County, where the species is apparently extirpated (Jennings and Hayes 1994). The species can occur from sea level to 1,830 meters (6,000) feet (Stebbins 1985).

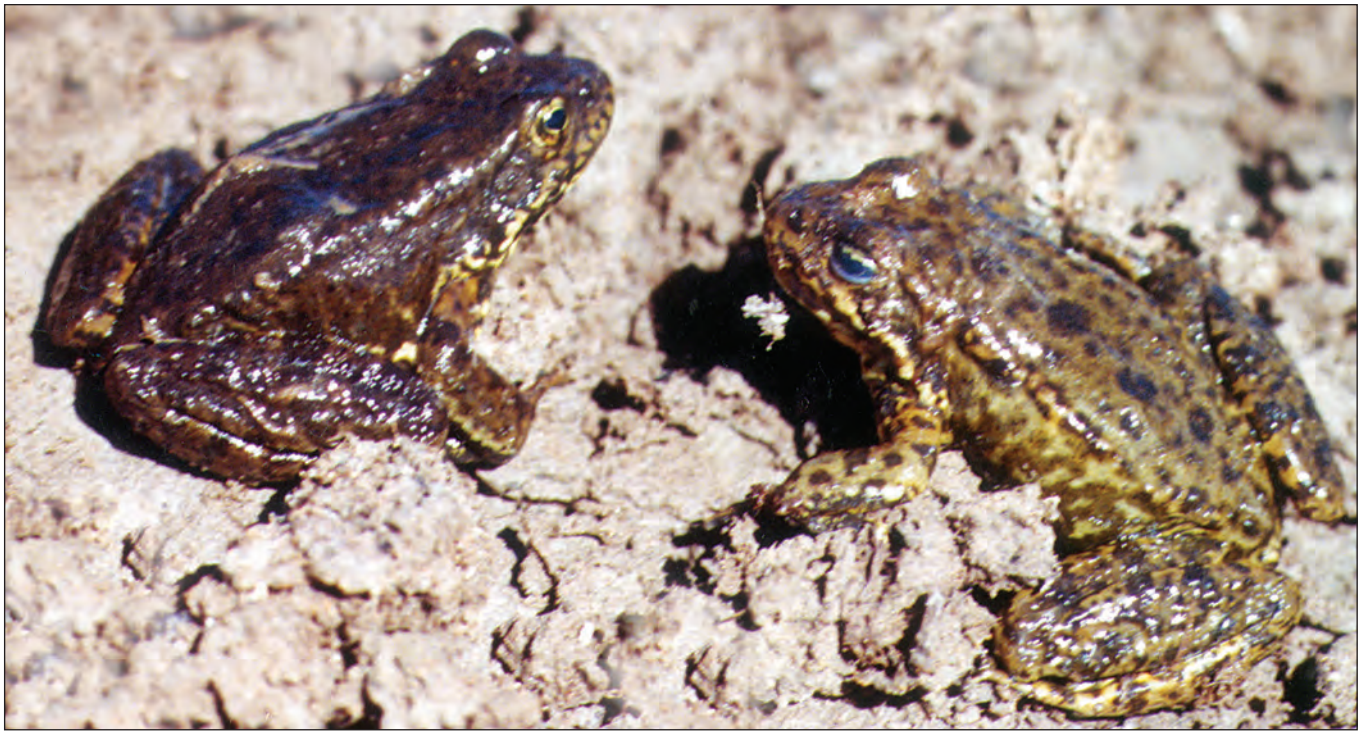
Placer County Phase I Planning Area

Historical

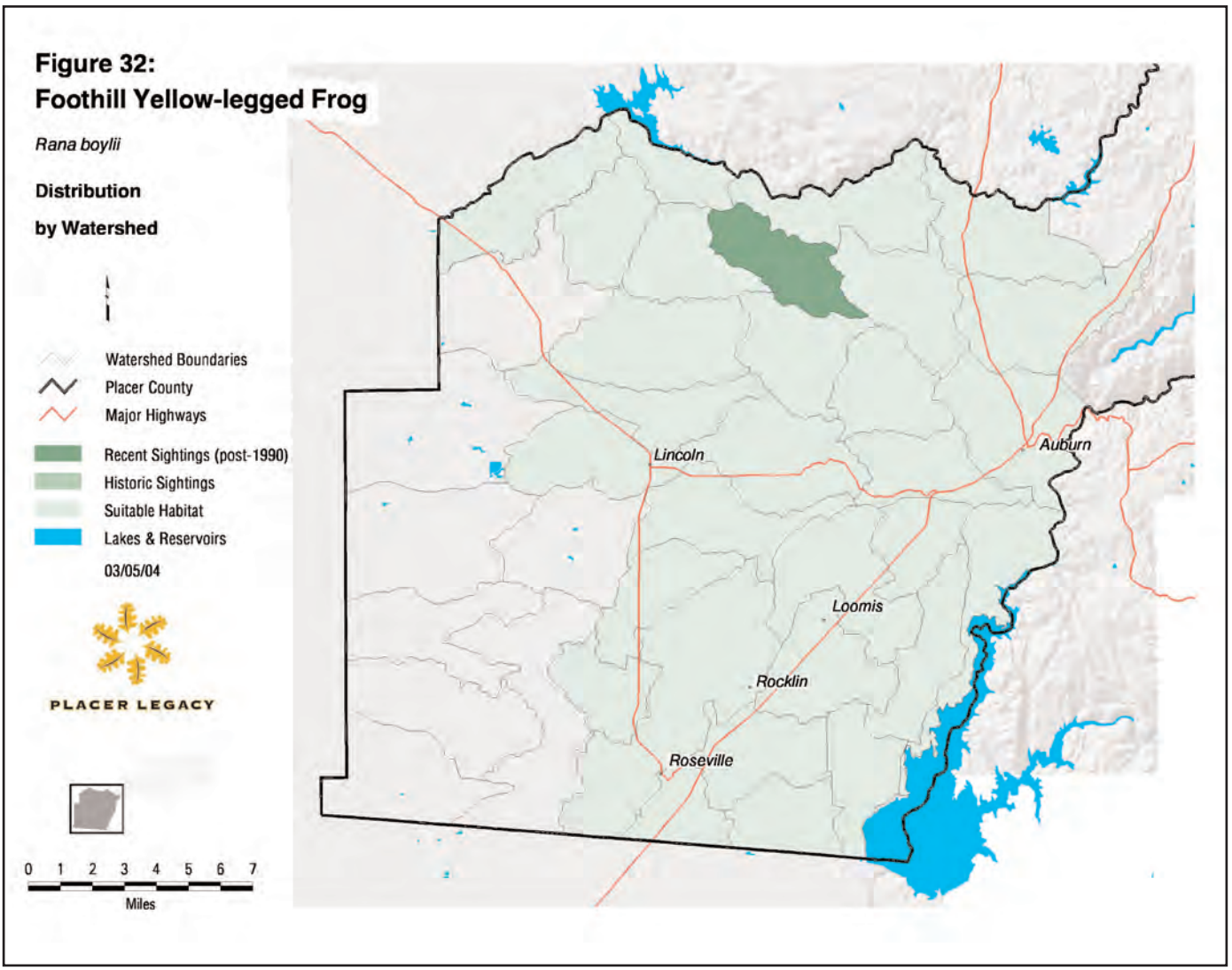
There is very little information on the historical occurrence of foothill yellow-legged frog in Placer County. Jennings and Hayes (1994) showed several records for foothill yellow-legged frogs in the foothill areas of Placer County.

Current

Jennings and Hayes (1994) reported that foothill yellow-legged frogs are widely scattered on the western slope of the northern Sierra Nevada and considered them threatened in this area. Brian Williams (pers. comm.) conducted surveys for foothill yellow-legged frogs in 2002 along tributaries of the American River and believes that foothill yellow-legged frogs are likely to be widespread throughout the foothill portions of Placer County. There are five recent records (1994–2000) for this species in the central portion of Placer County outside the Phase I Planning Area (California Natural Diversity Database 2003; Figure 32). Four of these records are 15 or more miles from the northeastern boundary of the Phase I Planning Area; one record is approximately 5 miles from the boundary.



Foothill Yellow-legged Frog Jones & Stokes file photo



Population Status and Trends

California

Foothill yellow-legged frogs have become rare in the west slope drainages of the Sierra Nevada and southern Cascade Ranges east of the Sacramento–San Joaquin Valleys. They have not been observed since the mid-1970s at 19 historical localities on the western slope of the southern Sierra Nevada (Jennings and Hayes 1994).

The species is extremely rare in California south of the Salinas River. The last reliable observation of a foothill yellow-legged frog in this region was in 1970 (Jennings and Hayes 1994). In the Coast Ranges north of the Salinas River the species still occurs at many locations but is subject to several risk factors (Jennings and Hayes 1994) that could threaten these populations (see *Population Threats* below).

Populations of foothill yellow-legged frog in Oregon also appear to be declining (Borisenko and Hayes 1999). Foothill yellow-legged frogs were absent from at least 55% of 90 historical locations in Oregon that were surveyed by Borisenko and Hayes (1999). Recent information is not available on the status of populations in Baja California.

Placer County Phase I Planning Area

Foothill yellow-legged frogs may have occurred historically in the eastern portion of the Phase I Planning Area near Auburn; however, these populations were determined to be extirpated (Jennings and Hayes 1994). There are no recent records for foothill yellow-legged frogs in this portion of the county (Jennings and Hayes 1994; California Natural Diversity Database 2003).

Natural History

Habitat Requirements

Foothill yellow-legged frogs occupy rocky streams in valley-foothill hardwood, valley-foothill hardwood-conifer, valley-foothill riparian, ponderosa pine, mixed-conifer, coastal scrub, mixed chaparral, and wet meadow habitat types (Zeiner et al. 1988). They are nearly always found within a few feet of water. Foothill yellow-legged frogs are frequently found in moving but not swiftly flowing water (Stebbins 1954). The species is most common along streams with rocky bottoms but has also been found along streams with mud bottoms (Stebbins 1951). Foothill yellow-legged frogs require permanent streams or, at a minimum, streams where pools persist through the dry season (Stebbins 1951).

Reproduction

Foothill yellow-legged frogs breed from mid-March to May after the high-water stage in streams has passed and less sediment is being conveyed (Stebbins 1954). Eggs have been observed in early and mid-May in streams in southern California, indicating that oviposition occurs later in the south than in the north (Stebbins 1951). Eggs are deposited in clusters near margins of steams in shallow water. The egg clusters are attached to stones, vegetation, or the bank itself (Stebbins 1954.) Tadpoles metamorphose in approximately 3–4 months (Storer 1925; Stebbins 1951).

Dispersal Patterns

Information on demographics and dispersal patterns could not be found.

Longevity

Adult size is attained in 2 years (Storer 1925), but no longevity data are available for this species (Jennings and Hayes 1994). Other anurans have been reported to live 6–36 years in captivity (Duellman and Trueb 1986).

Sources of Mortality

Foothill yellow-legged frogs are preyed upon by garter snakes, fish, birds, and mammals (see Ecological Relationships below). In addition, bullfrogs are known to feed on native ranid larvae (Jennings 1996) and are likely to feed on foothill yellow-legged frog larvae where they co-occur (Moyle 1973). Other sources of mortality include desiccation from drought or unnatural fluctuations in flow releases (Moyle 1973; Kupferberg 1996); scouring of egg masses from floods or dam releases (Lind et al. 1996; Kupferberg 1996); urbanization; habitat alteration (Jennings 1996); and pesticides (Davidson et al. 2002).

Behavior

Foothill yellow-legged frogs are active from late February or early March through summer and into fall. The beginning of seasonal activity appears to correspond with the warming of streams to suitable temperatures. It is not known where these frogs spend the winter but it is thought that they remain close to streams. Few foothill yellow-legged frogs have been observed in hibernation areas away from streams. (Zweifel 1955.)

Normal home ranges of foothill yellow-legged frogs are probably not more than 10 meters (33 feet) in the longest dimension (Zeiner et al. 1988). If they follow the pattern of other ranid frogs, males probably defend areas around themselves during the breeding season (Martof 1953; Emlen 1968).

Foothill yellow-legged frogs eat aquatic and terrestrial arthropods, particularly insects. Insects found in the stomachs of these frogs include grasshoppers, hornets, carpenter ants, water striders, small beetles, and dipterans (mosquitoes and others). (Storer 1925; Stebbins 1951.)

Movement and Migratory Patterns

Little information is available regarding movement patterns and the distances foothill yellow-legged frogs will travel. Twitty et al. (1967) observed that newly metamorphosed foothill yellow-legged frogs consistently moved upstream during fall and winter over a 3-year period. During periods of high water conditions, foothill yellow-legged frogs may make occasional long-distance movements (up to 165 feet [50 meters]) (Zeiner et al. 1988).

Ecological Relationships

Bullfrogs are known to prey on foothill yellow-legged frogs (see *Population Threats* below). Several species of garter snakes, including red-sided garter snake (*Thamnophis sirtalis parietalis*), western terrestrial garter snake (*T. elegans*), and Oregon garter snake (*T. couchii hydrophilus*), are predators of posthatching stages of foothill yellow-legged frogs (Zweifel 1955; Jennings and Hayes 1994). Oregon garter snakes have been observed to feed more frequently on tadpoles,

whereas the other two species have been observed to feed more frequently on postmetamorphic individuals (Jennings and Hayes 1994). Rough-skinned newts (*Taricha granulosa*) have been observed preying on foothill yellow-legged frog eggs (Evenden 1948). In addition, when Centrarchid fishes were offered *Rana* tadpoles and eggs, they ate them readily (Werschkul and Christensen 1977). Fish, mammals (e.g., raccoons), and birds are likely to prey on one or more stages of foothill yellow-legged frogs (Zweifel 1955).

Foothill yellow-legged frogs coexist with Cascades frogs (*Rana cascadae*) and red-legged frogs (*Rana aurora*) at some localities; however, different microhabitat preferences may limit competition (Zeiner et al. 1988).

Population Threats

Habitat Alteration and Degradation

Tremendous population growth and the resulting urbanization of California since World War II have had devastating effects on native ranids in California. Controlling water flow, building roads into natural areas, and polluting waterways are examples of human activities that have modified and degraded amphibian habitat (Jennings 1988.) In areas where human activities have greatly altered habitats, amphibian populations have declined or been eliminated (Davidson et al. 2002).

Drought, Flooding, and Water Management

Poor timing and high-flow releases of water from upstream reservoirs have contributed to the decline of foothill yellow-legged frogs by scouring eggs from their oviposition substrates (Kupferberg 1996; Lind et al. 1996). Eggs are washed away when streams experience high-flow velocities for several days in a row (Kupferberg 1996). Decreased water flows can also have a negative result by stranding eggs or forcing foothill yellow-legged frogs into permanent pools where they may be more susceptible to predation (Moyle 1973; Kupferberg 1996). Results of surveys in Oregon by Borisenko and Hayes (1999) suggest a direct correlation between the absence of frogs and the presence of dams and grazing activities. Large dams substantially alter habitat by changing the hydrology and geomorphology of the water system, resulting in degraded habitat for foothill yellow-legged frogs (Borisenko and Hayes 1999).

Introduction of Nonnative Predators

At least 60 species of fish, many of them predatory, have been introduced in western North America over the past 120 years (Jennings 1988). Fish are known to prey upon various life stages of anurans (Hayes and Jennings 1986) and have been implicated in the decline of populations of frogs in some areas (Cory 1963; Knapp and Mathews 2000). Locations where introduced fishes were abundant contained few foothill yellow-legged frogs, indicating that the presence of these fish had a negative influence on the abundance of the frogs (Moyle 1973; Hayes and Jennings 1986).

Bullfrogs were introduced to California in the late 1800s (Hayes and Jennings 1986) and have spread throughout the state (Bury and Luckenbach 1976). Several observations provide evidence that bullfrogs prey upon foothill yellow-legged frogs: yellow-legged frog abundance was inversely correlated with bullfrog abundance, bullfrogs occupied areas that once had yellow-legged frogs, and captive bullfrogs ate yellow-legged frogs soon after the two species were placed together (Moyle 1973; Hayes and Jennings 1986). In Oregon, foothill yellow-legged frogs were rarely found co-occurring with bullfrogs (Borisenko and Hayes 1999). The alteration of foothill streams by human activities has increased the amount of suitable habitat for bullfrogs, which compete with and prey on foothill yellow-legged frogs (Moyle 1973; Borisenko and Hayes 1999).

Pesticides and Herbicides

Researchers have found that amphibians absorb pesticides in aquatic and terrestrial systems. Pesticides, herbicides, and other toxins are known to be harmful to various life stages of ranid frogs (Hayes and Jennings 1986). The pesticides disrupt the amphibian's nervous system and cause death by respiratory failure (Yosemite Association 2001). Predominant winds flow from the coast through the Central Valley and into the foothills and mountains of the Sierra Nevada; Davidson et al. (2002) compared the spatial pattern of declines for eight species of California amphibians with the amount of upwind agricultural land use. Declines of four ranid species, including California red-legged frog, were strongly associated with upwind agriculture use, suggesting that this group of amphibians may have a high sensitivity to agrochemicals.

Pathogens

Recent attention has been given to the role pathogens may play on localized population fluctuations of amphibians. The chytrid fungus has been found in several species of amphibians in California (U. S. Fish and Wildlife Service 2000a; Vrendenburg 2001). Chytrid fungus causes deterioration of the mouthparts of tadpoles and has been implicated in amphibian declines in Australia, Central and South America, and Europe (Vrendenburg 2001). The fungus has been found in a number of amphibians whose populations are declining, but it is not yet known if the fungus is playing a role in the decline (Fellers 2000).

California Red-Legged Frog (*Rana aurora draytonii*)

Status

FEDERAL: Threatened. Critical habitat designated on April 12, 2001 (66 FR 14626).

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: *Recovery Plan for the California Red-legged Frog* (2002).

PLACER NCCP/HCP CATEGORY: Class 2.

Distribution

California

California red-legged frogs are endemic to California. Their historical range extended coastally from the vicinity of Point Reyes National Seashore in Marin County, and inland from the vicinity of Redding, south to northwestern Baja California (Storer 1925; Jennings and Hayes 1985; Hayes and Krempels 1986).

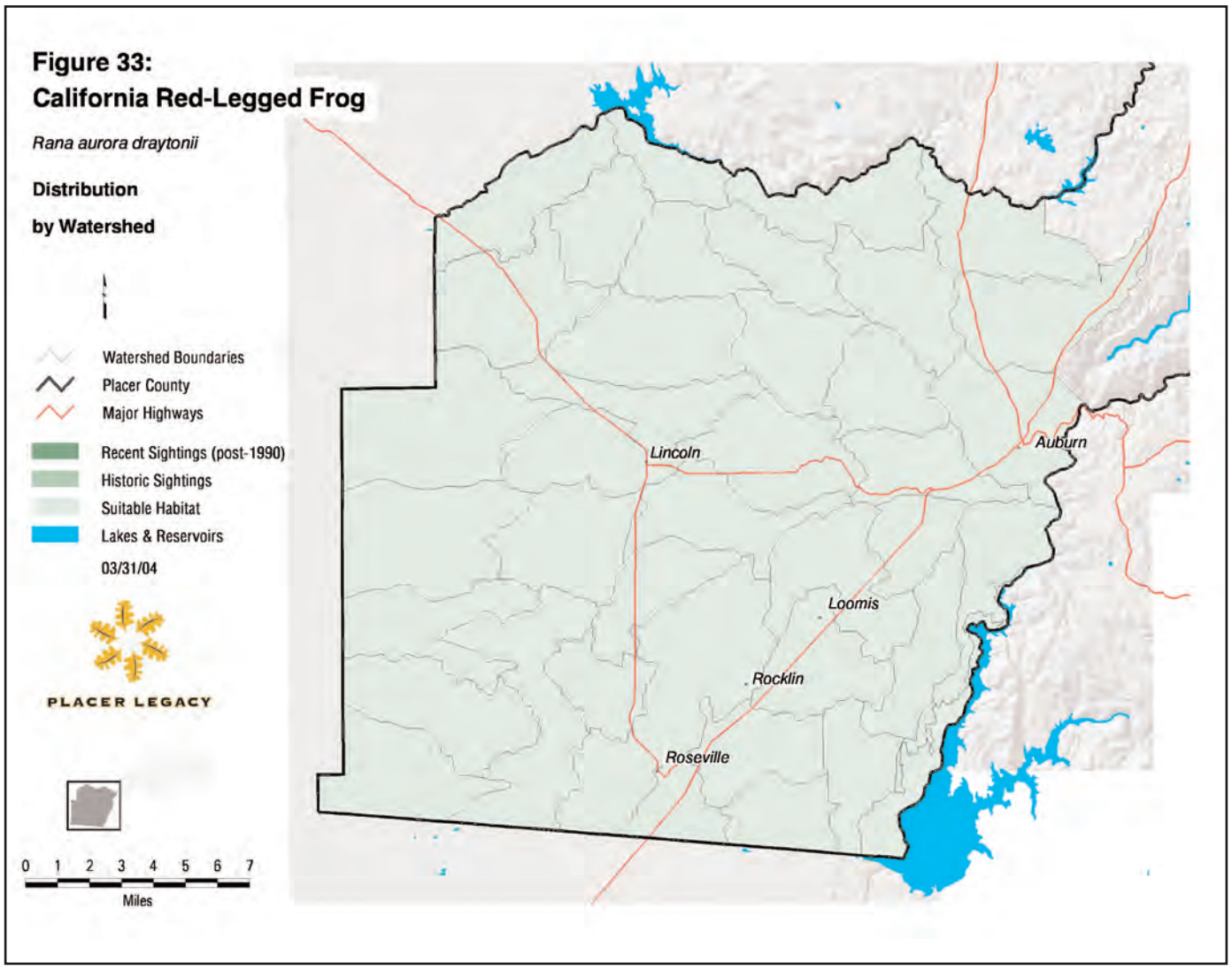
The species' current coastal distribution extends from Sonoma to Los Angeles Counties; it also occurs in isolated locations in the Sierra Nevada (including Butte, El Dorado, and Nevada Counties) and northern Transverse Ranges. It is relatively common in the San Francisco Bay area and along the central coast. California red-legged frogs are believed extirpated from the floor of the Central Valley (U. S. Fish and Wildlife Service 2000a). Two populations were recently discovered in the southern Transverse and Peninsular Ranges, where the species had been believed extirpated (66 FR 14626).

Placer County Phase I Planning Area

Historical

Historically, California red-legged frog occupied sections of the western slope of the Sierra Nevada from

California Red-Legged Frog Photo by Ed Ely



Shasta to Tulare Counties. The species occurred in at least 30 foothill drainages bordering the Central Valley; however, no specific location information is available for Placer County (U.S. Fish and Wildlife Service 2000a.) There are three historical records for California red-legged frogs in Placer County (Jennings and Hayes 1994; Williams pers. comm.). One of these occurrences was reportedly near Newcastle/Auburn within the Phase I Planning Area. The other two occurrences were near Michigan Bluff (California Natural Diversity Database 2003) and Dutch Flat (Williams pers. comm.), 15–20 miles outside the Phase I Planning Area.

Current

Most of the known populations on the western slope of the Sierra Nevada have been eliminated or fragmented. Only a few Sierra Nevada foothill drainages are currently known to support California red-legged frog populations. However, much of the Sierra Nevada has not been surveyed; consequently, the actual status in this area is uncertain (U.S. Fish and Wildlife Service 2000a). California red-legged frogs were observed on the Eldorado National Forest near Ralston Ridge in 2001 (California Natural Diversity Database 2003). Although located in Placer County, this sighting is approximately 16 miles from the northeastern boundary of Phase I Planning Area (Figure 33). A previously unknown population was recently discovered on San Juan Ridge, Nevada County (Williams pers. comm.).

Population Status and Trends

California

California red-legged frog has sustained a 70% reduction in its geographic range as a result of several factors acting singly or in combination (Jennings et al. 1992). The taxon was previously documented in 46 counties in California, but currently it occurs in only 26 (U.S. Fish and Wildlife Service 2000a; 61 FR 25813). Habitat loss and alteration, overexploitation, and introduction of

exotic predators were significant factors in the species' decline in the early- to mid-1900s. Reservoir construction, expansion of introduced predators, grazing, and prolonged drought fragmented and eliminated many of the Sierra Nevada foothill populations (61 FR 25813). California red-legged frog still occurs in Baja California (66 FR 14626); however, information on the species' status in that portion of the range could not be located.

Placer County Phase I Planning Area

Population status and trends in Placer County are difficult to determine because little information on locations of California red-legged frog is available. There are only three historical records for California red-legged frogs in Placer County, including a specimen collected in 1946 from a location vaguely described as Auburn (Jennings pers. comm.). This occurrence may or may not pertain to the Phase I Planning Area. The Auburn area population is considered extirpated (Jennings and Hayes 1994). Only one recent Placer County record of California red-legged frog could be found (California Natural Diversity Database 2003); however, this individual was located approximately 16 miles from the Phase I Planning Area.

Natural History

Habitat Requirements

California red-legged frogs have been found at elevations from sea level to about 1,500 meters (5,000 feet). They use a variety of habitat types; these include various aquatic, riparian, and upland habitats (U.S. Fish and Wildlife Service 2000a). However, individual frogs may complete their entire life cycle in a pond or other aquatic site that is suitable for all life stages (66 FR 14626). California red-legged frogs breed in aquatic habitats such as marshes, ponds, deep pools and backwaters in streams and creeks, lagoons, and estuaries. Breeding adults are often associated with dense, shrubby riparian or emergent vegetation and areas with deep (>0.7 meter [27 inches]) still or slow-moving water (U.S. Fish and Wildlife Service 2000a; 66 FR 14626). However, these frogs often successfully breed in artificial ponds with little or no emergent vegetation and have been observed in stream reaches that are not covered in riparian vegetation. An important factor influencing the suitability of aquatic breeding sites is the general lack of introduced aquatic predators (U.S. Fish and Wildlife Service 2000a).

Red-legged frogs (*R. aurora*) spend a substantial amount of time resting and feeding in riparian and emergent vegetation. The moisture and camouflage provided by the riparian plant community may provide good foraging habitat and may facilitate dispersal in addition to providing pools and backwater aquatic areas for breeding. Dispersal sites typically provide forage or cover opportunities and include boulders or rocks and organic debris such as downed trees or logs; industrial debris; and agricultural features such as drains, watering troughs, spring boxes, and abandoned sheds (66 FR 14626.) California red-legged frogs also use small mammal burrows and moist leaf litter (Jennings and Hayes 1994). Incised stream channels with portions narrower and deeper than 18 inches (46 centimeters) may also provide habitat (61 FR 25813). Use of this habitat type by California red-legged frogs is most likely dependent on year-to-year variations in climate and habitat suitability and varying requisites per life stage (66 FR 14626).

Reproduction

California red-legged frogs breed from November through March, although earlier breeding has been recorded in southern localities (Storer 1925). Males have paired vocal sacs and call in air (Hayes and Krempels 1986). Males appear at breeding sites 2–4 weeks before females (Storer 1925). Female California red-legged frogs deposit egg masses on emergent vegetation so that the masses float on the surface of the water (Hayes and Miyamoto 1984). Egg masses contain about 2,000–5,000 moderate-sized (2.0–2.8 millimeters [0.08–0.11 inches] in diameter), dark reddish brown eggs (Storer 1925; Jennings and Hayes 1985). Eggs hatch in 6–14 days (Storer 1925). Larvae generally undergo metamorphosis 3.5–7 months after hatching (Storer 1925; Wright and Wright 1949; Jennings et al. 1992); however, California red-legged frog tadpoles have recently been observed to overwinter in some areas (Fellers et al. 2001). Males attain sexual maturity by 2 years and females by 3 years of age (Jennings and Hayes 1985).

Dispersal Patterns

Because California red-legged frogs live in a Mediterranean climate where habitat quality varies spatially and over time, population sizes can vary widely between years. During favorable years, reproductive rates can be high and large numbers of dispersing young can increase the number of occupied sites.

Conversely, during extended droughts, frogs may disappear from previously occupied sites (66 FR 14626.) Survival from hatching to metamorphosis has been estimated as ranging from less than 1% (Jennings et al. 1992) to 1.9% (Cook 1997). In one pond study fewer than 5% of California red-legged frog larvae reached metamorphosis when bullfrog (*Rana catesbeiana*) tadpoles were placed in the ponds with them, whereas 30–40% reached metamorphosis in ponds that did not have bullfrog tadpoles (Lawler et al. 1999).

Longevity

No data are available on the longevity of California red-legged frogs. Minimum life spans for adults were estimated to be 8 years for males and 10 years for females in one central California population (Jennings et al. 1992).

Sources of Mortality

California red-legged frogs are eaten by native predators such as raccoons (*Procyon lotor*), great blue herons (*Ardea herodias*), and garter snakes (*Thamnophis* sp.), as well as by nonnative predators such as bullfrogs (U. S. Fish and Wildlife Service 2000a). Hayes and Jennings (1986) suggested that a variety of fish species may feed on California red-legged frogs and foothill yellow-legged frogs (*Rana boylii*). Smallmouth bass (*Micropterus dolomieui*) are known to eat larval and postmetamorphic red-legged frogs (Kiesecker and Blaustein 1998). Calef (1973) looked extensively into predation of red-legged frogs and found that Odonata (dragonfly and damselfly) larvae and giant diving beetles (*Lethocerus americanus*) prey on red-legged frog eggs. He also found that larvae were regularly preyed upon by salamanders and newts. Fungal infection and localized desiccation were reported as sources of egg mortality (Calef 1973).

Behavior

Hayes and Tennant (1985) found juvenile frogs to be active diurnally and nocturnally, whereas adult frogs were largely nocturnal. The season of activity for red-legged frogs seems to vary with the local climate (Storer 1925); individuals from coastal populations with more constant temperatures are rarely inactive (Jennings et al. 1992). Individuals from inland sites, where temperatures are lower, may become inactive for long intervals (Jennings et al. 1992).

California red-legged frogs have a varied diet that includes both invertebrates and vertebrates (U.S. Fish and Wildlife Service 2000a). The tadpoles are believed to eat algae; however, this has not been studied (Jennings et al. 1992). In general, amphibian larvae feed on bacteria, protozoans, free-floating algae, and other small particles suspended in water (Stebbins and Cohen 1995). Invertebrates were found to be the most common food item consumed by adult frogs. However, larger California red-legged frogs will eat vertebrates, such as Pacific tree frogs (*Hyla regilla*) and California mice (*Peromyscus californicus*) (Hayes and Tennant 1985). Vertebrates represented more than half the prey mass eaten by larger frogs examined by Hayes and Tennant (1985). Feeding activity probably occurs along the shoreline and on the surface of the water as evidenced by the types of foods eaten (Hayes and Tennant 1985).

Movement and Migratory Patterns

California red-legged frog juveniles and adults may disperse from breeding sites at any time of year. California red-legged frogs have been found more than 1.8 miles (2.9 kilometers) from their breeding site living within streams. They have been observed in adjacent dense riparian vegetation more than than 328 feet (100 meters) from water, although they typically occur within 200 feet (60 meters) of water (66 FR 14626.)

Beginning with the first rains in the fall, some individuals make overland excursions through upland habitats during periods of wet weather. Most of these overland movements take place at night. Evidence from marked frogs on the California coast near San Simeon indicate that frogs may move over upland habitats for a distance of about 1 mile during the course of a wet season. In addition, California red-legged frogs have been observed to make long-distance movements that are straight-line, point-to-point migrations rather than using corridors for moving between habitats. (66 FR 14626.)

Ecological Relationships

Red-legged frogs are preyed upon during every life stage (Lawler et al. 1999; Calef 1973). However, postmetamorphic red-legged frogs will consume anything they can catch that is not distasteful (Jennings et al. 1992). Red-legged frog larvae appear to reduce their level of activity while in the presence of potential predators. This may affect their ability to forage and can potentially reduce the size of the animal at metamorphosis (Lawler et al. 1999.) Intraspecific competition among adult red-legged frogs is rare. Calef (1973) suggested that natural mortality is so high during the premetamorphic stage that competition among postmetamorphic frogs is negligible.

Red-legged frogs occasionally co-occur with foothill yellow-legged frogs at some localities; however, different microhabitat preferences may limit competition (Zeiner et al. 1988). The introduction of nonnative predatory fishes and bullfrogs has had a significant effect on this species. These impacts are discussed in more detail in other sections (*Sources of Mortality* above and *Population Threats* below).

Population Threats

Threats to populations of red-legged frogs have only recently been addressed. Reasons for declines and potential threats to populations of California red-legged frogs include habitat loss, degradation, and fragmentation; introduced nonnative predators; water management; pesticides; and natural pathogens. These are discussed below.

Habitat Loss, Degradation, and Fragmentation

The primary factors that have led to declining populations of California red-legged frogs are loss, degradation, and fragmentation of habitat. Urbanization, conversion of lands to agriculture, overgrazing, and timber harvesting are some of the activities that have resulted in habitat loss, degradation, and fragmentation. The rate of urbanization in California is considered a significant threat to California red-legged frog. Urbanization often isolates or fragments existing habitat. In addition, habitat adjacent to urbanized areas may become more susceptible to exotic invasions. Drainages in the area of developments often experience changes in hydroperiod due to water diversions and/or wastewater effluent. Conversion of intermittent drainages to perennial ones may create suitable conditions for nonnative predators (U. S. Fish and Wildlife Service 2000a.) Land use changes such as conversion of natural lands to agriculture, livestock grazing in riparian areas, timber harvesting, and historic placer mining not only changed or eliminated existing habitat but contributed significantly to the siltation of aquatic habitats (Jennings et al. 1992; Fisher and Shaffer 1996; Jennings 1996; U.S. Fish and Wildlife Service 2000a).

Introduced Predators

Predatory nonnative fish and amphibians are particularly significant threats to California red-legged frogs. Moyle (1973) suggested that bullfrogs were the single most important factor leading to the elimination of California red-legged frogs from the San Joaquin Valley floor. Several studies provide evidence that bullfrogs may play a role in the decline of California red-legged frog populations (Fisher and Shaffer 1996; Kiesecker and Blaustein 1998; Lawler et al. 1999). Although California red-legged frogs can occasionally persist when they occur with bullfrogs, survivorship of red-legged frogs substantially declines when nonnative fish are also present (Kiesecker and Blaustein 1998). Locations where introduced fishes were abundant contained few California red-legged frogs, indicating that the presence of these fish had a negative influence on the abundance of the frogs (Hayes and Jennings 1986).

In addition to predation, bullfrogs may also have a competitive advantage over red-legged frogs (Hayes and Jennings 1986). Bullfrogs are larger, have more generalized food habits, and have an extended breeding season; moreover, bullfrog eggs and larvae are unpalatable to predatory fish (Walters 1975). Bullfrogs also can interfere with red-legged frog reproduction; both California and northern red-legged frogs have been observed in amplexus with both male and female bullfrogs (U.S. Fish and Wildlife Service 2000a).

Water Management

Water diversions and impoundments have altered habitat and made them less suitable for many ranid species (Jennings 1996). Water diversions change the hydrology of drainage systems; these changes can isolate populations and restrict the use of or eliminate dispersal corridors. Water impoundments have a similar effect where they create barriers to dispersal (Jennings 1996; U.S. Fish and Wildlife Service 2000a). Additionally, reservoirs provide suitable habitat for bullfrogs and are typically stocked with nonnative fish. These nonnative species frequently disperse upstream and downstream of reservoirs into California red-legged frog habitat (U.S. Fish and Wildlife Service 2000a).

Pesticides

Researchers have found that amphibians absorb pesticides in aquatic and terrestrial systems. Pesticides, herbicides, and other toxins are known to be harmful to various life stages of ranid frogs (Hayes and Jennings 1986). The pesticides disrupt the amphibian's nervous system and cause death by respiratory failure (Yosemite Association 2001). Predominant winds flow from the coast through the Central Valley and into the foothills and mountains of the Sierra Nevada; Davidson et al. (2002) compared the spatial pattern of declines for eight species of California amphibians with the amount of upwind agricultural land use. Declines of four ranid species, including California red-legged frog, were

strongly associated with upwind agriculture use, suggesting that this group of amphibians may have a high sensitivity to agrochemicals.

Pathogens

Recent attention has been given to the role pathogens may play on localized population fluctuations of amphibians. The chytrid fungus has been found in several species of amphibians in California (U. S. Fish and Wildlife Service 2000a; Vrendenburg 2001). Chytrid fungus causes deterioration of the mouthparts of tadpoles and has been implicated in amphibian declines in Australia, Central and South America, and Europe (Vrendenburg 2001). The fungus has been found in a number of amphibians whose populations are declining, but it is not yet known if the fungus is playing a role in the decline (Fellers 2000).

Western Spadefoot (*Scaphiopus hammondii*)

Status

FEDERAL: None.

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

Western spadefoot primarily occurs in California but it has been recorded from the vicinity of Redding in Shasta County south into Baja California (Stebbins 1985). Coastal populations occur in the Salinas River basin (Monterey, San Benito, and San Luis Obispo Counties); on the Carrizo Plain (San Luis Obispo County); and in the foothills northeast of Point Conception (Santa Barbara County). Southern California populations have been found in the Los Angeles basin, in the interior valleys of Riverside and northern San Diego Counties, and on the coastal plains of Orange and San Diego Counties. Western spadefoot populations extend south on the Pacific slope to Mesa de San Carlos in Baja California Norte (Storer 1925; Stebbins 1985; Jennings and Hayes 1994; Museum of Vertebrate Zoology and California Academy of Sciences catalogue records).

Western spadefoots have been recorded from seventeen counties either in or bordering the Central Valley (Jennings and Hayes 1994; Barry 2000; Jones & Stokes file information). Western spadefoots have been recorded along the eastern edge of the Central Valley from Redding in Shasta County to southern Kern County, along the western edge of the valley west of Orland in Glenn County, and near Corral Hollow in Alameda and San Joaquin Counties.

Placer County Phase I Planning Area

Western spadefoot occurs in the western portion of Phase I Planning Area along the interface of the Central Valley and Sierra Nevada foothills (Figure 34). However, the species is probably uncommon in the county; the few recorded occurrences, which are from the Roseville area, are summarized below.

In 1990 more than 30 metamorphs were observed in a drying intermittent drainage near the intersection of Woodcreek Oaks Boulevard and Pleasant Grove Boulevard in western Roseville (California Natural Diversity Database 2003).

In 1991 several tadpoles were observed in an intermittent tributary of Kaseberg Creek and adjacent vernal pools in Roseville (California Natural Diversity Database 2003).

In 1993, one adult was observed crossing Phillip Road 1.5 mile west of the junction with Fiddyment Road (California Natural Diversity Database 2003).

In 1994, five tadpoles were observed in vernal pools near Taylor Road in Roseville (California Natural Diversity Database 2003).

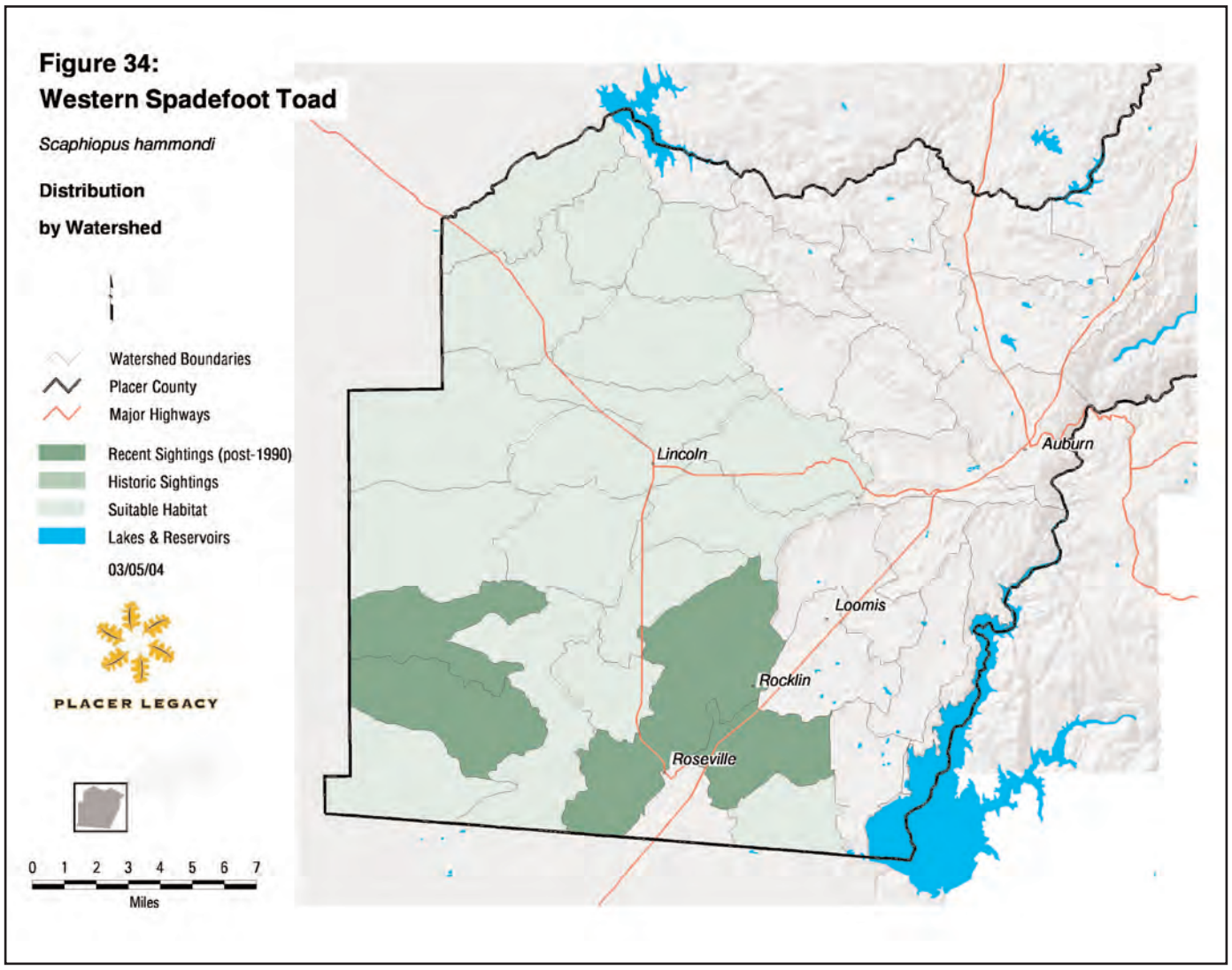
There is a reported record west of Roseville near Fiddyment Road in a small intermittent drainage (Placer NCCP/HCP Scientific Working Group undated).

Population Status and Trends

California

Limited available data indicate that western spadefoot populations are sometimes large and may include aggregations of up to 1,000 chorusing males (Stebbins 1951; Jennings and Hayes 1994). Spadefoots may be more likely to survive to metamorphosis than most other pond-breeding anurans because spadefoots metamorphose quickly and their natural predators are scarce (Barry 2000). However, Feaver (1971) found that

Western Spadefoot Photo by Alison Sheehey



even though western spadefoots metamorphose rapidly, more than 70% of larvae died because their ponds dried too quickly.

According to Jennings and Hayes (1994), more than 80% of the habitat once occupied by western spadefoot south of Santa Barbara and Kern Counties has been lost to urban and agricultural development. More than 30% of western spadefoot habitat north of that region has been lost, primarily to agricultural development. Although the decline in southern California is clearly documented, the decline in northern California is much less clear. Previously undocumented populations may yet be discovered that could fill distribution gaps in the species' known range; a recent example is the 1992 discovery of a Glenn County population (Fisher cited in Barry 2000).

Placer County Phase I Planning Area

Detailed population studies have not been conducted in Placer County.

Natural History

Habitat Requirements

Western spadefoots can be found in dry grassland habitat close to seasonal wetlands such as vernal pool complexes, typically near extensive areas of friable (but not usually sandy) soil (Storer 1925; Stebbins 1951; Barry 2000). Although spadefoot populations primarily occur in grassland settings, they are occasionally found in valley-foothill woodlands (Zeiner et al. 1988). Western spadefoots can also be found in creeks, drainages, and ponds (Jones & Stokes file information; Westphal pers. comm.).

Western spadefoots require seasonal wetlands for reproduction and metamorphosis. The specific physical attributes that make such wetlands and adjacent uplands suitable for spadefoots are not well known, but suitable ponds must exhibit sufficient depth and surface area to persist at least several weeks (Stebbins 1951; Feaver 1971; Jennings and Hayes 1994; Morey 1998). It is frequently assumed that spadefoots require loose soils for subsurface dormancy (Jennings and Hayes 1994); however, there is some evidence that spadefoots may also use rodent burrows (Stebbins 1951; Barry 2000). Also, most sites that support western spadefoots are moderately to heavily grazed (Morey pers. comm.).

Several aspects of the life history of western spadefoot are unknown, including the parameters that define suitable habitat and the behavior of metamorphs following pool drying. Perhaps the most significant data gaps involve the species' metapopulation structure (i.e., the distribution of populations relative to one another); movement requirements; and the effect of habitat fragmentation on metapopulation structure and viability (Jennings and Hayes 1994; Morey pers. comm.). This information is important for understanding the amount and distribution of habitat necessary to sustain long-term persistence of this species (Jennings and Hayes 1994). Although the spatial requirements of western spadefoots are unknown, it has been postulated that populations are more likely to persist in large complexes of vernal pools than in small, isolated pools. In general, small or isolated occurrences of most organisms are more likely to experience random local extirpation as a result of adverse genetic effects, natural disturbances, or demographic stochasticity (e.g., random variation in birth and death rates) (Goodman 1987; Hanski and Gilpin 1997).

Reproduction

Adult western spadefoots spend most of the year in self-excavated underground retreats or, possibly, mammal burrows (Stebbins 1951). They emerge from underground retreats during heavy rains in autumn and winter and spawn in seasonal wetlands, such as vernal pools, in late winter or early spring (Storer 1925; Stebbins 1951). Spawning cues are believed to include the onset of heavy, warmer winter rains of subtropical origin; the filling of vernal pools with rainwater; and the increase of ambient temperatures (Feaver 1971; Jennings and Hayes 1994; Morey 1998; Barry 2000). During dry years, spawning pools may not fill, and western spadefoots may not reproduce. Eggs hatch in less than a week and larvae metamorphose in 30–80 days; the period to metamorphosis apparently depends on the duration of pool depth that is sufficient to support larvae, and possibly on pool temperatures (Feaver 1971; Morey 1998).

Dispersal Patterns

There are no available data on movement patterns or colonization abilities of adult western spadefoots (Jennings and Hayes 1994). Following metamorphosis, western spadefoots disperse after spending a few hours or days near the pond margins (Zeiner et al. 1988). Little is known about the behavior of metamorphs once they leave the pool. They may use a variety of underground retreats, such as burrows and desiccation cracks in the dried pond bottom (as has been reported for *S. intermontanus*) (Barry 2000), as well as seeking cover under surface objects such as boards and decomposing cow dung (Zeiner et al. 1988).

Longevity

No published information regarding the longevity of western spadefoot is available.

Sources of Mortality

Because spadefoots occur primarily in seasonal wetlands and have a short breeding cycle (approximately 30 days), aquatic predators are few, especially for adult spadefoots. California tiger salamander (*Ambystoma californiense*), garter snakes (*Thamnophis* spp.), great blue heron (*Ardea alba*), and raccoon (*Procyon lotor*) are probably the most important predators of larval and postmetamorphic western spadefoots (Childs 1953; Feaver 1971).

In a study of amphibian reproduction in 30 vernal pools in Madera County, Feaver (1971) reported a mortality rate of 81% for larval spadefoot populations, 10% of which he attributed to *Ambystoma* larvae, while the remainder was due to pools drying before metamorphosis was completed. Because vernal pools are seasonal, they do not support predatory species such as bullfrogs (*Rana catesbeiana*), bass, and crayfish (except for stocked mosquitofish and transient bullfrogs). Other potential sources of mortality are described in *Population Threats* below.

Behavior

Western spadefoots are rarely found above ground except following relatively warm (≥10.0–12.8°C [50–55°F]) rains. They may emerge from their burrows in any month between October and April if sufficient rain has fallen to stimulate surface activity; however, the cues that induce surface activity remain poorly understood. (Jennings and Hayes 1994.)

Adult western spadefoots consume a variety of invertebrates, including various insects and annelids (worms and leeches), mostly during the short time they are active on the surface (Morey and Guinn 1992). Larvae filter feed on smaller materials in the water column and scrape larger organisms such as algae (McCready pers. comm.). However, spadefoot larvae are opportunistic feeders and will take advantage of dead larvae if present in the pool (McCready pers. comm.). Western spadefoot larvae are not known to be as cannibalistic as either southern spadefoot (*S. muliplicata*) or plains spadefoot (*S. bombifrons*), both of which are dimorphic, occurring in both omnivorous and carnivorous morphs (Pomeroy 1981; McCready pers. comm.).

Ecological Relationships

Because they spend most of the year below ground, adult spadefoots are able to avoid most predators. However, dense populations of tadpoles may be heavily preyed upon by wading birds, raccoons, or garter snakes. Introduced predators such as mosquitofish, other fish species, and bullfrogs are found in more permanent water habitats than those that typically provide suitable habitat for western spadefoots.

Population Threats

Loss of vernal pool habitat from agricultural and urban development is likely the most significant threat to western spadefoots. Jennings and Hayes (1994) noted the loss of more than 80% of occupied habitat in southern California and more than 30% in northern California; they noted severe declines east of Sacramento, Fresno, and Bakersfield. McCready (pers. comm.) found a significant decline in the number of adult spadefoots over a 20-year period (1960s through 1980s) in Sacramento County, near Sunrise Boulevard, State Route 16, Grant Line Road, and Douglas Road. Urban development along with increased traffic has increased in this area since his original survey work in the late 60s and early 70s.

In addition to habitat loss, habitat fragmentation by urban development, agricultural conversion, and road construction is likely to threaten western spadefoot populations. The migratory and spatial requirements of western spadefoots are not well understood (Morey pers. comm.), but it is likely that development and land conversion result in smaller and more isolated populations by removing habitat and imposing barriers to movement. Also, vehicle-related mortality on roads may be an important threat to western spadefoot populations in areas where occupied vernal pools are adjacent to roads and potential movement and dispersal corridors.

California Tiger Salamander (*Ambystoma californiense*)

Status

FEDERAL: Proposed Threatened (68 FR 100 28648–28670, May 23, 2003); Endangered (Santa Barbara population only) (65 FR 184 57242–57264); Endangered (68 FR 53 13498–13520, March 19, 2003) (Sonoma County population).

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 2.

Distribution

California

California tiger salamander is endemic to California. Historically, they probably occurred in grassland habitats throughout much of the state. Habitat conversion has reduced the species' range and decreased breeding populations (Stebbins 1985). Currently, California tiger salamander occurs in the Central Valley and Sierra Nevada foothills from Yolo and Colusa Counties south to Tulare County, and in the coastal valleys and foothills from Sonoma to Santa Barbara Counties (Zeiner et al. 1988). Isolated populations are found at the Gray Lodge Wildlife Area in Butte County (Jennings and Hayes 1994) and at Grass Lake in Siskiyou County (Zeiner et al. 1988). Most populations occur at elevations below 460 meters (1,500 feet), but tiger salamanders have been recorded at elevations up to 1,370 meters (4,500 feet). Although populations have declined, the species continues to breed at a relatively large number of locations within its current range (U.S. Fish and Wildlife Service 1994).

Placer County Phase I Planning Area

The California Department of Fish and Game does not consider this species to occur in Placer County; however, habitat that appears suitable for California tiger salamander occurs in the western part of the county (Figure 35). Because aquatic tiger salamander larvae are imported from other states for fishing bait and transplanted by fishermen, any new locality records should be evaluated with caution (U.S. Fish and Wildlife Service 2000b).

Population Status and Trends

California

Although California tiger salamander still occurs over much of its previous distribution, the species appears to be in the initial stages of fragmentation and decline. It is still present in most counties; however, peripheral populations in the Sacramento and San Joaquin Valleys are no longer extant. (Fisher and Shaffer 1996). California tiger salamander populations have declined as a result of two primary factors: widespread habitat loss and habitat fragmentation (U.S. Fish and Wildlife Service 1994).

Placer County Phase I Planning Area

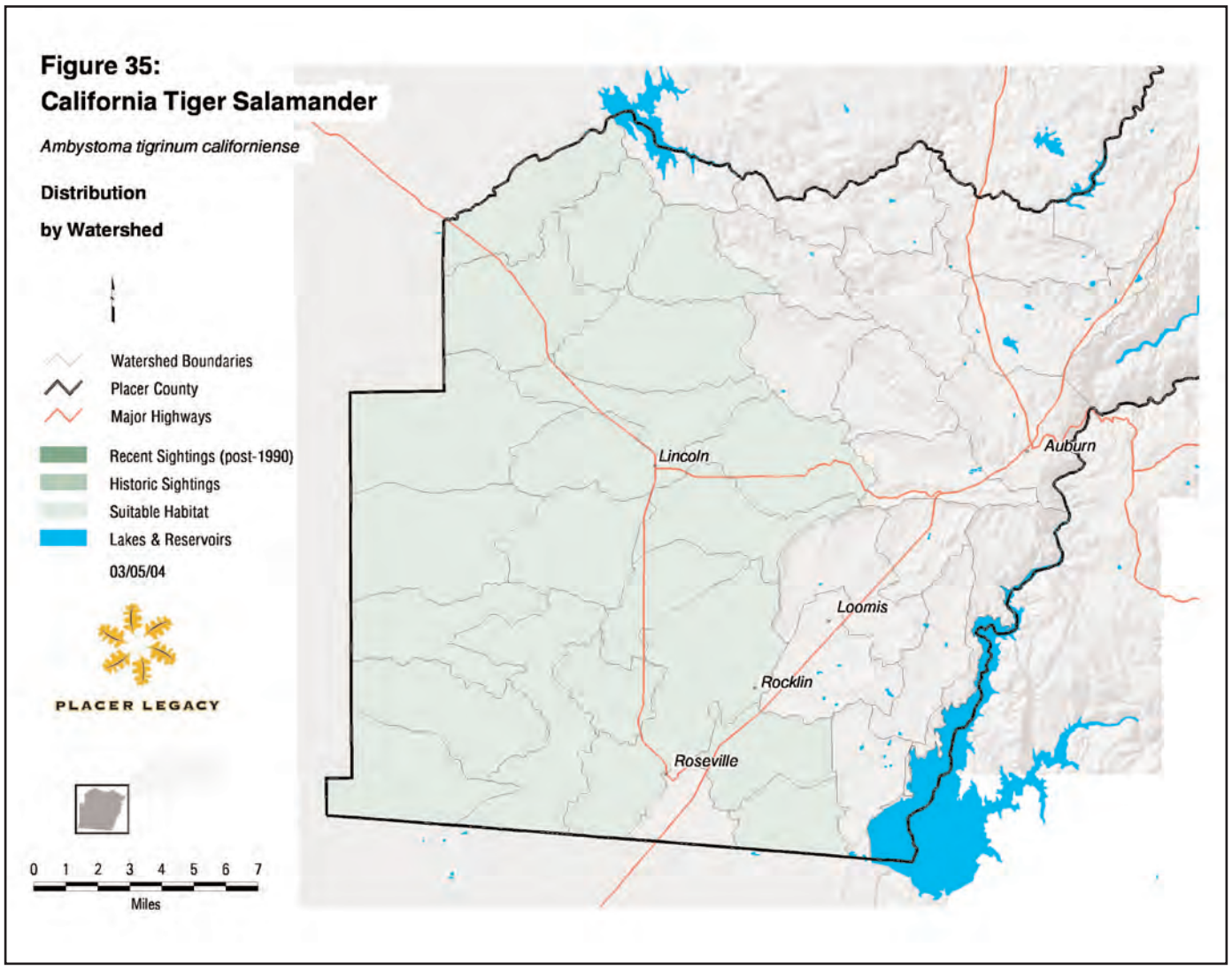
California tiger salamander is not known to occur in Placer County.

Natural History

Habitat Requirements

California tiger salamanders require two major habitat components: aquatic breeding sites and terrestrial aestivation or refuge sites. California tiger salamanders inhabit valley and foothill grasslands and the grassy understory of open woodlands, usually within one mile of water (Jennings and Hayes 1994). California tiger salamander is terrestrial as an adult and spends most of its time underground in subterranean refugia, especially ground-squirrel burrows and occasionally human-made structures, emerging only for brief periods to breed. Tiger salamanders breed and lay their eggs primarily in vernal pools and other ephemeral ponds that

California Tiger Salamander Jones & Stokes file photo



fill in the winter and often dry by summer (Loredo et al. 1996); they sometimes use permanent human-made ponds (e.g., stockponds), reservoirs, and small lakes, although they are subject to reproductive failure in water bodies that support introduced predatory fish species (Stebbins 1972; Zeiner et al. 1988). Streams are rarely used for reproduction. Adult salamanders migrate from upland habitats to aquatic breeding sites during the first major rainfall events of fall and early winter and return to upland habitats after breeding. This species requires small mammal (e.g., California ground squirrel) burrows for cover during the non-breeding season and during migration to and from aquatic breeding sites (Zeiner et al. 1988). California tiger salamanders also use logs, piles of lumber, and shrink-swell cracks in the ground for cover (Holland et al. 1990). California tiger salamanders can overwinter in burrows as far as 1 mile from their breeding site (Jennings and Hayes 1994).

California tiger salamander is particularly sensitive to the duration of ponding in aquatic breeding sites. Because tiger salamanders have an approximately 10-week-long developmental period, the longest lasting seasonal ponds or vernal pools are the most suitable type of breeding habitat; such pools are also typically the largest (Jennings and Hayes 1994). Moreover, large vernal pool complexes, rather than isolated pools, probably offer the best quality habitat; these areas can support a mixture of core breeding sites and nearby refuge habitat (Shaffer et al. 1993; Jennings and Hayes 1994). The range of physical characteristics of aquatic sites that allow successful reproduction and metamorphosis is poorly understood; however, these factors are under investigation (Shaffer and Stanley 1992; Shaffer et al. 1993; Jennings and Hayes 1994). This area of knowledge is particularly important for protecting and managing habitat for the viability of this species (Jennings and Hayes 1994).

The suitability of California tiger salamander habitat is proportional to the abundance of upland refuge sites that are near aquatic breeding sites. California tiger salamanders primarily use California ground squirrel burrows as refuge sites (Loredo et al. 1996); Botta's pocket gopher burrows are also frequently used (Barry and Shaffer 1994; Jennings and Hayes 1994). Loredo et al. (1996) emphasized the importance of California ground squirrel burrows as refugia for California tiger salamanders in the population they studied near Concord and suggested the existence of a commensal relationship, in which tiger salamanders benefit from the burrowing activities of California ground squirrels. Furthermore, tiger salamanders apparently do not avoid burrows occupied by ground squirrels (Loredo et al. 1996).

The proximity of refuge sites to aquatic breeding sites also affects the suitability of salamander habitat. Although the variation in distances between breeding and refuge sites is poorly studied (Jennings and Hayes 1994), juvenile salamanders are known to migrate distances up to 1.6 kilometers (1 mile) from breeding sites (Austin and Shaffer 1992). Loredo et al. (1996) found that tiger salamanders may use burrows that are first encountered during movements from breeding to upland sites. In their study area, where the density of California ground squirrel burrows was high, the average migration distances between breeding and refuge sites for adults and juveniles was 35.9 meters (118 feet) and 26.0 meters (85 feet), respectively. These findings suggest that, although salamanders may migrate up to 1.6 kilometers (1 mile), migration distances are likely to be less in areas supporting a large number of refugia sites closer to breeding sites. Also, habitat complexes that include upland refugia relatively close to breeding sites are considered more suitable, because predation risk and physiological stress may increase with migration distance.

Barriers that impede seasonal migrations (e.g., road berms, walls, buildings) may have affected tiger salamander populations in some areas (Shaffer and Fisher 1991; Shaffer and Stanley 1992; Barry and Shaffer 1994). Also, because California tiger salamanders readily attempt to cross roads during migration, roads that sustain heavy vehicle traffic can be a major source of salamander mortality (Jones & Stokes file data). Accordingly, development of artificial structures that could impede movements and maintenance of roads that support a considerable amount of vehicle traffic in areas that support California tiger salamander populations can severely degrade salamander habitat (see Jennings and Hayes 1994).

California tiger salamander embryos and larvae are susceptible to predation by fish (Stebbins 1972; Zeiner et al 1988; Shaffer et al. 1993), and larval tiger salamanders are rarely found in aquatic sites that support fish (Shaffer and Fisher 1991; Shaffer and Stanley 1992; Shaffer et al. 1993). However, because adult tiger salamanders are fairly long-lived, they may return annually to breed at sites with fish even though the larvae do not survive (Jones & Stokes file information). Shaffer et al. (1993) also found a negative correlation between the occurrence of California tiger salamanders and the presence of bullfrogs; however, this relationship was observed only in unvegetated ponds, suggesting that vegetation structure in aquatic breeding sites may be important for survival.

Reproduction

Adult California tiger salamanders migrate to and congregate at aquatic breeding sites during warm rains, primarily between November and February (Shaffer and Fisher 1991; Barry and Shaffer 1994; Jennings and Hayes 1994). Adult tiger salamanders are rarely observed except during this period, when they are aboveground (Loredo et al. 1996). During this period, tiger salamanders breed and lay eggs primarily in vernal pools and other shallow ephemeral ponds that fill in the winter and often dry by summer (Loredo et al. 1996); they also use permanent human-made ponds (e.g., stockponds). Spawning usually occurs within a few days after migration, and adults probably leave the breeding sites at night soon after spawning (Storer 1925; Barry and Shaffer 1994). Eggs are laid singly or in clumps on both submerged and emergent vegetation and on submerged debris in shallow water (Stebbins 1972; Shaffer and Fisher 1991; Barry and Shaffer 1994; Jennings and Hayes 1994). Larvae develop rapidly, and metamorphosis begins in late spring or early summer (Loredo-Prendeville 1995). At least 10 weeks are required to complete metamorphosis (Feaver 1971). Juveniles disperse from aquatic breeding sites to upland habitats after metamorphosis (Storer 1925; Holland et al. 1990).

Local populations of California tiger salamanders may not reproduce during years of low rainfall when ephemeral pools do not fill (Barry and Shaffer 1994; Jennings and Hayes 1994). However, it is presumed that the longevity of this species allows local populations to persist through all but the longest drought periods (Barry and Shaffer 1994).

Dispersal Patterns

Dispersal of juveniles from natal ponds to underground refugia occurs during summer months when breeding ponds dry out. Juveniles disperse from natal sites after spending a few hours or days near the pond margin (Jennings and Hayes 1994). Dispersal distance varies and may increase with an increase in precipitation. Juveniles have been found more than 1,200 feet from breeding ponds, but most juveniles tend to remain closer. (U.S. Fish and Wildlife Service 2000b.)

Longevity

Individual California tiger salamanders have been known to live more than 10 years (U.S. Fish and Wildlife Service 2000b).

Sources of Mortality

See *Population Threats* and *Ecological Relationships* below.

Behavior

California tiger salamanders are mostly inactive during cold or hot temperatures. Adult salamanders live in subterranean refugia most of the year, but they emerge at night during rains before and after breeding. Some diurnal activity occurs during breeding. Postmetamorphic juveniles disperse from breeding sites in late spring or early summer (U.S. Fish and Wildlife Service 2000b).

Aquatic larvae feed on algae, small crustaceans, and small mosquito larvae for about 6 weeks after hatching (U.S. Fish and Wildlife Service 2000b). Larger larvae feed on zooplankton, amphipods, mollusks, and smaller tadpoles of Pacific treefrog (*Pseudacris regilla*), red-legged frog (*Rana aurora*), western toad (*Bufo boreas*) and western spadefoot (*Scaphiopus hammondi*) (Zeiner et al. 1988; U.S. Fish and Wildlife Service 2000b). Postmetamorphic juveniles and adults appear to be "sit and wait" predators (Lindquist and Bachmann 1980). During the fall and winter, adult salamanders emerge from underground retreats during rain events and on nights of high relative humidity to feed and migrate to breeding ponds (U.S. Fish and Wildlife Service 2000b). Adults eat earthworms, snails, insects, fish, and small mammals (Stebbins 1972). During aestivation, California tiger salamanders eat very little (Shaffer et al. 1993; U.S. Fish and Wildlife Service 2000b).

Movement and Migratory Patterns

California tiger salamander movement patterns consist primarily of seasonal movements between aquatic breeding and terrestrial aestivation/refugia habitats. Refer to preceding discussions.

Ecological Relationships

California tiger salamander embryos and larvae are susceptible to predation by fish (Stebbins 1972; Zeiner et al. 1988; Shaffer et al. 1993), and tiger salamander larvae are rarely found in aquatic sites that support predatory fish (Shaffer and Fisher 1991; Shaffer and Stanley 1992; Shaffer et al. 1993). Aquatic larvae are taken by herons and egrets and possibly by garter snakes (Zeiner et al. 1988). Shaffer et al. (1993) also found a negative correlation between the occurrence of California tiger salamanders and the presence of bullfrogs; however, this relationship was observed only in unvegetated ponds. Because of their secretive behavior and limited periods above ground, adult California tiger salamanders have few predators (U.S. Fish and Wildlife Service 2000b).

Population Threats

California tiger salamander populations have declined dramatically throughout the species' range as a result of two primary factors: widespread habitat loss and habitat fragmentation. These factors have both been caused by conversion of valley and foothill grassland and oak woodland habitats to agricultural and urban development (Stebbins 1985). For example, residential development and land use changes in the species' range have removed or fragmented vernal pool complexes, eliminated refuge sites adjacent to breeding areas, and reduced habitat suitability for the species over much of the Central Valley (Barry and Shaffer 1994; Jennings and Hayes 1994). Moreover, grading activities have probably directly eliminated large numbers of salamanders (Barry and Shaffer 1994).

The introduction of bullfrogs, Louisiana red swamp crayfish, and nonnative fishes (mosquitofish, bass, and sunfish) into aquatic habitats has also contributed to declines in tiger salamander populations (Jennings and Hayes 1994; Sorensen 1994; U.S. Fish and Wildlife Service 2000b). These nonnative species prey on tiger salamander larvae and may eliminate larval populations from breeding sites (Jennings and Hayes 1994).

Burrowing mammal control programs are considered a threat to California tiger salamander populations. Rodent control through destruction of burrows and release of toxic chemicals into burrows can cause direct

mortality to individual salamanders and may result in a decrease of available suitable habitat (U.S. Fish and Wildlife Service 2000b).

Barriers that impede seasonal migrations (e.g., road berms, walls, buildings) may have affected tiger salamander populations in some areas (Shaffer and Fisher 1991; Shaffer and Stanley 1992; Barry and Shaffer 1994). Also, because California tiger salamanders readily attempt to cross roads during migration, roads that sustain heavy vehicle traffic can be a major source of salamander mortality (Jones & Stokes file data). Accordingly, development of artificial structures that could impede movements and maintenance of roads that support a considerable amount of vehicle traffic in areas that support California tiger salamander populations can severely degrade salamander habitat (Jennings and Hayes 1994).

Giant Garter Snake (*Thamnophis gigas*)

Status

FEDERAL: Threatened October 20, 1993, 522–58 FR 54065.

STATE: Threatened (June 27, 1971).

OTHER: World Conservation Union (IUCN): Vulnerable.

RECOVERY PLAN: *Draft Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*)*. (U.S. Fish and Wildlife Service 1999).

PLACER NCCP/HCP CATEGORY: Class 2.

Distribution

California

Giant garter snake is a California endemic species found only in the Sacramento and San Joaquin Valleys (Fitch 1941; Hansen and Brode 1980; Rossman and Stewart 1987; U.S. Fish and Wildlife Service 1999). Records of giant garter snakes coincide roughly with the historical distribution of the large flood basins, freshwater marshes, and tributary streams of the Central Valley of California (Hansen and Brode 1980). The distributional range of this species probably extended from Butte County in the north to Buena Vista Lake in Kern County in the south. The eastern and western boundaries of the range are believed to be the foothills of the Coast Ranges and the Sierra Nevada (U.S. Fish and Wildlife Service 1999). Rossmann et al. (1996) described an elevational range for giant garter snake of 0–122 meters (0–400 feet). Agricultural and flood control activities have extirpated the species from the southern one-third of its range in the former wetlands associated with the historic Buena Vista, Tulare, and Kern lakebeds (Hansen and Brode 1980; Hansen 1986, 1988; California Department of Fish and Game 1992b; U.S. Fish and Wildlife Service 1999). Today, populations of giant garter snake are found in the Sacramento Valley and isolated portions of the San Joaquin Valley (U.S. Fish and Wildlife Service 1999). Recent records indicate 13 recognized populations distributed from near Burrel in Fresno County north to the vicinity of Chico in Butte County (Hansen and Brode 1980; Rossman and Stewart 1987; U.S. Fish and Wildlife Service 1999).

Placer County Phase I Planning Area

Historical

The western third of the Phase I Planning Area occurs within the Central Valley proper and supports numerous low-elevation tributaries and wetlands that could have provided suitable habitat for this species. However, there are no historical records of the species in the Phase I Planning Area.

Current

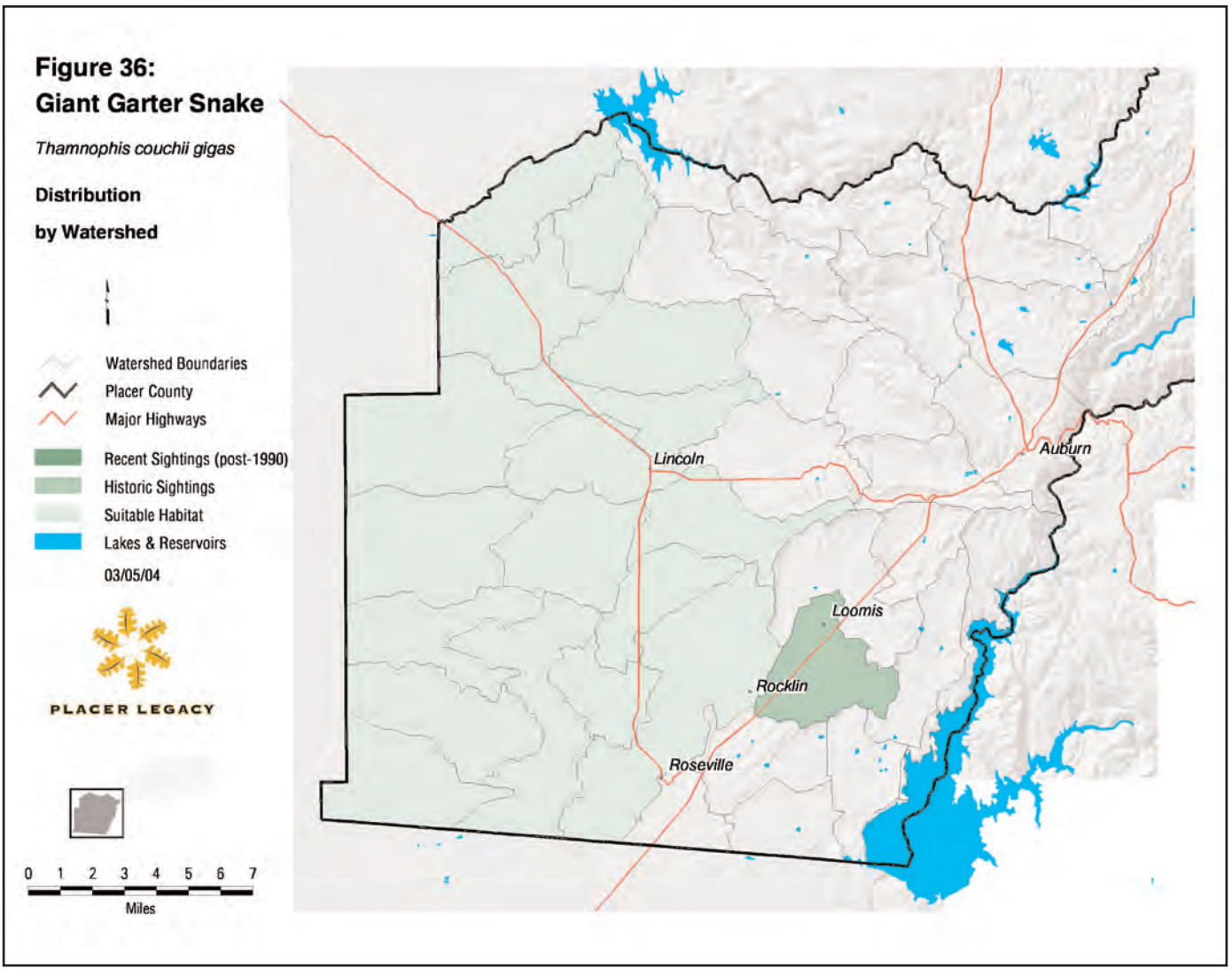
There are no current records of giant garter snake within the Phase I Planning Area (Figure 36). However, suitable habitat occurs in the drainage network associated with agricultural fields in the western section of the county (U.S. Fish and Wildlife Service 1999). Numerous occurrences of giant garter snakes have been reported in the American Basin west of the Placer County line in Sutter County (California Natural Diversity Database 2003).

Population Status and Trends

California

The current distribution and abundance of giant garter snake is reduced and declining due to loss, degradation, and fragmentation of habitat (U.S. Fish and Wildlife Service 1999). Prior to 1970, the species was known from 17 populations (Hansen and Brode 1980). At the time of listing in 1993, 13 of these populations were extant; only three of these populations are currently considered stable and safe from threats.

Giant Garter Snake Jones & Stokes file photo



Placer County Phase I Planning Area

There are no known records of giant garter snakes in the Phase I Planning Area. Consequently, the status of any population that may occur there is unknown.

Natural History

Habitat Requirements

Giant garter snakes inhabit agricultural wetlands and associated waterways. These include irrigation and drainage canals, rice fields, marshes, sloughs, ponds, small lakes, low-gradient streams, and adjacent uplands (U.S. Fish and Wildlife Service 1999). Features of these habitats important to giant garter snakes include:

- sufficient water during the snake's active season (early spring through mid-fall) to maintain an adequate prey base;
- emergent vegetation such as cattails (Typha spp.) and bulrushes (Scirpus spp.) for escape cover and foraging habitat;
- upland habitat with grassy banks and openings to waterside vegetation for basking; and
- adjacent upland areas for cover and refuge from floodwaters during the species' inactive season (Hansen 1980; Hansen 1988; Brode and Hansen 1992; Hansen and Brode 1993).

Giant garter snakes are absent from the larger rivers; from wetlands with sand, gravel, or rock substrates; and from riparian woodland areas lacking suitable basking sites or suitable prey populations (Hansen 1980; Rossman and Stewart 1987; Brode 1988; Hansen 1988; U.S. Fish and Wildlife Service 1999). They inhabit small mammal burrows and other soil crevices above prevailing flood elevations throughout the winter dormancy period (November to mid-March). Giant garter snakes typically select burrows with sunny exposure along south- and west-facing slopes; they also use riprap rock piles along irrigation canals. (U.S. Fish and Wildlife Service 1999.)

Reproduction

Giant garter snakes are ovoviviparous (live bearing). The breeding season lasts from March into May and resumes briefly during September (U.S. Fish and Wildlife Service 1999). Males begin searching for females immediately after emergence from overwintering sites. Females brood young internally and typically give birth to 10–46 young (mean = 23) from late July through early September (Hansen and Hansen 1990).

Foraging Behavior

Giant garter snakes feed primarily on fish and amphibians, taking advantage of pools that trap and concentrate prey (Brode 1988; Hansen 1980; Hansen 1988; Hansen and Brode 1993). Prey species include bullfrog (*Rana catesbeiana*), Pacific chorus frog (*Pseudacris regilla*), carp (*Cyprinus carpio*), mosquitofish (*Gambusia affinis*), and blackfish (*Othodox microlepidotus*) (Fitch 1941; Fox 1952; Cunningham 1959; Hansen 1980; Brode 1988; Hansen and Brode 1993; Rossman et al. 1996).

Dispersal Patterns

No estimates of dispersal distances have been reported for giant garter snake. Newborn giant garter snakes disperse into dense cover immediately after birth and absorb their yolk sacs, after which they begin fending for themselves (U.S. Fish and Wildlife Service 1999). Adults may disperse away from seasonal wetlands or rice fields when they dry up.

Demography

Young giant garter snakes are about 20 centimeters (8 inches) at birth. They typically double in size by 1 year of age (U.S. Fish and Wildlife Service 1999); males usually reach sexual maturity in 3 years and females in 5 years. Sex ratios of adult females to males vary from 1:1 to 2:1, but this variance may be a function of capture methods employed in different studies (Hansen and Brode 1993; Wylie et al. 1997; U.S. Fish and Wildlife Service 1999). Adult females are on average longer and heavier than males; males can reach 820 millimeters (mm) (32.3 inches) in snout-vent length (mean = 665 mm [26.2 inches]) and females reach 1,080 mm (42.5 inches) snout-vent length (mean = 887 mm [34.9 inches]). Males weigh up to 289 grams (10.2 ounces) (mean = 140 grams [4.9 ounces]) and females weigh up to 785 grams (27.7 ounces) (mean 433 grams [15.3 ounces]) (U.S. Fish and Wildlife Service 1999).

There are few population estimates for giant garter snakes. Mark and release studies have produced varied results. Some of these estimates are: 84 snakes in a 2.6-square-kilometer (1-square-mile) area of rice land in the Natomas Basin (Hansen and Brode 1993); 1,000 snakes within 2.6 square kilometers (1 square mile) (U.S. Fish and Wildlife Service 1999); 206 individuals in Gilsizer Slough (1,430 hectares [3,500 acres] (U.S. Fish and Wildlife Service 1999); 132 individuals in the Colusa National Wildlife Refuge (4,500 hectares [11,120 acres]); and 191 giant garter snakes in Badger Creek Marsh (235 hectares [580 acres]).

Longevity

No information is available on the longevity or survival rates of giant garter snake; such estimates are very limited for the genus as a whole. The best survivorship data available for garter snakes are from a study of *T. sirtalis* in northern California. The results of this study showed 1- and 2-year survivorship of neonates to be 28.7% and 16.4%, respectively; yearly survivorship was 50.8%, and annual survivorship of individuals more than 2 years old was only 32.7 % (Rossman et al. 1996).

Sources of Mortality

Giant garter snakes are subject to widespread mortality from habitat loss, increased predation in degraded habitats, vehicular traffic, contamination from pesticides and other toxins, agricultural practices, water maintenance activities, and flooding (U.S. Fish and Wildlife Service 1993, 1999).

Behavior

Home range estimates for giant garter snakes based on radio telemetry data vary with location; estimates averaged 19.0 hectares (47 acres) in Gilsizer Slough (N = 27; range: 0.8–259 hectares [2.0–640 acres]); 53.2 hectares (131 acres) in Colusa National Wildlife Refuge (N = 29; range: 1.3–1,130 hectares [3.2–2,792 acres]); and 9.2 hectares (23 acres) at Badger Creek (N = 8; range: 4.2–82.0 hectares [10.4–202.6 acres]) (U.S. Fish and Wildlife Service 1999).

Movement and Migratory Patterns

Giant garter snakes are most active from early spring through mid-fall; activity is dependent on local weather conditions (Brode 1990; Hansen and Brode 1993). During winter, giant garter snakes are generally inactive, although some individuals may bask or move short distances on warmer days (U.S. Fish and Wildlife Service 1999). During the active season, giant garter snakes generally remain near wetland habitats but can move more than 244 meters (800 feet) from the water (Hansen 1988; Wylie et al. 1997) during the day. Some individuals may move up to 5 miles over a period of several days if the conditions of their habitat become unsuitable (e.g., as a result of flooding) (Wylie et al. 1997).

Ecological Relationships

Giant garter snakes prey on a variety of fish and amphibians available within their habitat; they are in turn prey for raccoons (*Procyon lotor*), striped skunks (*Mephitis mephitis*), opossums (*Didelphis virginiana*), red foxes (*Vulpes vulpes*), gray foxes (*Urocyon cinereoargenteus*), hawks (*Buteo* spp.), northern harriers (*Circus cyaneus*), great egrets (*Ardea alba*), snowy egrets (*Egretta thula*), american bitterns (*Botaurus lentiginosus*), and great blue herons (*Ardea herodias*). Giant garter snakes may coexist with two other species of garter snake: valley garter snake (*T. sirtalis fitichi*) and western terrestrial garter snake (*T. elegans*) (Hansen 1980; Hansen 1986). This coexistence may be possible because of differences in foraging behavior (U.S. Fish and Wildlife Service 1999).

Population Threats

Loss, degradation, and fragmentation of habitat are the primary threats to the viability of giant garter snake populations (U.S. Fish and Wildlife Service 1999). Conversion of wetlands for agricultural, urban, and industrial development has resulted in the loss of more than 90% of suitable habitat for this species in the Central Valley. Degradation of habitat, including maintenance of flood control and agricultural waterways, weed abatement, rodent control, discharge of contaminants into wetlands and waterways, and overgrazing in wetland or streamside habitats, may also cumulatively threaten the survival of some giant garter snake populations (Hansen 1988; Brode and Hansen 1992; California Department of Fish and Game 1992b; Hansen and Brode 1993).

The introduction of nonnative predators, including bullfrog, largemouth bass (*Micropterus salmoides*), and catfish (*Ictalurus* spp.), has been responsible for eliminating many species of native fishes and aquatic vertebrates in the western United States (Minkley 1973; Moyle 1976; Holland 1992). Exotic species have probably had detrimental affects on the giant garter snake through direct predation (Treanor 1983; sensu Bury and Whelan 1984) and competition for smaller forage fish (Hansen 1986; Schwalbe and Rosen 1989; California Department of Fish and Game 1992b).

Toxic contamination, particularly from selenium, and impaired water quality have also been identified as threats to some populations of giant garter snake (Ohlendorf et al. 1986; Saiki and Lowe 1987; U.S. Fish and Wildlife Service 1993). Preliminary studies have documented potential bioaccumulative effects on giant garter snakes or their prey species caused by agriculturally derived contaminants (Saiki et al. 1992, 1993). Disease and parasitism, potentially exacerbated by compromised immune response ability as a result of contaminant exposure, may also pose a threat to this species (U.S. Fish and Wildlife Service 1999).

Western Pond Turtle (*Clemmys marmorata*)

Status

FEDERAL: None.

STATE: None.

OTHER: California Department of Fish and Game Species of Special Concern; USDA Forest Service Region 5 Regional Forester's Sensitive Species.

RECOVERY PLAN: Although a recovery plan has been developed by the State of Washington, no recovery plans have been developed for California or Oregon.

PLACER NCCP/HCP CATEGORY: Class 3.

Systematics

Western pond turtle (*Clemmys marmorata*) is the only species in its genus that occurs in the western United States. Janzen et al. (1997) suggested that southern populations may be sufficiently distinct to warrant recognition as a separate species. However, all populations continue to be classified as western pond turtle, with two recognized subspecies: northwestern pond turtle (*C. m. marmorata*) and southwestern pond turtle (*C. m. pallida*). Genetic research supports the distinctiveness of the two subspecies (Gray 1995; Janzen et al. 1997). The area of the Central Valley of California between the American River drainage and the Transverse Ranges is considered to be a zone of intergradation between the two subspecies (Seeliger 1945; U.S. Fish and Wildlife Service 1999). Accordingly, while some authorities consider the American River Drainage to lie within the range of northwestern pond turtle, this report treats the Placer County populations as *western pond turtle*.

Distribution

North America

Western pond turtle occurs from Puget Sound in Washington south through Oregon and California generally west of the Cascade-Sierra crest and into Baja California. In the Central Valley, western pond turtle historically inhabited the vast permanent and seasonal wetlands of the area, with the Tulare Lake Basin as a major population center (Hays et al. 1999; U.S. Fish and Wildlife Service 1999).

California

Today, northwestern pond turtle occurs in 90% of its historic range in the Central Valley and west of the Sierra Nevada, but in greatly reduced numbers (Jennings and Hayes 1994). It occurs from the Oregon border south to the American Basin in the Central Valley, where it intergrades with southwestern pond turtle.

Placer County Phase I Planning Area

Historical

Western pond turtles occurred historically in suitable habitat throughout the American Basin, including the Placer County Phase I Planning Area (U.S. Fish and Wildlife Service 1999). The CNDDB (2003) lists one current population is at the site of a historic record.

Current

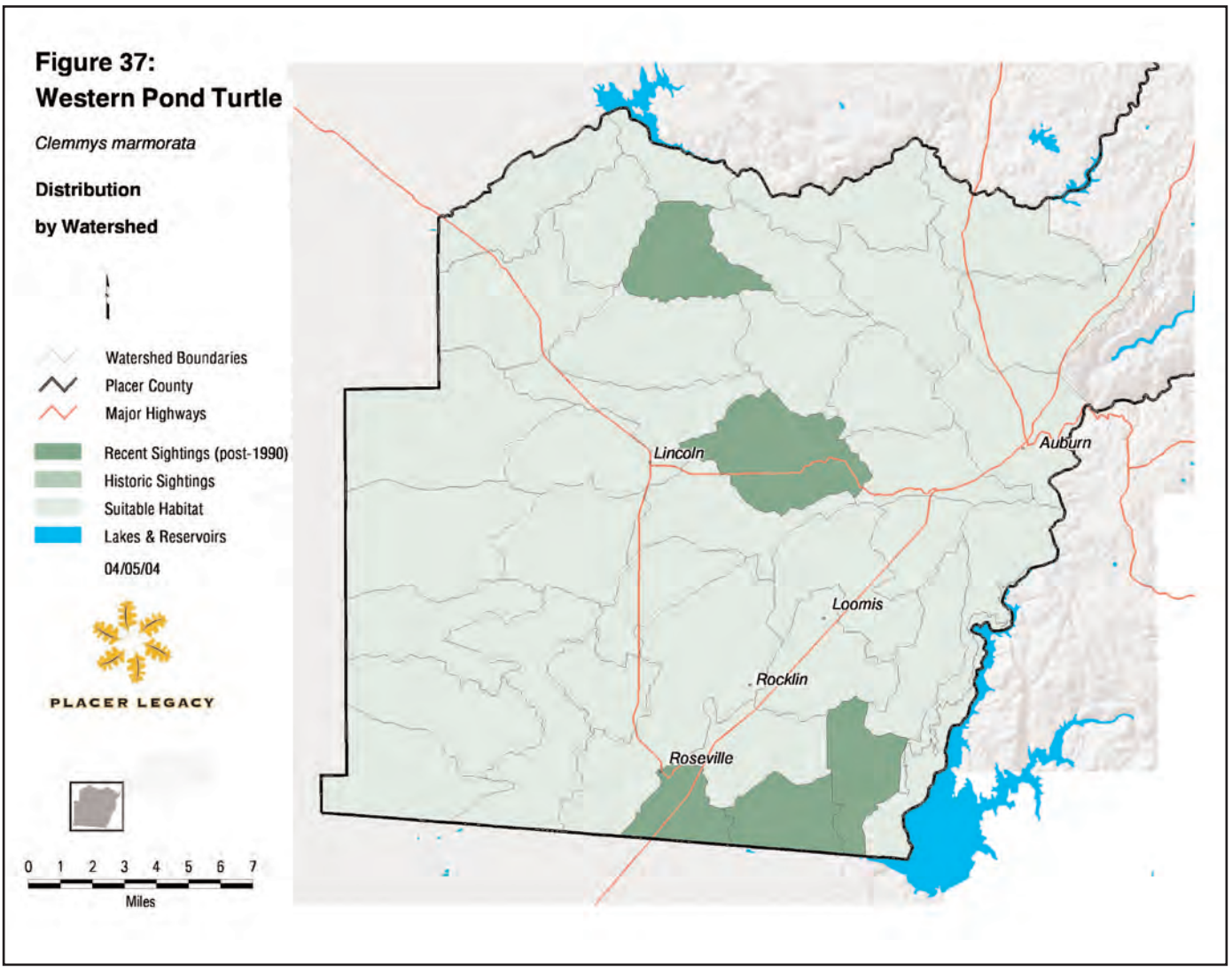
Six occurrences of western pond turtles have been documented in the Placer County Phase I Planning Area and vicinity (1988–2000). This area includes the U.S. Geological Survey 7.5-minute topographic quadrangles for Folsom, Wheatland, Camp Far West, Wolf, Lake Combie, Colfax, Auburn, Pilot Hill, Citrus Heights, Rio Linda, Pleasant Grove, Sheridan, Lincoln, Roseville, Rocklin, and Gold Hill. All six populations are believed to be extant (California Natural Diversity Database 2003; Figure 37).

Population Status and Trends

North America

Western pond turtle was once abundant in California, Oregon, and locally in Washington, but is declining in

Western Pond Turtle Photo by Stephanie Myers



numbers throughout its range, particularly in Washington, northern Oregon, southern California, and Baja California (Holland and Bury 1998; Hayes et al. 1999). Loss, degradation, and fragmentation of habitat are the primary factors contributing to the decline of the species (Hayes et al. 1999).

California

Western pond turtle is declining in California primarily as a result of habitat loss and alteration; more than 90% of California's historic wetlands have been diked, drained, and filled—primarily for agricultural development and secondarily for urban development (Frayer et al. 1989). Commercial harvesting of western pond turtles for food during the 1890s to 1920s is also believed to have contributed significantly to the decline of this species in the San Francisco area and Central Valley (Storer 1930; Hayes et al. 1999). More than 18,000 pond turtles were offered for sale in San Francisco markets, presumably in one year, in the 1890s (Smith 1895).

Placer County Phase I Planning Area

The population status and trends of western pond turtle in the Phase I Planning Area are unknown. The taxon is believed to have been abundant in the area when it supported extensive wetlands (Hayes et al. 1999), but some conversion of former wetlands to agricultural lands has likely resulted in local declines of these populations (Jennings and Hayes 1994).

Natural History

Habitat Requirements

Western pond turtles inhabit a variety of aquatic habitats from sea level to elevations of 1,980 meters (6,500 feet). They are found in rivers, streams, lakes, ponds, wetlands, reservoirs, and brackish estuarine waters. (Holland 1994; Jennings and Hayes 1994.) Western pond turtles use aquatic habitats primarily for foraging, thermoregulation, and avoidance of predators; they

require emergent basking sites, and have been observed to avoid areas of open water lacking them (Holland 1994). Basking sites can include rocks, logs, or emergent vegetation, and are used by the turtles for thermoregulation. Western pond turtles can be found in waters with temperatures as low as 1°C (34°F), and rarely in water with temperatures exceeding 39–40°C (102–104°F) (Jennings and Hayes 1994).

Western pond turtles overwinter in both aquatic and terrestrial habitats. Aquatic refugia consist of rocks, logs, mud, submerged vegetation, and undercut areas along banks. Terrestrial overwintering habitat consists of burrows in leaf litter or soil. The presence of a duff layer seems to be a general characteristic of overwintering habitat. In woodland and sage scrub habitats along coastal streams in central California, most pond turtles leave the drying creeks in late summer and return after winter floods. These turtles spend an average of 111 days at upland refuges that are an average of 50 meters (164 feet) from the creeks (Rathbun et al. 2002).

Upland nesting sites must be dry and often have a high clay or silt fraction. Typically, western pond turtles dig nests on unshaded slopes no steeper than 25°. Gravid females leave drying creeks in June to oviposit in sunny upland habitats, including grazed pastures. Nesting has been reported to occur up to 402 meters (1,391 feet) from water (Jennings and Hayes 1994), but is usually closer, averaging 28 meters (92 feet) from aquatic habitat (Rathbun et al. 2002).

Reproduction and Demography

Western pond turtles first breed at 10–14 years of age (U.S. Fish and Wildlife Service 1999). Most females lay eggs in alternate years. Clutch size ranges from 1 to 13 eggs, with larger females generally laying larger clutches (Holland 1985a, 1991a). Females move inland 12–402 meters (39–1,319 feet) to upland habitat to nest from May through July. Incubation lasts 80–100 days, and the normal hatch success is approximately 70%. Nest predation rates are high and complete failure of nests is common. In southern California, juveniles emerge from the nest in early fall. (Holland 1994.) Most hatchlings overwinter in the nest and move to water in March–April, although some leave the nest in September (Holland 1985a, 1991a, 1991b).

Survivorship in western pond turtles is apparently dependent on age and sex. Hatchlings and first-year juveniles average only 8–12% survivorship; this rate may not increase significantly until turtles are 4–5 years old (U.S. Fish and Wildlife Service 1999). Once the turtles reach adult size, survivorship increases dramatically, with an average adult turnover rate of only 3–5%. Adult males generally have a higher probability of survivorship than adult females, with skewed sex ratios reaching 4:1 (males to females). The apparent cause for this difference is a higher mortality experienced by females from predation during overland nesting attempts (Holland 1991a).

Foraging Behavior

Western pond turtles are omnivorous feeders, opportunistic predators, and occasional scavengers (Holland 1985a, 1985b, Bury 1986). The majority of the diet consists of crustaceans, midges, dragonflies, beetles, stoneflies, and caddisflies, but pond turtles also feed on mammal, bird, reptile, amphibian, and fish carrion. Western pond turtles will eat plant matter and have been observed foraging on willow and alder catkins and on ditch grass inflorescences (Holland 1991b). Partial herbivory in adults may provide an important source of readily available nutrients and some proteins when animal food is unavailable. Adults, especially females, consume a greater percentage of plant material than do juveniles. (Bury 1986.)

Longevity

The maximum recorded age for western pond turtle is 39–40 years, but the expected longevity for this species probably reaches 50–70 years (Holland 1991a).

Sources of Mortality

Western pond turtle is preyed upon by a wide variety of native and introduced predators, including raccoon (*Procyon lotor*), western spotted skunk (*Spilogale gracilis*), northern river otter (*Lontra canadensis*), black bear (*Ursus americanus*), coyote (*Canis latrans*), bullfrog (*Rana catesbeiana*), and largemouth bass (*Micropterus salmoides*) (Moyle 1973; Holland 1991a; Hayes et al. 1999). Bobcat (*Lynx rufus*), Great Blue Heron (A*rdea herodias*), Black-crowned Night-Heron (*Nycticorax nycticorax*), Golden Eagle (*Aquila chrysaetos*), Red-shouldered Hawk (*Buteo lineatus*), and giant garter snake (*Thamnophis gigas*) are also believed to be predators of western pond turtles (Holland 1994).

Prolonged drought, contaminants, disease, and parasites also contribute to mortality in western pond turtle populations (Frye et al. 1977; Hayes et al. 1999).

Behavior

Western pond turtles are not known to be territorial, but aggressive encounters, including gesturing and physical combat (Bury and Wolfheim 1973), are common, and may function to maintain spacing on basking sites and to settle disputes over preferred spots. Competing individuals may push and ram each other, threaten one another with open-mouthed gestures, and occasionally bite one another.

Measured home ranges of western pond turtles average 1 hectare (2.5 acres) for males, 0.3 hectare (0.7 acre) for females, and 0.4 hectare (1 acre) for juveniles (Bury 1972a). Males generally move greater distances than females or juveniles (Bury 1972a), but there is little movement between drainages (Holland 1991b).

Western pond turtles commonly forage during late afternoon or early evening. They also bask intermittently throughout the day in order to maintain a body temperature of 24–32°C (75–90°F). In general, these turtles typically become more active in water that consistently reaches 15°C (60°F) (Jennings and Hayes 1994). Extreme heat is avoided by moving to cooler areas on the bottom of pools.

In some parts of the range, western pond turtles are seasonally active, overwintering from October/November through March/April. However, in the Central Valley and along the California coast they may be active throughout the year. (Holland 1991a.)

Movement and Migratory Patterns

During spring or early summer, females move overland up to 402 meters (1,319 feet) to find suitable sites for egg laying (Hays et al. 1999). Other long-distance movements may occur in response to drying of local water bodies or other factors. These turtles are capable of moving long distances (at least 1 mile overland) to find water.

Ecological Relationships

Introduced species have altered the ecological conditions of many areas inhabited by western pond turtles. Bullfrogs and warmwater fish are significant predators on hatchlings and small juvenile western pond turtles. Sunfish compete for invertebrate prey. Carp can cause turbidity (Lampman 1946), which can influence the densities of zooplankton important in the diet of hatchlings and young turtles (Holland 1985b). Introduced turtles, such as sliders, snapping turtles (*Chelydra serpentina*), and painted turtles, may compete with pond turtles and expose them to diseases for which pond turtles have no resistance (Hayes et al. 1999). In California, Oregon, and Nevada, 17 species of exotic aquatic or semiaquatic turtles have been found in pond turtle habitats (Holland and Bury 1998). Additionally, in ranching areas cattle trample and eat aquatic vegetation that serves as habitat for hatchlings, and they may crush pond turtle nests. Domestic dogs may occasionally mutilate turtles (Hayes et al. 1999).

Population Threats

Numerous factors, including loss, degradation, and fragmentation of habitat; disease; introduced predators and competitors; and other natural and anthropogenic conditions present ongoing threats to western pond turtle throughout 75–80% of its range (U.S. Fish and Wildlife Service 1999; Holland 1991a). Extant wetlands are often indirectly affected by adjacent agricultural practices. Many aquatic habitats (e.g., rice lands) are used to convey and store agricultural water and are consequently subject to changes in the timing and amount of water flow. Many wetlands are channelized and periodically cleaned of aquatic vegetation, rendering them unsuitable for pond turtles. Farming activities conducted to the edge of occupied aquatic habitat may limit or eliminate upland nesting opportunities for pond turtles. Because pond turtles are long-lived, populations may persist in these isolated wetlands long after recruitment of young has ceased (Holland 1991a; U.S. Fish and Wildlife Service 1999).

Changes in the nature and timing of water releases from reservoirs may adversely affect downstream habitat by eliminating or altering basking sites, refugia, foraging areas, and hatchling microhabitat (Holland 1991a; U.S. Fish and Wildlife Service 1999). The reservoirs themselves generally provide poor habitat for turtles because of the lack of emergent aquatic vegetation and basking sites, high levels of recreational disturbance, and the presence of exotic predatory species. Water diversions for agriculture can also have negative impacts on turtle populations by resulting in very low or no flows for miles of stream habitat during summer months. Agricultural diversions have resulted in the elimination of pond turtles from such streams and isolation of turtle populations located in other portions of affected drainages (Holland 1991a).

Additional threats include habitat degradation from cattle grazing; instream and streamside sand and gravel mining operations; removal of basking sites (e.g., logs, snags, and rocks) for aesthetic purposes or to facilitate recreational use; and collection of turtles for food or for the pet trade. Incidental collection of turtles, exposure to diseases from introduced exotic species, introduced predators, indiscriminate shooting, construction of highway barriers in upland nesting/migration corridors, off-road vehicle activity, boat activity, and increased exposure to contaminants are also likely to contribute to population declines (Bury 1972b; Holland 1991a). Finally, extended drought and associated fire can also result in significant mortality of western pond turtles (Holland 1991a).

Northern Harrier (*Circus cyaneus*)

Status

FEDERAL: None.

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Northern harriers breed throughout Alaska and Canada (generally south of the tundra) and through much of the northern Midwest and West. East of the Mississippi River, northern harriers breed south to northern Texas; along the Pacific coast, they breed south to northern Baja California. In winter, northern harriers can be found from southeast Alaska south throughout the continental United States and Mexico to Panama and in portions of the Caribbean (MacWhirter and Bildstein 1996).

California

Northern harriers occur throughout lowland California and they may be recorded at high elevations in fall (Zeiner et al. 1990, MacWhirter and Bildstein 1996).

Placer County Phase I Planning Area

Historical

Grinnell and Miller (1944) and the CNDDB (2003) list no historical records of northern harrier breeding in Placer County.

Current

Northern harriers are known to breed in small numbers throughout western Placer County (Figure 38). This population is greatly augmented by migrants and wintering individuals. (Webb 2003.). During the 2002 Lincoln Christmas Bird Count, 175 harriers were tallied, although there was likely some double counting of individuals (Pandolfino pers. comm.).

Population Status and Trends

North America

In North America, the number of breeding northern harriers has declined over the last 100 years, primarily due to habitat loss (Martin 1989; MacWhirter and Bildstein 1996). Johnsgard (1990) estimated the North American wintering population in 1986 to be 111,500 birds, based on data from the National Audubon Society's Christmas Bird Counts (CBCs). Migration counts in Minnesota and several western U.S. locations suggest a population decline of about 3% per year in these areas since the late 1970s, while counts in areas to the east indicate stability or a slight population increase (MacWhirter and Bildstein 1996).

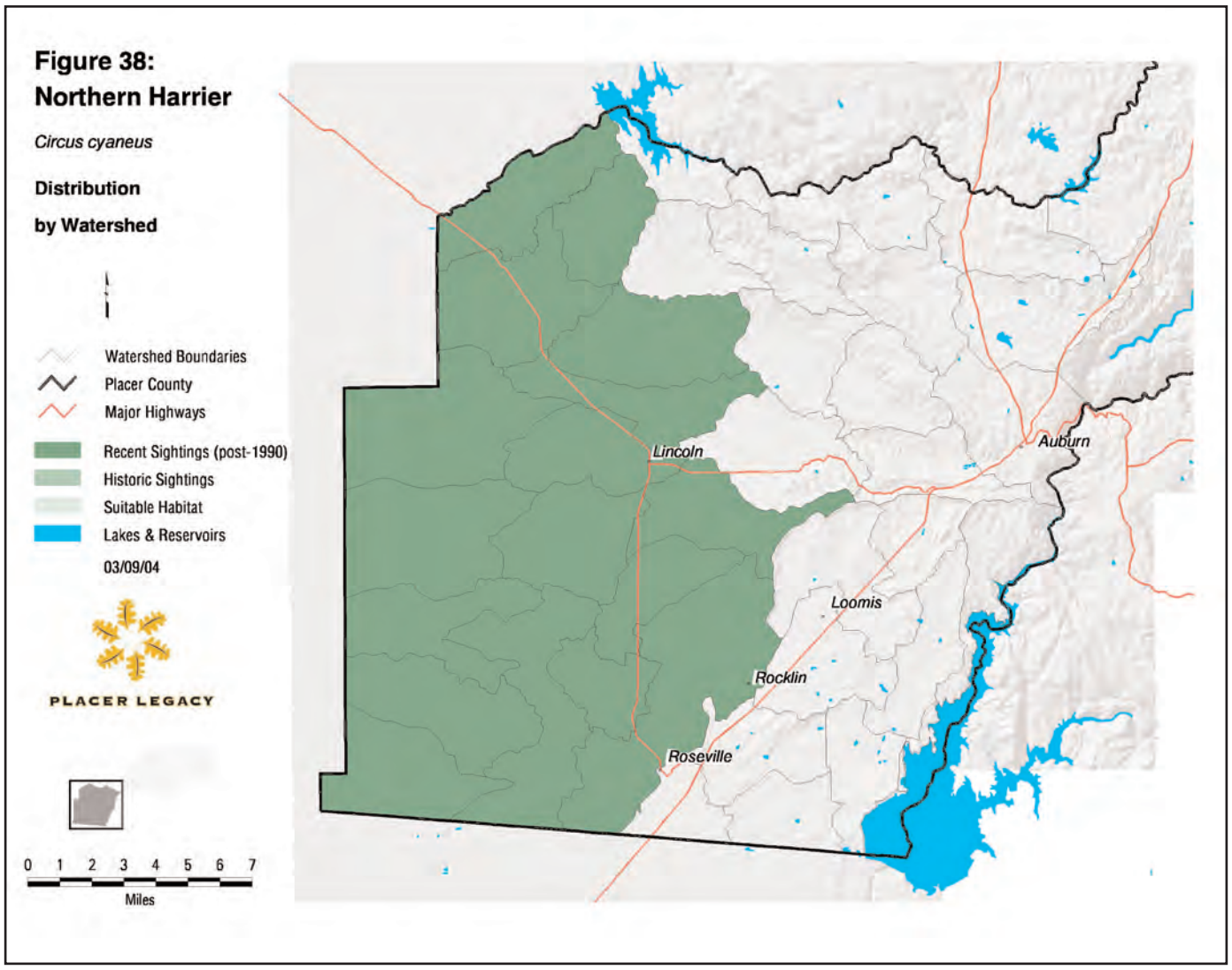
California

Johnsgard (1990) estimated the California population of wintering birds to be 13,200 in 1986 based on CBC data. Analyses of Breeding Bird Survey data for California from 1966 through 2000 indicate a non-significant increasing trend for the Central Valley, the California foothills, and the state as a whole (Sauer et al. 2002).

Placer County Phase I Planning Area

As noted above, northern harriers are known to breed in small numbers in western Placer County and are commonly observed in winter and during migration (Webb 2003). There is no information on population trends in the Phase I Planning Area.

Northern Harrier Photo by Pete LaTourette



Natural History

Habitat Requirements

The northern harrier is a bird of open country, breeding primarily in open grassland and wetland habitats, including brackish (rarely) and freshwater (commonly) marshes, wet meadows, mesic pastures, croplands, prairies, old fields, and shrub-steppe habitats (Johnsguard 1990). Northern harriers are rarely found in densely wooded habitats, although they will breed in riparian woodlands. The highest density breeding populations tend to occur in larger tracts of undisturbed habitat dominated by thick vegetation. (MacWhirter and Bildstein 1996.)

In the Central Valley, northern harriers often attempt to nest in agricultural fields. Summer harvesting regularly destroys nests and kills nestlings. Nesting also occurs in bypass channels (e.g., Willow Slough bypass, Yolo Basin) where strips of semi-natural grassland/wetland habitat exist adjacent to agricultural fields.

In winter, northern harriers commonly perch on the ground, fenceposts, low trees, or rocks. This species typically roosts on the ground, using tall grasses and forbs in wetland habitats or field edges for cover (Zeiner et al. 1990).

Reproduction

The Northern harrier is one of the few birds of prey that is frequently polygynous when ecological conditions permit. Nests are constructed on the ground, typically in dense, low vegetation that provides a visual barrier and cover. In drier habitats, the nest consists of a loose, thin layer of sticks. In wetter situations, nests are larger, more substantial structures. Nests are built by the female and typically consist of grass, reeds, and small sticks. Breeding activity begins in April, concluding in September, with a peak in activity from June to July. A single brood of four to six (rarely up to 12) eggs are incubated by the female. Incubation begins with the last egg and lasts about 29–39 days. The female broods the

young for about 4 weeks while the male provisions the female and young with prey items. Young begin to leave the nest, moving around into the surrounding vegetation, at about 2 weeks of age. The amount of time spent at the nest steadily decreases after this point until fledging. First flight generally occurs at 29–34 days. Young remain in the vicinity of the nest until dispersal. (Zeiner et al.1990; MacWhirter and Bildstein 1996; Baicich and Harrison 1997.)

Dispersal Patterns

Young northern harriers appear to exhibit little to no fidelity to their natal sites or to contribute to the local population in their natal areas. In Wisconsin, less than 1% of banded nestlings were ever recaptured in the study area (MacWhirter and Bildstein 1996.)

Similarly, breeding site fidelity also appears to be low. In Wisconsin, less than 50% of adults banded during the breeding season were detected in the study area in subsequent years, and the species has been described by some as nomadic (MacWhirter and Bildstein 1996). However, a degree of fidelity to wintering sites is indicated by the presence of communal roosts that occur in the same areas year after year.

Longevity

The oldest known northern harrier attained an age of 16 years 5 months (U.S. Geological Survey 2003). The mean age at death of a sample of 114 banded birds was 16.6 months (Keran 1981). Mortality rates prior to the 1950s were estimated to be 59% for first-year birds and 30% for adults (Bildstein 1988).

Sources of Mortality

There is little information on sources of mortality other than predation. Collisions with wind turbines, power lines, automobiles, and other structures have not been reported, but such incidents undoubtedly contribute to mortality rates. Mortality of nestlings from agricultural activities is also likely to be a relatively common occurrence. Known predators of eggs or nestling include coyote (*Canis latrans*), feral dog (*Canis familiaris*), red fox (*Vulpes vulpes*), striped skunk (*Mephitis mephitis*), raccoon (*Procyon lotor*), great horned owl (*Bubo virginianus*), American crow (*Corvus brachyrhynchos*), and common raven (*Corvus corax*) (MacWhirter and Bildstein 1996).

Behavior

Foraging behavior of northern harriers is unique among diurnal North American birds of prey. This species hunts on the wing, coursing over the ground and relying heavily on auditory clues to detect prey. The main prey items are rodents, small birds, reptiles, and frogs. During winter, northern harriers often defend territories against conspecifics, although they often roost communally on the ground. During the breeding season, northern harriers are not strongly territorial except near the nests. (MacWhirter and Bildstein 1996.)

Movement and Migratory Patterns

Northern harrier is a partial migrant, although birds from the northern part of the species' range will migrate long distances (>1,500 kilometers [932 miles]). The fall migration period of northern harriers is more protracted than that of other raptors, lasting over 3 months is some populations. Fall migration generally occurs in September and October. Birds wintering in the United States abandon winter territories and communal roosts in late February and early March. (MacWhirter and Bildstein 1996.)

Ecological Relationships

Northern harriers often compete with other large raptors for food on winter territories. The degree of defensive behavior varies depending on several factors, including the species of the competitor, prey abundance, and the potential for kleptoparasitism (MacWhirter and Bildstein 1996). Northern harriers may also compete with other raptors during the breeding season for food resources, although direct evidence of this is lacking.

Population Threats

Currently, the main threat to northern harrier populations is habitat loss resulting from destruction of marshes, reforestation (in the eastern United States), and agricultural and urban development (del Hoyo et al. 1995). Agricultural practices such as mechanized agriculture, early mowing, overgrazing, and heavy use of insecticides and rodenticides have also contributed to a decline in the suitability of some habitats (MacWhirter and Bildstein 1996).

Bald Eagle
(*Haliaeetus leucocephalus*)

Status

FEDERAL: Threatened. Proposed for delisting by the U.S. Fish and Wildlife Service July 4, 1999 (64 FR 128).

STATE: Endangered.

OTHER: Fully Protected (California Fish and Game Code 3511).

RECOVERY PLAN: *The Pacific Bald Eagle Recovery Plan* (U.S. Fish and Wildlife Service 1986) covers Idaho, Nevada, California, Oregon, Washington, Montana, and Wyoming.

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

North America

The Bald eagle occurs throughout most of North America. The breeding range extends from the Aleutian Islands and Alaska in the north; east through Canada to Labrador; and south to Florida, Baja California, and other scattered locations in northern Mexico (U.S. Fish and Wildlife Service 1999; Buehler 2000). In the contiguous United States, the breeding distribution is concentrated in the Cascade Range of Washington, Oregon, and northern California; the Rocky Mountains; the Great Lakes region; Maine; the Atlantic coast; Florida; the Gulf Coast in Louisiana and Texas; and central Arizona (Buehler 2000). The species' winter range includes coastal Alaska and Canada, southern mainland Canada, and nearly the entire continental United States.

California

Bald eagles breed or winter throughout California except in the desert areas (Zeiner et al. 1990). Most breeding activity occurs in Butte, Lake, Lassen, Modoc, Plumas, Shasta, Siskiyou, and Trinity Counties (Zeiner et al. 1990). California's breeding population of bald eagles is resident year-round in most areas where the climate is relatively mild (Jurek 1988). Bald eagles nest in only a few scattered locations south of the northern Sacramento Valley.

Placer County Phase I Planning Area

Historical

The CNDDB contains no records of breeding bald eagles in Placer County (California Natural Diversity Database 2003). Although Grinnell and Miller (1944) do not mention bald eagles breeding in Placer County, Detrich (1985) cited one and possibly two records prior to 1940 of bald eagles nesting near Lake Tahoe.

Current

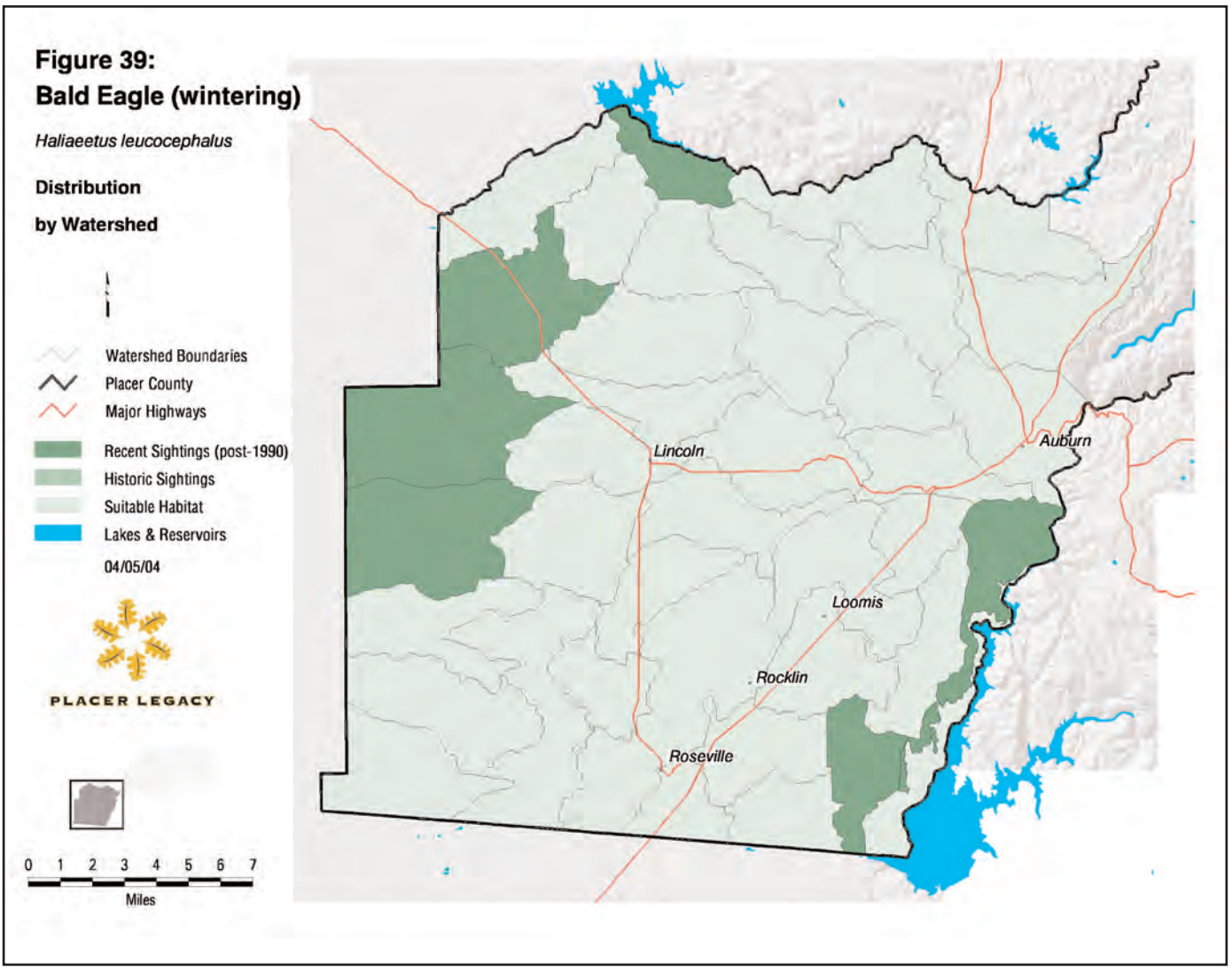
There are currently no bald eagle pairs breeding in Placer County, although individuals are frequently observed during winter (Figure 39).

Population Status and Trends

North America

Since bald eagles were listed throughout the lower 48 states, the species has dramatically increased in numbers and expanded its range. This improvement is a

Bald Eagle Photo by Ted Beedy



direct result of the banning of DDT and other persistent organochlorines, habitat protection, and other recovery efforts. (U.S. Fish and Wildlife Service 1999.) In addition to a constant upward trend in population, productivity data for the past 10 years show that the target for productivity identified in the recovery plan has been met and remains relatively constant.

Most bald eagle population goals set in the recovery plan have been met or exceeded (U.S. Fish and Wildlife Service 1999). In 1994, populations were estimated at approximately 4,450 occupied breeding areas, with 1.16 young produced per occupied area. This estimate reflected a 462% increase over 1974 estimates. In 1998, population estimates showed 5,748 breeding areas with all but two states supporting nesting pairs. Sprunt et al. (1973) estimated that an eagle population requires a rate of 0.7 young per pair per year to be sustainable. In the Pacific Region, the rate has averaged 1 young per pair; accordingly, the population is expected to grow (U.S. Fish and Wildlife Service 1999).

Although the bald eagle recovery is impressive, not all goals have been reached. In the Pacific Region, 28 of 37 (76%) management zones have met population goals (U.S. Fish and Wildlife Service 1999). Eleven of the 28 zones have more than doubled their goals, but the Pacific Region recovery plan states that the goal requires 80% of management zones to meet population goals. This goal may not be reached because not all management zones have preferred habitat. Success rates for breeding areas have exceeded 65% for several years.

California

The California breeding population is continuing to increase in numbers and range. The number of breeding pairs occupying territories was 124 in 1996, 142 in 1997, and 146 in 1998. Productivity has remained high, with the number of "young produced per occupied territory of known success" averaging 1.1 in 1996 and 1997. Productivity declined to 0.97 in 1998 but appears

to have rebounded to 1.1 young per occupied territory of known success in 1999. (California Department of Fish and Game 2000a.)

During 1996, 1997, and 1998, 32 new bald eagle breeding territories were reported in California, resulting in a total of 180 territories known to have been occupied at sometime in the 1990s. The breeding range has expanded from portions of eight of California's 58 counties in 1981 to 27 counties at present.

Placer County Phase I Planning Area

Although bald eagles regularly occur within the Phase I Planning Area during winter, no systematic monitoring of this wintering population has been conducted. The CNDDB contains no records of bald eagles breeding in Placer County.

Natural History

Habitat Requirements

Historically, bald eagles established breeding sites in a variety of habitats in California, including offshore islands; coastal cliffs and pinnacles; and along coastal rivers, interior valley streams and wetlands, and mountain lakes and rivers (Detrich 1985). Nest trees include a variety of hardwoods as well as conifers. Most eagle nesting territories are now found in montane habitat in ponderosa pine and mixed conifer forests (Lehman 1979; Detrich 1985; Jurek 1990).

Bald eagle nest sites are always associated with bodies of water, usually lakes and rivers that support abundant fish, waterfowl, or other waterbird prey. In California, approximately 70% of the breeding eagle population is associated with water bodies larger than 200 hectares (494 acres) (Detrich 1985). Nest trees are usually found within 1,584 meters (5,197 feet) of water and are typically in mature and old-growth conifer stands (Buehler 2000). Nest trees usually have an unobstructed view of a water body and are typically the dominant or co-dominant trees in their surrounding stands (Lehman 1979). Snags and dead-topped live trees are important for perch and roost sites. Lehman et al. (1980) and Anthony et al. (1982) reported that the mean diameter of nest trees in California and Oregon was 104-117 centimeters (41-46 inches) at breast height.

Bald eagles winter along rivers, lakes, or reservoirs that support abundant fish or waterbird prey and that have large trees or snags for perch or roost sites. Bald eagles typically forage in waters less than 500 meters (1,641 feet) from perching habitat. Bald eagles often roost communally during the winter, typically in mature trees or snags that are isolated from human disturbance. Communal night roosting sites are often different from diurnal perch sites. Night roost sites often possess different habitat components than daytime use areas, including day perch sites. While day perches are generally snags or dead-topped trees, night roost groves generally have live trees and a more closed canopy. Night roosts are often in sites that are sheltered from the weather by landforms and in areas of coniferous stands that provide insulation from the weather. (Buehler 2000.)

Reproduction

Bald eagles are monogamous and thought to mate for life unless one member of the pair dies, although this characteristic has not been thoroughly documented. Courtship displays, which can begin as early as September, include the "cartwheel display," in which eagles lock talons and fall toward the ground, letting go at the last moment to avoid colliding with the ground; and the "roller coaster flight," in which eagles fold their wings and dive straight at the ground, swooping up at the last moment. Courtship and nest-building begin up to 3 months prior to egg laying. Clutch size is typically two, although one- and three-egg clutches are common. Incubation lasts approximately 35 days. The young fledge at about 11-12 weeks, but parental care may continue for another 4-11 weeks. Bald eagles reach sexual maturity in 4-5 years. (Buehler 2000.)

Dispersal Patterns

Upon leaving the nesting site, most juveniles migrate a few hundred miles to wintering areas (U.S. Fish and Wildlife Service 1999). However, there is little information on natal dispersal in bald eagles because of the length of time (4–5 years) between fledging and sexual maturity. Most juveniles that were color marked in the greater Yellowstone ecosystem were subsequently found breeding within that system. However, two males nested up to 328 kilometers (204 miles) from their natal sites. Anecdotal information suggests that fidelity to breeding sites in adults is high (Buehler 2000).

Longevity

Bald eagles are relatively long-lived. The age of the oldest wild bald eagle recorded was 28 years, and the oldest captive bald eagle reached 36 years. Studies of juvenile survival across the species' range are generally consistent with a California estimate of 77%. Most estimates of adult survival are in the vicinity of 80%. (Buehler 2000.)

Sources of Mortality

Eggs and nestlings are vulnerable to predation from a variety of birds and mammals, including black-billed magpie (*Pica hudsonia*), American crow (*Corvus brachyrhynchos*), common raven (*Corvus corax*), raccoon (*Procyon lotor*), and black bear (*Ursus americanus*). Few other nonhuman species have either the inclination or the capability to predate on immature or adult bald eagles unless they are compromised by starvation, disease, or other debilitating factors (Buehler 2000).

Behavior

Bald eagles are territorial during the breeding season. Distances to the nearest neighbor and home range sizes are highly variable because of large variations in the dispersion and availability of prey. Estimated densities of bald eagles ranged from 0.08 nest per kilometer (0.13 nest per mile) of shoreline in British Colombia to 0.56 nest per kilometer (1.11 nests per mile) in Alaska. In Oregon, the average inter-nest distance among 8 pairs was 3.2 kilometers (2 miles). (Johnsgard 1990.)

Bald eagles are generalized and opportunistic scavengers and predators (Jurek 1988). The most common prey items on the west coast are fish, waterfowl, and jackrabbits (*Lepus* spp.); various types of carrion, such as fish, mammals, and waterbirds, are also significant components of the diet (Zeiner et al. 1990; Buehler 2000). Bald eagles feed gregariously on abundant prey, such as spawning fish, as well as individually (Zeiner et al. 1990). Diurnal perches used during foraging usually have a good view of the surrounding area and are often the highest perch sites available (Buehler 2000). In general, foraging habitat consists of large bodies of water or free-flowing rivers with abundant fish and adjacent snags and other perches (Zeiner et al. 1990).

Movement and Migratory Patterns

Migration patterns in bald eagles are complex, with variations related to age, location of breeding site, climate, and food availability. Many bald eagles from nesting territories in the northwestern United States migrate south to winter in California (Buehler 2000).

Ecological Relationships

Bald eagles defend territories against conspecifics during the breeding season, and often behave aggressively towards one another during any time of year in disputes over food resources. Bald eagles steal food from ospreys (*Pandion haliaetus*); they also harass and are harassed by golden eagles (*Aquila chrysaetos*), other raptors, and corvids (Corvidae) (Buehler 2000).

Population Threats

Early declines of bald eagle populations resulted from persecution, shooting, egg collection, and habitat loss and disturbance. After 1945, population declines were exacerbated by the widespread use of DDT and other pesticides that led to eggshell thinning and reproductive failure. By 1960, the southern California breeding population was extirpated, and by 1970, the species no longer bred in central California (Detrich 1985). Certain areas within the species' range, including the Great Lakes, Maine, the Columbia River, and southern California, continue to have problems with contamination.

Power line construction and human disturbances can threaten eagle populations in some areas. Because of their large wingspan, bald eagles are particularly susceptible to electrocution on power lines. Bald eagles are also sensitive to human disturbance during the breeding season; human disturbance can cause abandonment or relocation of nest sites (Buehler 2000). In Washington, the vast majority of wintering bald eagles tolerated human activities at a distance of 300 meters (985 feet), and only half tolerated activity at a distance of 150 meters (492 feet) (Stalmaster and Newman 1978; Buehler 2000). The most disturbing human activity appears to be boating, although hiking and car traffic are also significant disturbances (Buehler 2000).

Loss of perching habitat and communal roosting sites can also be a factor in some populations (U.S. Forest Service 2000).

Swainson's Hawk (*Buteo swainsoni*)

Status

FEDERAL: None.

STATE: Threatened.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 1.

Distribution

North America

Swainson's hawks inhabit grasslands, sage-steppe plains, and agricultural regions of western North America during the breeding season and winter in grassland and agricultural regions from Central Mexico to southern South America (Woodbridge et al. 1995a; England et al. 1997; Bradbury et al. in prep.). The North American breeding range extends north from California to British Columbia east of the Sierra Nevada and Cascade Ranges, east to Saskatchewan, and south to northern Mexico. Several disjunct populations occur throughout the breeding range; these include populations in Alaska, western Missouri, and the Sacramento and San Joaquin Valleys of California (England et al. 1997).

California

In California, the nesting distribution includes Great Basin sage-steppe communities and associated agricultural valleys in extreme northeastern California, isolated valleys in the Sierra Nevada in Mono and Inyo Counties, the Sacramento and San Joaquin Valleys, and at least one known isolated breeding site in the Mojave Desert.

The historic breeding distribution also included much of southern California, particularly the inland valleys, where the species was once considered common (Sharp 1902; Bent 1937).

Swainson's hawk are long distance migrants. The majority of the population winters in South America, primarily on the Argentine Pampas. It appears, however, that the California population is distinctive in that it winters in Mexico, Central America, and Columbia, although a few have been discovered spending a portion of the winter in the Sacramento–San Joaquin Delta in the last 5 years (England et al. 1997b).

Placer County Phase I Planning Area

Historical

Suitable habitat for Swainson's hawk is limited to extreme western Placer County, where breeding habitat was probably limited to very large openings in oak woodland/savanna and riparian corridors along Auburn Ravine and Coon Creek.

Current

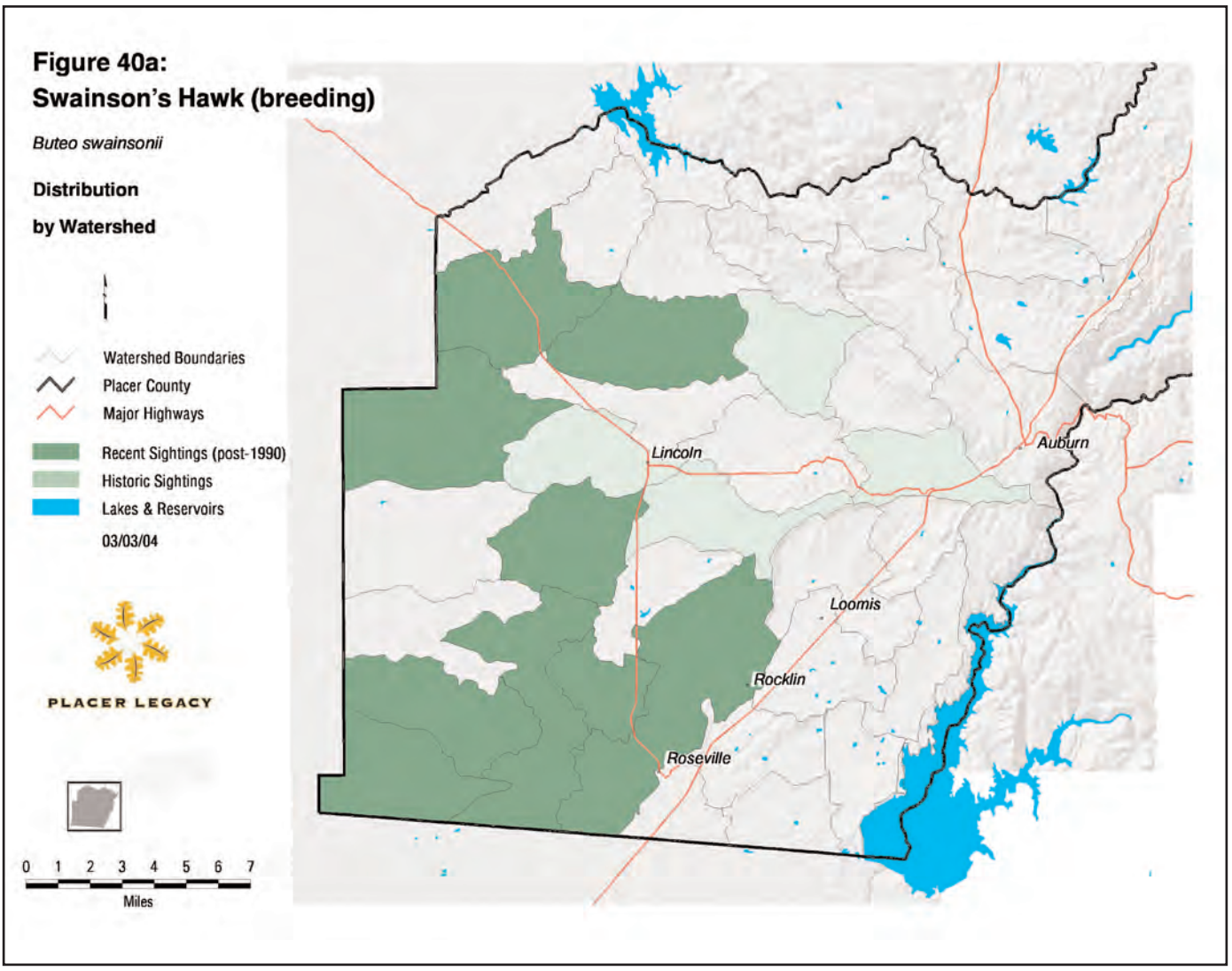
Swainson's hawk has a very patchy distribution in western Placer County. The California Natural Diversity Database (2003) lists two relatively recent records of nesting by this species in the Phase I Planning Area, both from the vicinity of Roseville (Figure 40a). DFG field surveys in 2001 located seven active nests in western Placer County (California Department of Fish and Game 2001b). They forage over a large portion of the Phase I Planning Area (Figure 40b).

Population Status and Trends

North America

There is no comprehensive estimate of the overall size of the Swainson's hawk population in North America; however, as many as 845,000 migrants have been counted over Panama City, Panama, during migration.

Swainson's Hawk Photo by Jim Estep

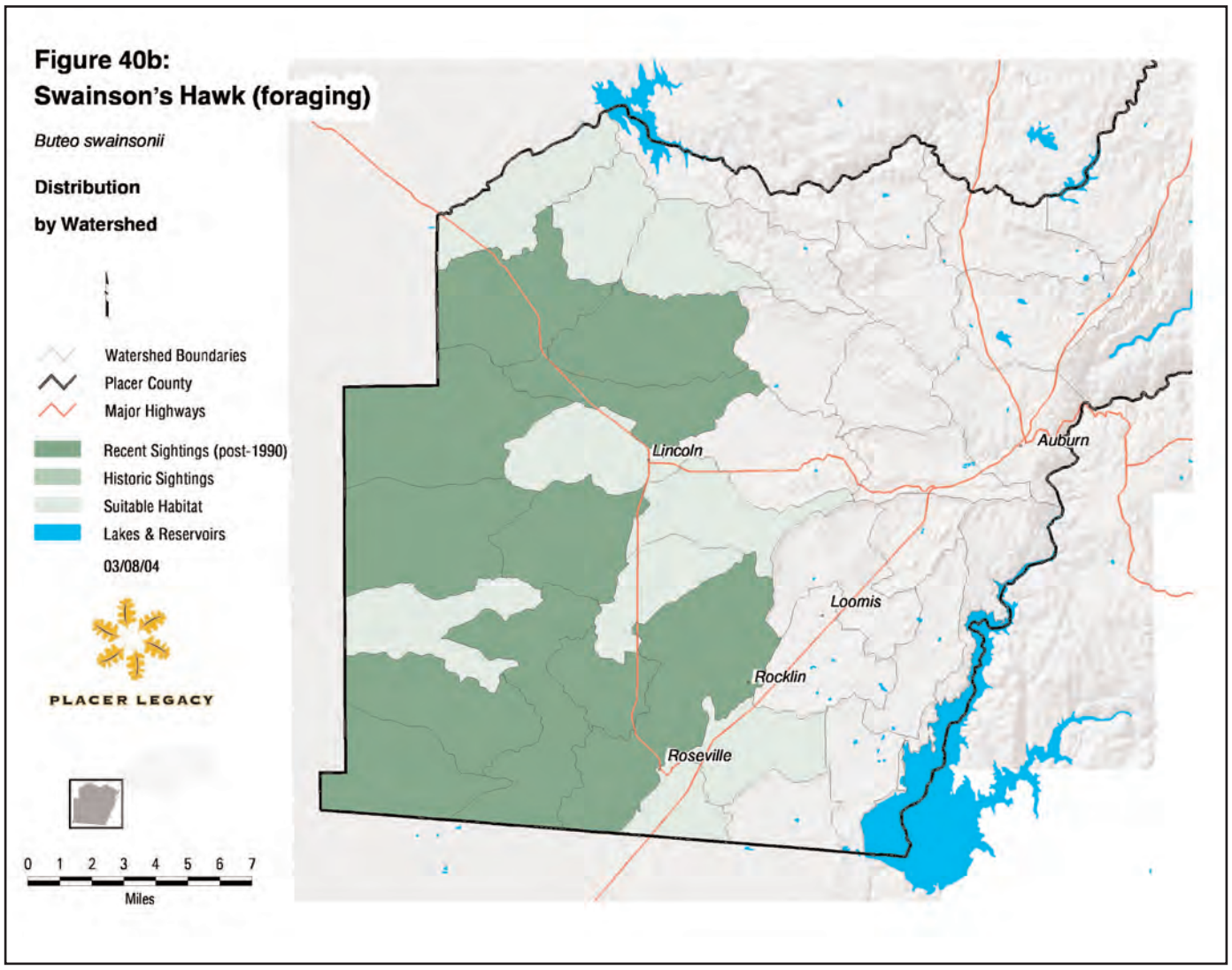


Population declines have been noted in several portions of the species' range, and the current range-wide population is likely reduced from historic times; however, the current overall population trend is undetermined (England et al. 1997.)

California

Early accounts described Swainson's hawk as one of the most common nesting raptors in California, occurring throughout much of lowland California (Sharp 1902). Since the mid-1800s, native habitats have undergone a gradual conversion to agricultural uses. Today, little native grassland remains in the state, and only remnants of the formerly extensive riparian forests and oak woodlands have survived (Katibah 1983). This habitat loss has caused a substantial reduction in the breeding range and the size of the breeding population in California (Bloom 1980; England et al. 1997). Swainson's hawks are also sensitive to habitat fragmentation (Estep and Teresa 1992). The state currently supports between 700 and 1,000 breeding pairs (Swainson's Hawk Technical Advisory Committee file data), or less than 10% of the estimated historic population (Bloom 1980).

Since 1980, based on nesting records alone, populations in California appear relatively stable. However, continued agricultural conversion and practices, urban development, and water development have reduced available habitat for Swainson's hawk throughout its range in California, thereby potentially contributing to a long-term declining trend. The status of populations, particularly with respect to juvenile survivorship, remains unclear.



Placer County Phase I Planning Area

Information on Swainson's hawk in western Placer County is limited. There are seven relatively recent records of Swainson's hawk nesting in western Placer County based on 2001 DFG surveys (California Department of Fish and Game 2001b). There is no information on trends in Swainson's hawk populations in western Placer County.

Natural History

Habitat Requirements

In California, Swainson's hawk habitat generally consists of large, flat, open, undeveloped landscapes that include suitable grassland or agricultural foraging habitat and sparsely distributed trees for nesting (England et al. 1997).

Swainson's hawks usually nest in large, native trees such as valley oaks (*Quercus lobata*), cottonwoods (*Populus fremontia*), and willows (*Salix* spp.), although nonnative trees such as eucalyptus (*Eucalyptus* spp.) are also used. Nests occur in riparian woodlands, roadside trees, trees along field borders, isolated trees, small groves, trees in windbreaks, and the edges of remnant oak woodlands. In some Central Valley locales, urban nest sites have been recorded (England et al. 1995). Stringers of remnant riparian forest along drainages contain the majority (87%) of known nests in the Central Valley (Schlorff and Bloom 1984; England et al. 1995). Nests are constructed using materials from the nest tree or nearby trees, are up to 60 centimeters (24 inches) in diameter, and are usually constructed as high as possible in the tree, providing optimal protection and visibility from the nest (England et al. 1997).

Populations in the Great Basin often use juniper trees (*Juniperus* sp.) for nesting (England et al. 1997), and at least three known nest sites in the Mojave Desert are in Joshua trees (*Yucca brevifolia*) (California Natural Diversity Database 2003).

Nesting pairs in California are highly traditional in their use of nesting territories and nesting trees. Many nest sites in the Sacramento Valley have been occupied annually since 1979 (Estep unpublished data), and banding studies conducted since 1986 confirm a high degree of nest and mate fidelity (Estep in preparation).

Swainson's hawks require wide-open landscapes for foraging. Historically, the species used grass-dominated and desert habitats throughout most of lowland California. Over the past century, conversion of much of the historic range to agricultural uses has shifted the nesting distribution into agricultural areas that mimic grassland habitats or otherwise provide suitable foraging habitat. Suitable agricultural crop patterns include a mixture of hay, grain, and row crops with low-lying vegetation that support adequate rodent prey populations (England et al. 1997).

Under optimal conditions, individual nesting pairs require a minimum of approximately 300 hectares (741 acres) of suitable foraging habitat; however, foraging ranges are geographically and temporally variable and are dependent largely on cover type and phenology and their relationship to prey availability (Fitzner 1978; Bechard 1982; Estep 1989; Babcock 1995).

Reproduction

In the Central Valley, Swainson's hawks arrive on the breeding grounds from early March to early April, significantly earlier than most other populations. Pair bonding begins immediately and involves courtship displays, reestablishment of territorial boundaries, and nest construction or repair. One to four eggs are usually laid in early to mid-April, and incubation continues for 34-35 days until mid-May, when young begin to hatch. The brooding period typically continues through early to mid-July when young begin to fledge (England et al. 1997). Nestlings fledge on average at 43 days (range 38-46 days) (Olendorff 1973; Fitzner 1978). Studies conducted in the Sacramento Valley indicate that one or two (occasionally three) young typically fledge from successful nests, with an average of 1.6 young per successful nest (England et al. 1995; Estep in preparation). After fledging, young remain near the nest and are dependent on the adults for approximately 4 weeks, after which they permanently leave the breeding territory (Anderson et al. in preparation). By mid-August, breeding territories are no longer defended, and Swainson's hawks begin to form premigratory communal groups.

Dispersal Patterns

Woodbridge et al. (1995b) noted an average dispersal distance of 8.8 kilometers (5.5 miles) between natal sites and subsequent breeding sites in northeastern California. In the Sacramento Valley, two birds banded as nestlings and subsequently relocated as breeding adults nested within 3.5 kilometers (2.2 miles) of their natal site (Estep 1989). Much greater dispersal distances from natal sites have been observed in other parts of the range, most notably distances up to 310 kilometers (193 miles) in Saskatchewan (Houston and Schmutz 1995).

A high degree of nest site fidelity has been noted in Swainson's hawks in California. Individuals often use the same nest, the same tree, or a nearby tree in subsequent years. In the Sacramento Valley, mean inter-territory adult movement was approximately 100 meters (328 feet). (Estep in preparation.) Less nest site fidelity was noted in northeastern California, where mean inter-territory movements between 1984 and 1994 were 2.2 kilometers (1.4 miles) (Woodbridge et al. 1995b).

Longevity

Very limited data are available on Swainson's hawk survivorship. In northeastern California, the mean age for hawks banded as nestlings in 1980-1992 and observed in 1993-1994 was 8.2 years (N = 36) (Woodbridge et al. 1995b). In the Sacramento Valley, the mean age for hawks banded as nestlings in 1980 and observed in 1988-1995 was 8.8 years (N = 5); the oldest was 13 years (Estep in preparation).

Sources of Mortality

There is no information on predation of adult Swainson's hawks; however, nestlings are susceptible to predation by great horned owls (*Bubo virginianus*), American crows (*Corvus brachyrhynchos*), and various mammalian predators (Dunkle 1977; Woodbridge 1991; Estep in preparation). Large die-offs of adult birds have been documented in Argentina on the wintering grounds following large-scale applications of insecticides (Woodbridge et al. 1995a).

Behavior

There are no data available on the size or characteristics of breeding territories; however, it has been noted that Swainson's hawks aggressively defend the area immediately surrounding nest sites (Fitzner 1978; Rothfels and Lein 1983; Janes 1984). Outside this relatively small area they appear more tolerant, and often forage communally with conspecifics and other buteos (Estep 1989; England et al. 1997). Once young have fledged, adults begin to form communal foraging and premigratory groups and exhibit little territorial behavior.

In California, home ranges are dependent largely on crop patterns and phenology, and they exhibit substantial annual and seasonal variations. Reported mean home ranges in the Central Valley range from 2,760 hectares (6,820 acres) (Estep 1989) to 4,038 hectares (9,978 acres) (Babcock 1995). In portions of the species' range where there is less dependence on agricultural habitats, reported home ranges are smaller (Fitzner 1978; Anderson 1995).

During the breeding season, Swainson's hawks feed primarily on small rodents, including voles (*Microtus* sp.), deer mice (*Peromyscus* sp.), house mice (*Mus musculus*), and pocket gophers (*Thomomys* spp.). Other, less frequent food items include reptiles, birds, and insects. Swainson's hawks typically forage in large fields that support low vegetative cover (to provide access to the ground) and provide the highest densities of prey (Bechard 1982; Estep 1989). In agricultural regions, these habitats include fields of hay and grain crops; certain row crops, such as tomatoes and sugar beets; 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).

Swainson's hawks are open-country hunters. The usual foraging technique involves searching for prey from a low-altitude soaring flight approximately 30-90 meters (100-300 feet) above the ground and attacking prey by stooping toward the ground (Estep 1989). Occasionally, Swainson's hawks hunt from a perch (e.g., fencepost or utility pole). In agricultural habitats, foraging ranges are highly variable depending on crop patterns and crop phenology (Bechard 1982; Estep 1989). Seasonal and annual foraging ranges are dependent on changes in vegetative height and density that fluctuate with the pattern of crop maturity and harvest.

Throughout their range, Swainson's hawks are known to exploit prey made available through ground-disturbing activities, particularly in agricultural areas. Swainson's hawks are regularly observed on the breeding and wintering grounds hunting behind farm machinery (Estep 1989; Bradbury et al. in preparation). Bent (1937) first reported this phenomenon in southern California, and Caldwell (1986) later measured prey capture success.

Movement and Migratory Patterns

In California, Swainson's hawks begin their fall migration from late August to mid-September (Bloom 1980; Estep 1989; England et al. 1997). Satellite radiotelemetry studies from 1995 to 2001 have identified migratory routes, timing, and wintering grounds (Woodbridge et al. 1995a; Bradbury et al. in preparation). According to these and other telemetry studies, all but the Central Valley population migrates along the eastern edge of Mexico through Central and South America and winters in the pampas region of Argentina. Unlike other populations of Swainson's hawk, the Central Valley population winters primarily in Central Mexico and, to a lesser extent, throughout portions of Central and South America. (Bradbury et al. in preparation). In California, breeding adults arrive on their nesting territory from approximately early March to early April. Courtship and nest construction begin immediately upon arrival.

Ecological Relationships

Swainson's hawks are territorial during the breeding season; however, away from the nest sites adults are more tolerant of conspecifics and other raptors. During the prenesting period, adults are highly aggressive around the nest as they reestablish their territorial boundaries. During communal foraging events and from postfledging through migration and wintering periods, adults are gregarious and tolerate conspecifics as well as other raptor species (Fitzner 1978; Estep 1989; England et al. 1997). Because Swainson's hawks generally arrive on the breeding grounds later than other sympatric buteos, they are often engaged in congeneric battles over control of nest sites.

Population Threats

The loss of agricultural lands and native grasslands to various residential and commercial developments is a serious threat to Swainson's hawks throughout California. Additional threats are habitat loss caused by riverbank protection projects; conversion from agricultural crops that provide abundant foraging opportunities to crops such as vineyards and orchards, which provide fewer foraging opportunities; shooting; pesticide poisoning of prey animals and hawks on wintering grounds; competition from other raptors; and human disturbance at nest sites (California Department of Fish and Game 2000b).

Ferruginous Hawk (*Buteo regalis*)

Status

FEDERAL: Species of Concern.

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Ferruginous hawks breed in western North America from southern Saskatchewan and Alberta south to central New Mexico and Arizona, east through the Great Plains to western Kansas, and west through the Great Basin in central Nevada and Oregon. Their winter range includes most of California and the southwestern United States east through Texas and south through central Mexico. (Bechard and Schmutz 1995.)

California

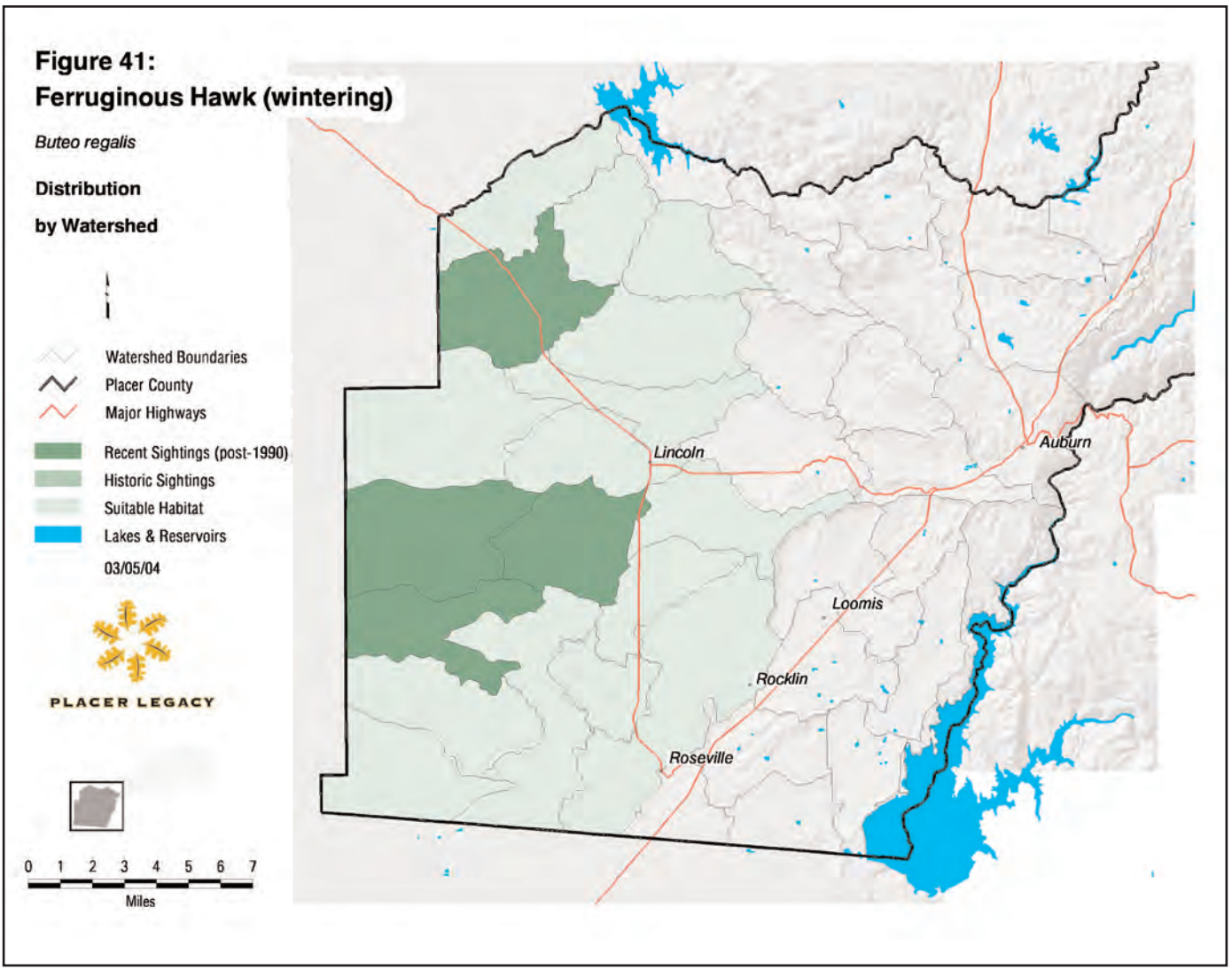
In California, ferruginous hawk is an uncommon winter resident and migrant at lower elevations in open grassland and agricultural habitats in the Modoc Plateau area, Central Valley, and Coast Ranges (Zeiner et al. 1990). It is a more common winter resident in southwestern California (Garrett and Dunn 1981). Garrison (1990) reported significant winter occurrences in the Central Valley and along the central and north coasts. The species appears to be a more casual visitor in the north interior and Sierra Nevada regions (Garrison 1990).

Placer County Phase I Planning Area

Historical

There is no information on historic population size and distribution of ferruginous hawks that is specific to the Placer County Phase I Planning Area. Grinnell and Miller (1944) indicated that the species was "formerly abundant, still more or less common locally" in the state, possibly suggesting that the species had already declined by that time.

Ferruginous Hawk Photo by Mike Danzenbaker



Current

Webb (2003) indicated that ferruginous hawks occur regularly in small numbers on the valley floor portions of western Placer County (Figure 41). Ferruginous hawks were detected regularly on more than 97% of the National Audubon Society's Christmas Bird Count (CBC) areas in the Central Valley between 1980 and 1997 (Hunting 2003).

Population Status and Trends

North America

Olendorff (1993) estimated that the entire North American population of ferruginous hawks was between 5,842 and 11,330 individuals, although Schmutz et al. (1992) estimated the Great Plains portion of the population alone to be approximately 14,000 individuals. Ferruginous hawks are thought to be declining in numbers, with the best-documented declines occurring at the northern edge of the species' range in Canada (Houston and Bechard 1984; Schmutz 1984; Bechard and Schmutz 1995) and in Utah (Woffinden and Murphy 1989).

California

There is no reliable information regarding the size of California's wintering ferruginous hawk population. Garrison (1990) noted from CBC data that the species wintered throughout the state, with the greatest number of observations in the south coast, Central Valley, central coast, and north coast regions. However, no statewide estimate could be derived from these surveys. These data indicate an increase in the number of birds detected each year, but this is probably due to an increase in the number of observers and the number of CBC survey routes counted.

Placer County Phase I Planning Area

Ferruginous hawks are considered rare on the valley floor portion of Placer County, where they occur regularly but in relatively low numbers during winter and in migration (Webb 2003). There is no information on population trends in Placer County.

Natural History

Habitat Requirements

Ferruginous hawks are open-country raptors that occur in open, flat, and rolling terrain in grassland or shrub-steppe regions and avoid high elevations, forests, and canyons (Bechard and Schmutz 1995). They forage in dry grassland and agricultural areas, sagebrush flats, desert scrub, and sparse piñon-juniper habitats (Schmutz and Fyfe 1987; Harlow and Bloom 1989). Nesting occurs most often on rock outcrops and cliffs, in isolated trees or groves of trees, and in sparse riparian woodlands in grassland and shrub-steppe habitats (Smith and Murphy 1973; Woffinden 1975; Lokemoen and Duebbert 1976; Cottrell 1981; Roth and Marzluff 1989; Bechard and Schmutz 1995). However, ferruginous hawks also use a variety of other substrates for nesting; these include large shrubs, utility poles, and haystacks. Additionally, unlike other *Buteo* species, ferruginous hawks nest on the ground when elevated sites are absent. (Bent 1937; Olendorff 1973; Call 1978; Bechard and Schmutz 1995.) Nesting areas typically occur within or adjacent to open foraging areas. Olendorff (1993) examined 2,119 nests throughout the range and found that 49% were in trees and shrubs, 21% were in cliffs, 12% were in utility structures, and 10% were on ground outcrops.

Ferruginous hawks also inhabit open terrain during the winter (Bechard and Schmutz 1995). West of the Rocky Mountains, ferruginous hawks use grasslands and arid areas, particularly where pocket gophers (*Thomomys* spp.), ground squirrels (*Spermophilus* spp.), rabbits (*Lepus* spp., *Sylvilagus* spp.), or prairie dogs (*Cynomys* spp.) are abundant. Ferruginous hawks also winter near cultivated fields that support populations of pocket gophers and other small rodents. On the wintering grounds, ferruginous hawks sometimes form communal roosts of up to 24 birds (Olendorff 1993). Roost sites include trees, cliff ledges, stream banks, large shrubs, and utility structures. Unlike most other hawks, ferruginous hawks frequently perch on the ground.

Reproduction

Pair formation in ferruginous hawks varies by geographic region; in Utah, pair formation begins between February and March (Smith and Murphy 1973). Both members of the pair usually select the nest site, and nests from previous years are often visited during nest site selection (Bechard and Schmutz 1995). Nest building generally begins in March in the southern portions of the species range (e.g., Utah and Colorado) and in April in the northern portions (e.g., North Dakota, Alberta, Saskatchewan) (Schmutz et al. 1980). Nest materials may include sticks, sagebrush stems, debris, and cow manure (Woffinden and Murphy 1982; Bechard and Schmutz 1995). Ferruginous hawks produce one brood per season. Average clutch size is two to four eggs, but clutches can contain from one to eight eggs, depending on local prey availability (Smith and Murphy 1978; Smith et al. 1981). Eggs are laid approximately 2 days apart, and incubation begins when the first egg is laid. Incubation generally lasts 32–33 days (Palmer 1988). Fledglings typically leave the nest at 38–50 days (Konrad and Gilmer 1986) and remain dependent on their parents for several weeks after fledging (Bechard and Schmutz 1995).

Dispersal Patterns

Little information is available on natal dispersal. However, Gosset (1993, cited in Bechard and Schmutz 1995) reported on 81 recoveries of young birds recovered within 90 days of banding; of these, 48 were recovered within 25 kilometers (15.5 miles) of their natal nest site, while the remaining 33 had dispersed as far as 1,700 kilometers (1,056 miles).

Other than a report of an 8-year-old male being captured at the same nest site in two consecutive years, there is no information on breeding site fidelity (Bechard and Schmutz 1995).

Longevity

Based on banding data, Schmutz and Fyfe (1987) estimated a first-year mortality rate of 65%, while Woffinden and Murphy (1989) estimated an adult mortality rate of 25% based on reoccupancy of nest sites. The oldest known wild ferruginous hawk was 17 years 11 months (U.S. Geological Survey 1993).

Sources of Mortality

Ferruginous hawk nestlings and adults are occasionally predated upon by great horned owl (*Bubo virginianus*) (Leslie 1993). American crow (*Corvus brachyrhynchos*) and common raven (*Corvus corax*) occasionally take eggs and nestlings. Ground predators such as coyotes (*Canis latrans*), badgers (*Taxidea taxus*), and foxes (*Vulpes* spp.) may also be an important source of nestling mortality in some areas. Other sources of mortality include shooting, vehicle collisions, and collisions with high-tension wires or towers. (Bechard and Schmutz 1995.)

Behavior

The spatial requirements of ferruginous hawks during the nonbreeding season have not been reported. In winter, ferruginous hawks overlap considerably in their use of foraging and roosting habitat. They often roost communally in large numbers and forage on the ground in groups in which individuals are spaced less than 0.9 meter (3 feet) apart. Winter territoriality has not been reported (Bechard and Schmutz 1995). Mean home range sizes of 5.9 and 7.6 square kilometers (2.3 and 2.9 square miles) have been reported in Utah and Idaho, respectively (Wakely 1978; McAnnis 1990).

Ferruginous hawks hunt from perches or from mounds on the ground. Adults and juveniles primarily consume mammals and prefer only a few species of prey. West of the Continental Divide, ferruginous hawks forage primarily on jackrabbits or cottontail rabbits; ground squirrels, pocket gophers, and prairie dogs are the primary prey species east of the divide (Olendorff 1993). Ferruginous hawks may also occasionally take birds, amphibians, reptiles, and insects (Bechard and Schmutz 1995). Prey selection is apparently related to prey abundance and availability (Steenhof and Kochert 1985).

Movement and Migratory Patterns

Northern populations of ferruginous hawks are completely migratory, while more southern populations may move short distances or not at all. More information on movement patterns of southern populations is needed. Fall migration begins in August or September, with birds returning to the breeding grounds in late February through April, depending on location. (Bechard and Schmutz 1995.)

Ecological Relationships

Ferruginous hawks harass, but are more often harassed by, other *Buteo* species. Where nest sites are limited, ferruginous hawks may experience reduced reproductive success due to harassment by later arriving Swainson's hawks (*Buteo swainsonii*) (Leslie 1993). Populations and reproductive success may fluctuate in response to changes in populations of prey species (White and Thurow 1985; Steenhof and Kochert 1985).

Population Threats

Population declines have been attributed to habitat degradation and loss on the breeding and wintering grounds (Bernard and Schmutz 1995). Factors contributing to habitat degradation include cultivation, grazing, fire, and control of small mammal prey species (Olendorff 1993). Habitat loss has been caused primarily by development and land use changes. In areas where ferruginous hawks subsist on ground squirrels, poisoning of ground squirrels for agricultural or levee maintenance may reduce hawk numbers through secondary poisoning.

Rough-legged Hawk (*Buteo lagopus*)

Status

FEDERAL: None.

STATE: None.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Rough-legged hawk is pan-boreal in distribution, breeding in taiga and tundra habitats in both the new and old worlds. In North America, rough-legged hawks breed in arctic and subarctic habitats in Canada and Alaska. Breeding occurs as far west as the Aleutian Islands and as far east as Newfoundland. (Bechard and Swem 2002.)

Rough-legged hawks migrate across the boreal forest to winter in open habitats in southern Canada and across most of the continental United States except the southeast and coastal portions of the southwest (Bechard and Swem 2002).

California

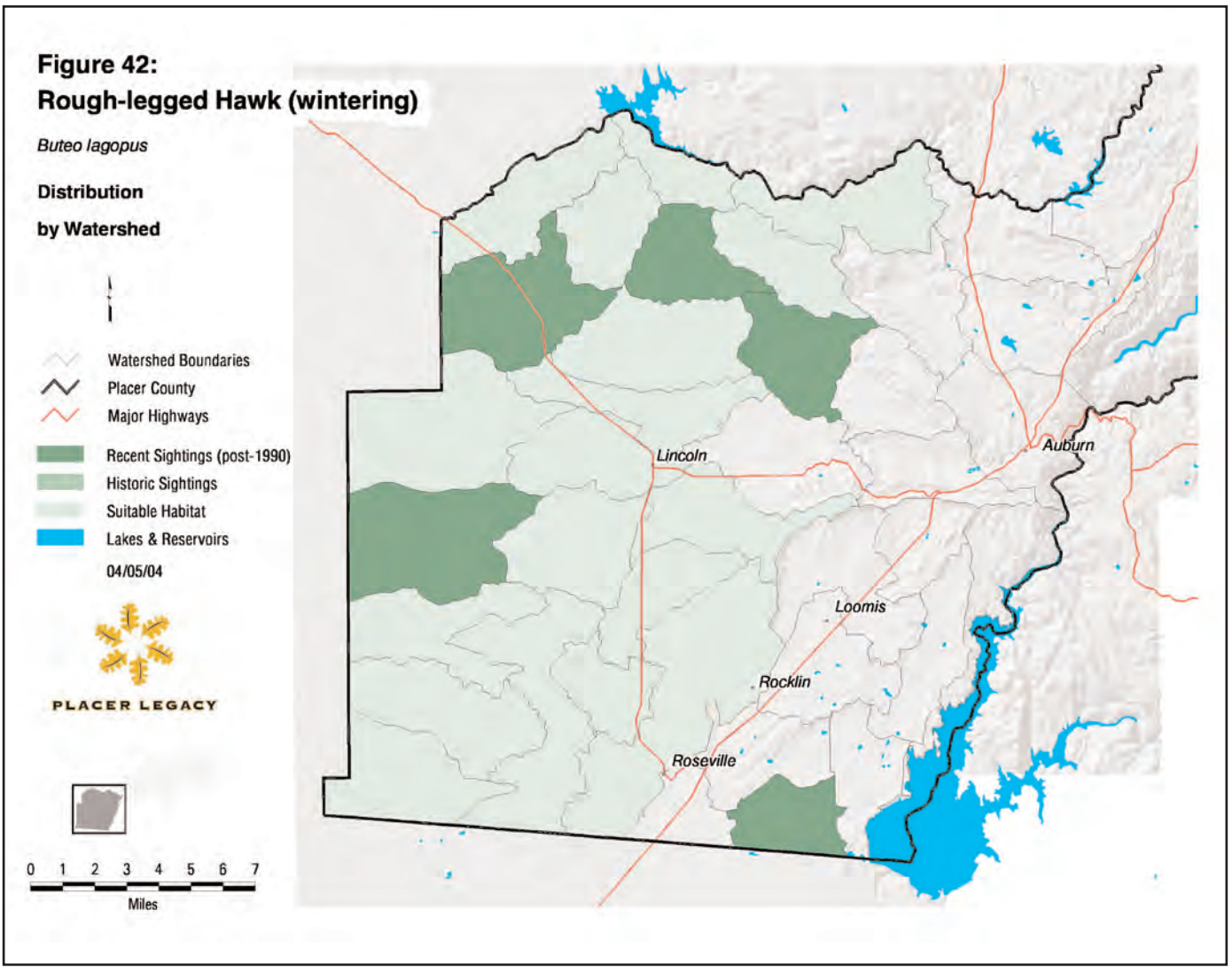
Rough-legged hawks do not breed in California. In winter, the species occurs throughout much of the state but appears to be most common in the northern portions, particularly along the east side of the Sierra Nevada (Garrison 1993). The abundance and distribution of this species in California during winter is strongly influenced by climate and food availability (Bechard and Swem 2002). Garrison (1993) concluded from the Audubon Society's Christmas Bird Count (CBC) data that rough-legged hawks were most abundant in the North Interior, North Sierra, South Sierra, and Central Valley CALVEG regions (Parker and Matyas 1979).

Placer County Phase I Planning Area

Historical

There is no information available on the historic distribution of rough-legged hawks in Placer County. The easternmost portion of the Phase I Planning Area is within the North Sierra CALVEG region.

Rough-legged Hawk Photo by Bill Schmoker



Current

Rough-legged hawks regularly occur in small numbers on the valley floor in western Placer County (Webb 2003; Figure 42).

Population Status and Trends

North America

Rough-legged hawk may be one of the most abundant raptor species in the world (Ehrlich et al. 1988; Bechard and Swem 2002). Although no historical data on rough-legged hawk populations are available, North American populations are probably stable (White 1994). There is no evidence for any change in breeding populations (Palmer 1988), although evaluation of population changes can be complicated by pronounced cycles in breeding and wintering populations that may be related to food resources. The North American wintering population south of the Canadian provinces and Alaska was estimated to be approximately 49,600 birds in 1986 based on CBC data (Johnsgard 1990). CBC data reflected declines from 1966 to 1988 in Delaware and in the 1990s in New Jersey (Walsh et al. 1999; Hess et al. 2000).

California

The California rough-legged hawk population has been found to follow a 3–4 year cycle, although this trend was not statistically significant in eight geographic regions. There is also some evidence to suggest that populations in California increased between 1950 and 1989; this possibility was based on CBC data, although the conclusions of these data are suspect due to other confounding factors. (Garrison 1993.)

Placer County Phase I Planning Area

As noted above, small numbers of rough-legged hawks occur regularly during the winter in western Placer County (Webb 2003). There is no evidence that populations of this species are declining or increasing in western Placer County.

Natural History

Habitat Requirements

Rough-legged hawks breed in taiga and tundra habitats at arctic and subarctic latitudes. Habitats used for nesting include boreal forest, boreal forest–tundra ecotones, treeless tundra, and uplands and alpine regions. Nesting occurs along the coast, along inland river valleys, and in other open country with cliffs, bluffs, or other substrates suitable for nest placement. Nests are typically built on cliffs, ledges, rock outcroppings, escarpments, and eroded riverbanks, although columnar rocks, artificial structures, and trees are also occasionally used (Palmer 1988; Johnsgard 1990; Bechard and Swem 2002).

Rough-legged hawks forage primarily in open areas such as wet meadows, bogs, marshes, riparian areas, pastures, and shrubland and grassland uplands. In winter, they also use cultivated and abandoned fields (Bechard and Swem 2002). Rough-legged hawks are absent from northern regions where the average minimum January temperature is below -23°C (-10°F) (Root 1988), and are most abundant in areas with less than 1,020 millimeters (40 inches) of annual precipitation (Palmer 1988).

Reproduction

Rough-legged hawks are at least serially monogamous, forming pair bonds that are maintained at least through the breeding season, if not longer. There is some evidence that pairs may form in wintering areas, where birds roost and perch in pairs; also, they sometimes migrate in pairs (Palmer 1988; Bechard and Swem 2002). They generally arrive on the breeding grounds in April and May, and nest building begins soon after arrival. The nest is constructed of sticks, bones, weeds, grass, and other debris and is lined with grass, down, feathers, and the fur of prey animals. A clutch of two to seven eggs is laid between early May and late June, depending on weather and prey abundance (Brown and Amadon 1968; Kessel 1989). Larger clutches may be laid in good lemming years (Brown and Amadon 1968). Incubation lasts for approximately 28–31 days and is conducted mostly by the female. Nestlings fledge at approximately 36–40 days. Fledglings remain dependent on their parents for another 4–6 weeks postfledging, with the period of dependence perhaps continuing into fall migration. (Bechard and Swem 2002.)

Dispersal Patterns

There is little information on natal dispersal distances, and what information is available is poorly documented. The distance between natal site and subsequent breeding site of rough-legged hawks in Russia was reported to be 1,955 kilometers (1,215 miles) ± 1.079 (0.67), but the number of birds in this study was not reported (Galushin 1974).

Pairs often use the same site year after year, a predictable pattern given the limited availability of nest sites. However, because no studies have been conducted using marked individuals, breeding site fidelity is essentially unknown (Bechard and Swem 2002). There is some evidence of fidelity to wintering sites in California (Garrison and Bloom 1993).

Longevity

An average life span of 20.7 months based on 48 recoveries of dead birds has been reported (Bechard and Swem 2002). The oldest known wild individual was 17 years 1 month (U.S. Geological Survey 2003).

Sources of Mortality

Forty-six of 48 rough-legged hawks recovered dead and reported to the U.S. Geological Survey Bird Banding Laboratory died from anthropogenic causes, primarily vehicle collisions. Other sources of human-related mortality include shooting, trapping, and collisions with towers and wires (Keran 1981). Eggs and nestlings are occasionally taken by jaegers (*Stercorarius* sp.), snowy owls (*Nyctea scandiaca*), arctic foxes (*Alopex lagopus*), and at least once by a grizzly bear (*Ursus arctos*) (Bechard and Swem 2002).

Behavior

Rough-legged hawks have been reported to defend a small territory around the nest site during the breeding season. In some areas, they have been reported to defend winter territories against conspecifics as well as red-tailed hawk (*Buteo jamaicensis*), northern harrier (*Circus cyaneus*), prairie falcon (*Falco mexicanus*), and common raven (*Corvus brachyrhynchos*) (Bechard and Swem 2002). However, where food is plentiful, rough-legged hawks can occur in concentrations and are apparently not territorial (Schnell 1967).

Home range size during the breeding season is unknown. In winter, rough-legged hawks occasionally roost communally and have overlapping home ranges. The reported sizes of winter home ranges are highly variable, ranging from 6 to more than 530 square kilometers (2.3 to 205 square miles). (Watson 1986.)

Rough-legged hawks feed primarily on lemmings (*Lemmus sibiricus, Dicrostonyx groenlandicus*) and voles (*Microtus* spp.) during the breeding season, and on voles, mice (*Peromyscus* spp., *Mus musculus*), and shrews (*Blarina* spp. and *Sorex* spp.) in winter. Occasionally, these hawks will take small birds, ground squirrels (*Spermophilus* spp.), and rabbits (*Sylvilagus* sp.), particularly in summer. (Bechard and Swem 2002.)

Rough-legged hawks hunt from perches or on the wing, probably depending on availability of perches, wind conditions, and possibly other factors. This species is known for its characteristic form of "hover hunting" in light winds (Bechard and Swem 2002).

Movement and Migratory Patterns

Rough-legged hawk is a medium-distance migrant. The entire population moves from the breeding grounds in the arctic and subarctic to open country in southern Canada and the continental United States. Departure from the breeding grounds begins in late August or September, with individuals arriving at wintering areas from October through December. The timing of migration is highly variable, possibly due to variations in cyclic prey populations. Fall migration can begin as early as February, but more typically occurs in mid-March and early April. (Bechard and Swem 2002.)

Ecological Relationships

Several aspects of the ecology of rough-legged hawks, including breeding phenology, reproductive success, timing of migration, social behavior, and distribution and abundance, have been linked to their reliance on cyclic prey populations, although supporting evidence has at times been wanting (Bechard and Swem 2002). Rough-legged hawks frequently share nesting cliffs with peregrine falcon (*Falco peregrinus*), gyrfalcon (*Falco rusticolus*), and common raven, nesting as close as 30 meters in some cases (White and Cade 1971). In winter, rough-legged hawks will kleptoparasitize (rob food from) red-tailed hawks, northern harriers, and short-eared owls (*Asio flammeus*), and are in turn occasionally kleptoparasitized by white-tailed kites (*Elanus leucurus*), red-tailed hawks, and common ravens. Ravens will often mob a rough-legged hawk in groups of two to seven. (Bechard and Swem 2002.)

Population Threats

Due to the remoteness of most breeding habitats, population threats are more likely to be a problem in wintering areas. Potential threats include shooting and trapping; collisions (e.g., with vehicles, wind turbines, power poles); and habitat loss (Bechard and Swem 2002). Of these, habitat loss in wintering areas is perhaps the greatest threat. Development of agricultural lands and urban sprawl have been identified as potential threats to wintering populations in California (Garrison and Bloom 1993).

Peregrine Falcon (*Falco peregrinus*)

Status

FEDERAL: Delisted in 1999

STATE: Endangered

OTHER: Fully Protected (California Fish and Game Code Section 3511)

RECOVERY PLAN: *Recovery plan for Peregrine Falcon (Pacific Population)*. (U.S. Fish and Wildlife Service 1982).

PLACER NCCP/HCP CATEGORY: Class 1

Distribution

North America

Peregrine falcons are found on all continents but Antarctica. The American peregrine falcon occurs in North America from south of the tundra to northern Mexico and migrates to Central America (Ferguson-Lees and Christie 2001).

California

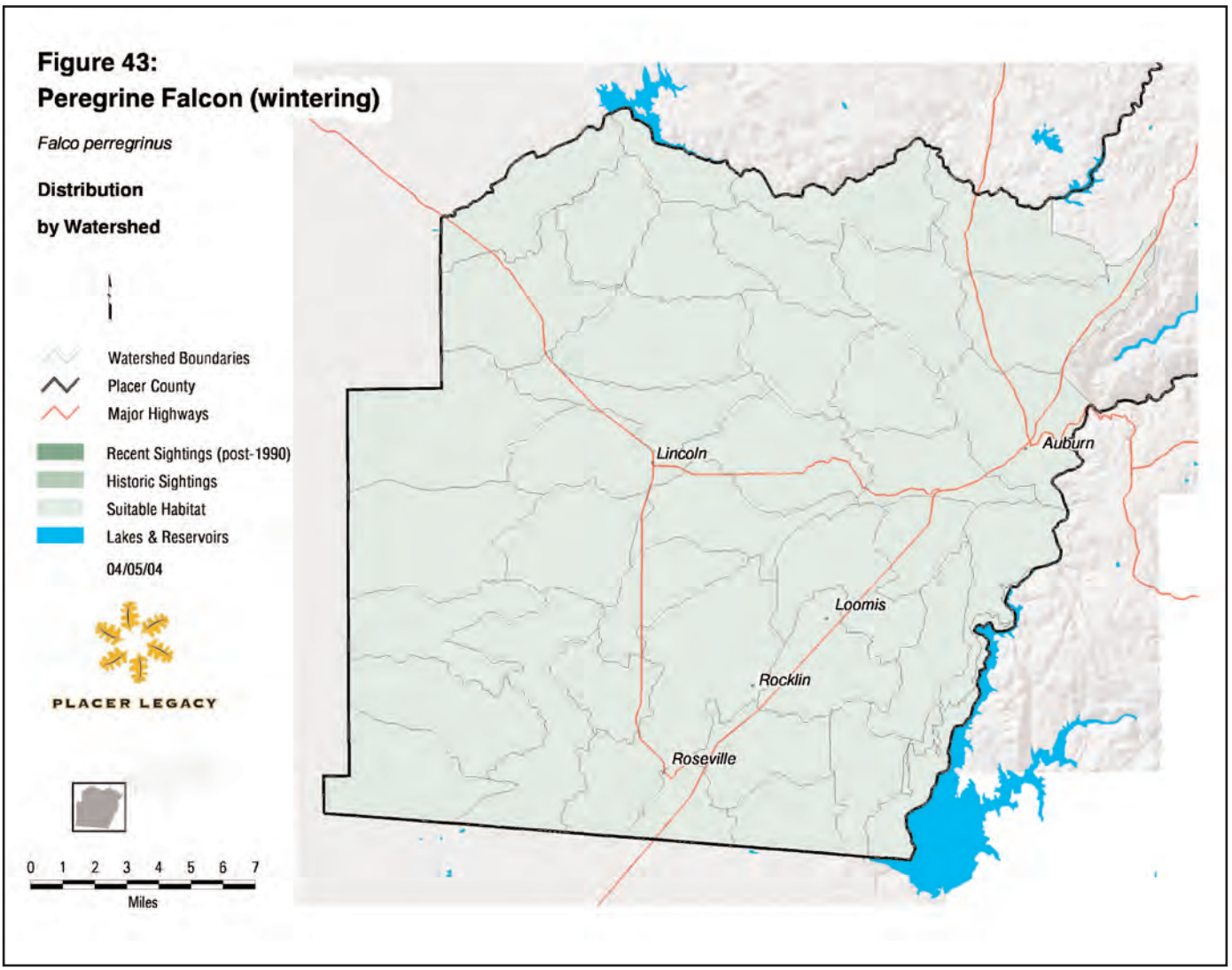
Historically, American peregrine falcons occurred throughout most of California (California Department of Fish and Game 1980; U.S. Fish and Wildlife Service 1982). The California breeding range has been expanding and now includes the central and southern California coast, inland North Coast Ranges, Klamath Mountains, Cascade Ranges, and Sierra Nevada (California Department of Fish and Game 2000c). Although relatively uncommon, wintering birds can be seen throughout California (Zeiner et al. 1990). Populations increase in winter with the arrival of migrating birds from the north (Grinnell and Miller 1944). Historically, American peregrine falcons nested throughout the state, with concentrations along the coast and around the Channel Islands (Grinnell and Miller 1944).

Placer County Phase I Planning Area

Historical

There are no historical records of American peregrine falcons nesting in Placer County (Figure 43), although the species can been seen in low numbers throughout the county in all seasons (Webb 2003).

Peregrine Falcon Photo by Mike Danzenbaker



Current

The California Natural Diversity Database lists no records of peregrine falcons nesting in Placer County, although the species is suspected to breed there (California Natural Diversity Database 2002; Webb 2003). However, there is a known breeding site in Yuba County approximately 30 miles from the boundary of the Phase I Planning Area (Beedy pers. comm.).

Population Status and Trends

North America

The American peregrine falcon declined after the 1940s as a result of the widespread use of chlorinated hydrocarbon pesticides (U.S. Fish and Wildlife Service 1982). Bans on the use of DDT in the 1970s and a major reintroduction program led by the Peregrine Fund have resulted in an impressive increase in the distribution and abundance of this species over the last 20 years. The population increase has been substantial enough to warrant the taxon's delisting, in August 1999, from the federal endangered species list (Mesta et al. 1995; Cade et al. 1997), although this decision is controversial (Pagel et al. 1996; Pagel and Bell 1997).

California

The American peregrine falcon declined after the 1940s as a result of the widespread use of chlorinated hydrocarbon pesticides (U.S. Fish and Wildlife Service 1982). Bans on the use of DDT in the 1970s and a major reintroduction program led by the Peregrine Fund have resulted in an impressive increase in the distribution and abundance of this species over the last 20 years. The population increase has been substantial enough to warrant the taxon's delisting, in August 1999, from the federal endangered species list (Mesta et al. 1995; Cade et al. 1997), although this decision is controversial (Pagel et al. 1996; Pagel and Bell 1997).

The American peregrine falcon population in California began to decline seriously in the 1950s. From a conservative historical estimate of 100 pairs breeding in California before 1947, fewer than 10 nesting sites were believed active in 1969 (Herman et al. 1970). Since 1970, additional nesting pairs have been located. This increase in numbers is probably due to the effects of more exhaustive search efforts in conjunction with an intensive captive breeding and nest augmentation program (White et al. 2002). Although monitoring efforts have been reduced since 1992, current data indicate the population has continued to increase (California Department of Fish and Game 2000c).

Placer County Phase I Planning Area

Peregrine falcons occur regularly but in small numbers throughout the year in Placer County (Webb 2003). There is no information on population trends that is specific to the Phase I Planning Area.

Natural History

Habitat Requirements

American peregrine falcons nest almost exclusively on protected ledges of high cliffs, primarily in woodland, forest, and coastal habitats (California Department of Fish and Game 1980; U.S. Fish and Wildlife Service 1982; White et al. 2002). A very small number of nests have been found on small outcrops and in trees, and a number of reintroduced pairs nest on tall buildings and bridges. Cliffs that provide ledges, potholes, or small caves (usually with an overhang), and that are relatively inaccessible to mammalian predators, are required components of nesting habitat. Nest sites usually provide a panoramic view of open country, are near water, and are associated with a local abundance of passerine, waterfowl, or shorebird prey (Johnsgard 1990; White et al. 2002). Peregrine falcons prefer to nest near marshes, lakes, and rivers that support an abundance of birds (Grinnell and Miller 1944; California Department of Fish and Game 1980), but they travel several miles from their nest sites to forage on pigeons, shorebirds, waterfowl, and songbirds (White et al. 2002). Coastal and inland marsh habitats are especially important to peregrine falcons in fall and winter because they attract large concentrations of waterbirds (California Department of Fish and Game 1980). Peregrine falcons have been known to nest at elevations as high as 3,048 meters (10,000 feet), but most occupied nest sites are below 1,200 meters (4,000 feet) (Shimamoto and Airola 1981; White et al. 2002).

Reproduction

The breeding season of American peregrine falcon generally begins in February and lasts until June. Courtship (in February) typically involves the male provisioning the female with food. A month or two after courtship begins, females normally lay four eggs (range three to five); egg-laying in California typically occurs in March. Both male and female incubate the eggs for 29–33 days. In California, fledging occurs in late May or early June when the young are 35–42 days old. Juvenile peregrine falcons begin hunting on their own and become independent 6–15 weeks after fledging. If a nest fails early due to predation or other factors, the pair may lay a second clutch at an alternate nest site. (California Department of Fish and Game 1989; Ferguson-Lees and Christie 2001; White et al. 2002).

Dispersal Patterns

Juvenile peregrine falcons from California range more widely than adults and can travel as far as Mexico and Oregon. Birds reach maturity at 2–3 years. Once a bird reaches maturity, it may replace a dead member of a breeding pair, acquire a mate and establish a new territory, or occupy and defend a potential territory prior to pairing. Though peregrine falcons occasionally establish territories far (up to hundreds of miles) from their natal site, they more commonly establish territories in the general vicinity of the natal site (White et al. 2002). No peregrine falcons banded as juveniles in California have been found outside the state as adults (California Department of Fish and Game 1989).

Longevity

Peregrine falcons are relatively long-lived. The longevity record for a wild bird is at least 19 years 3 months (U.S. Geological Survey 2002). A few captive birds have lived beyond 20 years (White et al. 2002).

Sources of Mortality

Known predators of peregrine falcon include great horned owl (*Bubo virginianus*), golden eagle (*Aquilla chrysaetos*), red-tailed hawk (*Buteo jamaicensis*), raccoon (*Procyon lotor*), and coyote (*Canis latrans*) (White et al. 2002). However, predation is not known to substantially affect peregrine falcon at the population level (U.S. Fish and Wildlife Service 1999).

Behavior

During the breeding season, adult peregrine falcons attack and chase other raptors away from the nest, especially golden eagles and other peregrine falcons that move through their territory. Adults hunt over a large area around the nest site; foraging may occur up to 19 kilometers (12 miles) from the nest (White et al. 2002).

Peregrine falcons feed almost exclusively on birds, generally of medium to moderately large size. They typically feed on highly mobile, flocking, and colonial nesting birds, such as shorebirds, waterfowl, doves, and pigeons (Johnsgard 1990). Peregrine falcons chase and grab their prey or dive down on them at speeds up to 161–322 kilometers per hour (100–200 miles per hour) (i.e., *stooping*). During the stoop, a peregrine falcon grasps its prey or strikes it with its talons and subsequently retrieves it on the ground. Peregrine falcons hunt during the day or at dusk (White et al. 2002).

Movement and Migratory Patterns

During fall and spring, the tundra subspecies (*F. p. tundrius*) and northern populations of American peregrine falcon migrate through California to and from breeding and wintering areas (White et al. 2002). In California, peregrine falcons usually remain near their breeding territories year-round. However, during late summer and winter, they apparently move relatively short distances to preferred foraging areas (California Department of Fish and Game 1989).

Ecological Relationships

Peregrine falcons compete with other raptors such as golden eagles, red-tailed hawks, and prairie falcons (*Falco mexicanus*) for cliff-nest sites (U.S. Fish and Wildlife Service 1982; White et al. 2002). There is also some concern that American peregrine falcons may predate on other endangered species of birds, such as marbled murrelets (*Brachyramphus marmoratus*) and California least terns (*Sterna antillarum browni*) (California Department of Fish and Game 2000c).

Population Threats

The widespread use of organochlorine pesticides, especially DDT, was a primary cause of the decline in peregrine falcon populations (U.S. Fish and Wildlife Service 1982; White et al. 2002). High levels of these pesticides and their metabolites (i.e., byproducts of organic decompositions) have been found in the tissues of peregrine falcons, leading to thin eggshells and reproductive failure. Other causes of decline include illegal shooting, illegal falconry activities, and habitat destruction and modification (California Department of Fish and Game 1980; White et al. 2002).

California Black Rail (*Laterallus jamaicensis coturniculus*)

Status

FEDERAL: Species of Concern.

STATE: Threatened.

OTHER: Fully Protected (California Fish and Game Code 3511).

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: 1.

Distribution

North America

Black rails breed primarily along the eastern seaboard from Long Island, New York to southern Florida and along parts of the Gulf Coast. There are scattered small populations in California, the Midwest to the southern Great Plains, and interior North Carolina to northern Georgia. (Eddleman et al. 1994.)

California

California black rail populations were previously thought to be restricted to the San Francisco Bay Area, Bolinas Lagoon, Tomales Bay, Morro Bay, Suisun Bay, the Delta region to White Slough in San Joaquin County, the Salton Sea area, and the Lower Colorado River Valley (Grinnell and Miller 1944; Manolis 1978; Garrett and Dunn 1981; Evens et al. 1991; Eddleman et al. 1994). In 1994, however, new populations were discovered in the western Sierra Nevada foothills of Yuba County (Aigner et al. 1995), and subsequent surveys revealed previously unknown populations in the foothills of Butte and Nevada Counties (Tecklin pers. comm.). As of 1999, there were 71 known black rail locations at elevations below 250 meters (820 feet) in the foothills of Butte, Nevada, and Yuba Counties (Tecklin 1999).

Placer County Phase I Planning Area

Historical

There are no historical records of California black rail in the Phase I Planning Area.

Current

There is one record of 3–4 individuals in western Placer County (Figure 44). This small population was discovered on April 15, 2003, near Camp Far West Reservoir in the Phase I Planning Area (Jones & Stokes file data).

Population Status and Trends

North America

Black rail populations have declined throughout the species' range, especially in the Midwest, due to habitat destruction (Eddleman 1994).

California

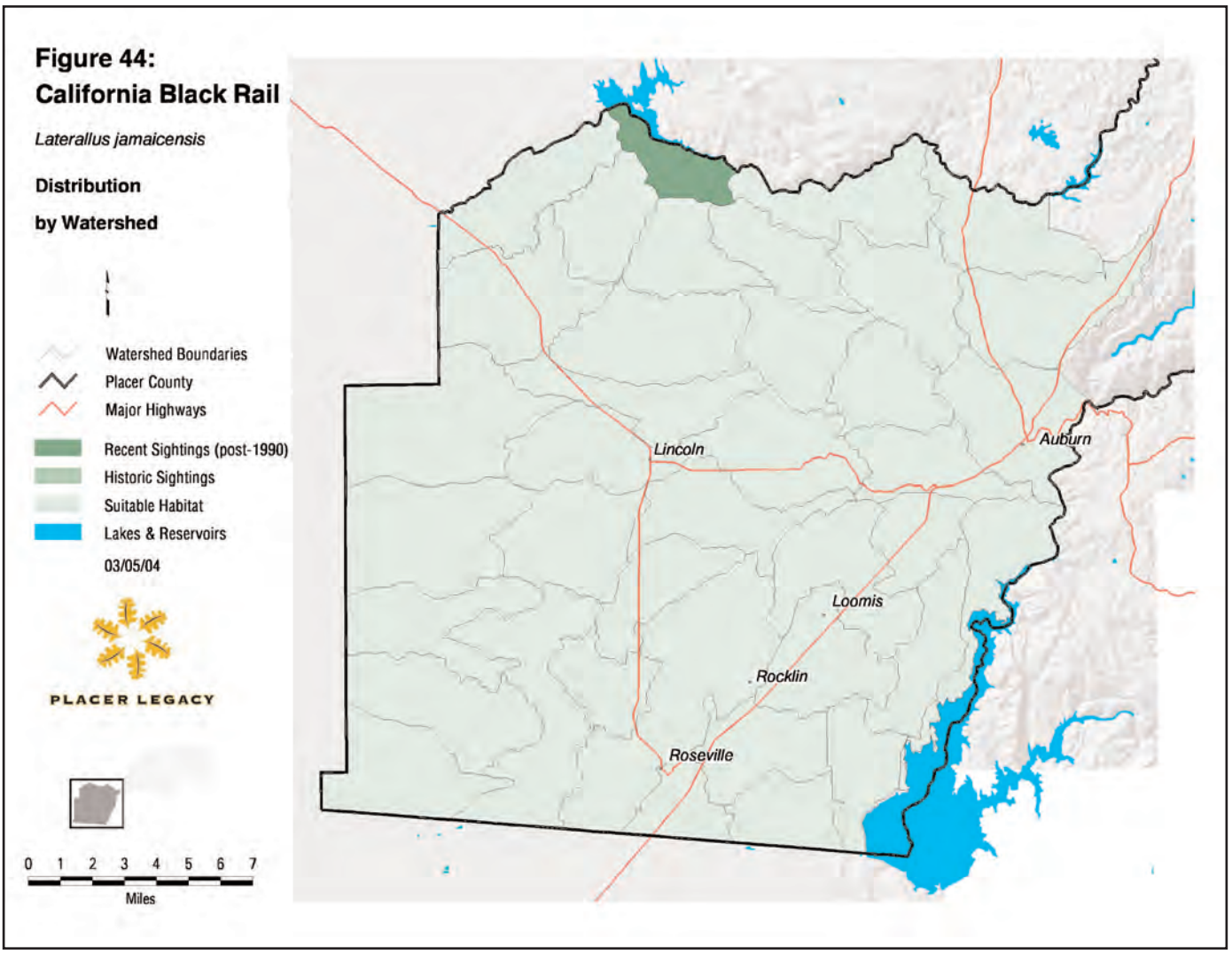
Black rail populations have been extirpated from Ventura to San Diego Counties (Garrett and Dunn 1981). The loss of 95% of marsh habitat in the San Francisco Bay Area likely had a substantial effect on black rail populations. Populations along the Lower Colorado River declined about 30% from 1973 to 1989. (Evens et al. 1991.)

The Sierra Nevada foothill population was estimated at 125–184 during 1997 and 1998 (Tecklin 1999).

Placer County Phase I Planning Area

Although black rail populations were discovered in Yuba and Nevada Counties in 1994, a small population was only recently discovered in Placer County. Accordingly, there is no information on population trends in the Phase I Planning Area (Jones & Stokes file data).

California Black Rail Photo by Pete LaTourette



Natural History

Habitat Requirements

Black rails in the Sierra Nevada foothills are found in perennial wetlands that are dominated by rushes (*Juncus* spp.) and cattails (*Typha* spp.), often with associated species such as *Scirpus, Eleocharis*, and *Paspalum* (Aigner et al. 1995; Tecklin 1999). These wetlands are in open grasslands, grazed pastures, or oak savannas (Tecklin 1999). Nesting habitat is characterized by water depths of less than 3 centimeters (1.2 inches) that do not fluctuate during the year and by dense vegetation that provides adequate cover (Eddleman 1994; Tecklin 1999). Wetlands larger than 0.4 hectare (1 acre) are more likely to support populations that persist over time. Black rails were not found during surveys of roadside ditches that had dense patches of *Typha* and *Scirpus.* (Tecklin 1999.)

Reproduction

California black rails lay three to eight eggs, incubate them for 17–20 days, and probably brood the precocial chicks for several days after hatching (Eddleman 1994). There is little information on parental care after hatching, and no information is available on reproductive success and survivorship.

Dispersal Patterns

There is no information on dispersal of birds from the Sierra Nevada foothill population, although it is likely that young birds disperse to seek new sites for colonization if densities in an occupied marsh exceed the habitat's carrying capacity or if an occupied marsh is degraded. This hypothesis is supported by records of juveniles from other populations appearing in atypical habitats, migrant rails striking television antennae and buildings, and low recapture rates of banded juveniles compared to those of adults (Eddleman et al. 1994).

Longevity

There are no published estimates of black rail longevity; however, one male along the Lower Colorado River lived for at least 2.5 years (Eddleman 1994).

Sources of Mortality

Documented predators of black rails include great blue heron (*Ardea herodias*), great egret (*Ardea alba*), northern harrier (*Circus cyaneus*), ring-billed gull (*Larus delawarensis*), great horned owl (*Bubo virginianus*), short-eared owl (*Asio flammeus*), and loggerhead shrike (*Lanius ludovicianus*) (Evens and Page 1986; Eddleman et al. 1994). In marshes around San Francisco Bay, rats (*Rattus* spp.) and red fox (*Vulpes vulpes*) are thought to prey on nests (Evens pers. comm. in Eddleman et al. 1994).

Behavior

Black rails forage on invertebrates, including snails, beetles, earwigs, grasshoppers, ants, and on seeds from bulrushes (*Scirpus* spp.) and cattails (*Typha* spp.) (Eddleman 1994). There is no specific information on the diet of the Sierra Nevada foothill population.

Movement and Migratory Patterns

California black rails are mostly resident, although there is some local movement from San Pablo Bay south to the southern San Francisco Bay (Evens et al. 1991). Based on continual presence throughout the year, the Sierra Nevada foothill population is thought to be nonmigratory (Tecklin 1999, pers. comm.).

Ecological Relationships

Black rails occupy marshes with Virginia rail (*Rallus limicola*) and sora (*Porzana carolina*) (Tecklin 1999) but there is no information on interspecific or intergeneric interactions (Eddleman et al. 1994).

Population Threats

The primary population threats are destruction, desiccation, flooding, grazing and other forms of degradation of marsh habitats; development-related increases in predation pressures from domestic cats, herons, egrets, and other predators; and pollution carried by runoff into occupied marshes (Eddleman et al. 1994). Grazing occurs at 60% of the known wetlands occupied by black rails in the Sierra Nevada foothills and is the most common threat to those wetlands (Tecklin 1999).

Western Yellow-Billed Cuckoo (*Coccyzus americanus occidentalis*)

Status

FEDERAL: Petition for listing under review.

STATE: Endangered.

OTHER: None.

RECOVERY PLAN: No recovery plan exists for this species.

PLACER NCCP/HCP CATEGORY: Class 2.

Distribution

North America

The breeding range of yellow-billed cuckoo (*C. americanus*) extends from southern Ontario and Quebec through the eastern United States and the Great Plains to northern Mexico, northern Yucatan, Cuba, and southern Hispanola. In the western United States, breeding populations occur very locally in riparian woodland in the Southwest, the Great Basin, and the California's Central Valley. The wintering range is limited to South America. (Hughes 1999.)

California

The California breeding range of western yellow-billed cuckoo is restricted to the Sacramento Valley, the South Fork of the Kern River, the Lower Colorado River Valley, and sometimes the Prado Basin in Riverside and San Bernardino Counties (Gaines and Laymon 1984). Most recent Sacramento Valley records are from the Sacramento River from Todd Island in Tehama County south to Colusa State Park in Colusa County; the Feather River in Yuba and Sutter Counties (Gaines and Laymon 1984); and the Sutter Bypass (Jones & Stokes file data).

Placer County Phase I Planning Area

Historical

Western yellow-billed cuckoo occurred as a breeding species along the Auburn Ravine during the early 1900s (Adams 1909; Figure 45).

Current

There are no recent records of western yellow-billed cuckoo in Placer County (Webb 2003).

Population Status and Trends

North America

Breeding Bird Survey data are inadequate to assess yellow-billed cuckoo population trends throughout their range (Sauer et al. 2001). Population trends are highly variable locally; in the eastern United States, they fluctuate with the population cycles of tent caterpillars and other insects (Hughes 1999). Yellow-billed cuckoos are still common throughout the midwestern and southeastern United States but are less so in the northeast states (Hughes 1999).

California

In California, much of the riparian habitat that historically supported nesting western yellow-billed cuckoo has been removed or degraded. Consequently, the subspecies has been extirpated from the San Francisco Bay Area; the San Joaquin Valley; the Los Angeles Basin; and Siskiyou, Modoc, Monterey, San Luis Obispo, Ventura, Santa Barbara, Orange, and San Diego Counties (Garrett and Dunn 1981; Gaines and Laymon 1984; Unitt 1984; Roberson and Tenney 1993; Lehman 1994; Gallagher 1997). Populations have also been greatly reduced or eliminated over much of the Sacramento Valley and the Lower Colorado River Valley, with a 92–96% decline at the latter location from 1977 to 1986 (Gaines and Laymon 1984; Laymon and Halterman 1987).

Western Yellow-Billed Cuckoo Photo by Mike Danzenbaker

Placer County Phase I Planning Area

Yellow-billed cuckoo has been extirpated from the Phase I Planning Area (Webb 2003).

Natural History

Habitat Requirements

In the western United States, yellow-billed cuckoos breed in broad, well-developed, low-elevation riparian woodlands comprised primarily of mature cottonwoods (*Populus* spp.) and willows (*Salix* spp.). Studies along the Lower Colorado River Valley and throughout their range have shown that smaller willow-cottonwood stands (<40 hectares [99 acres]) have low rates of occupancy, whereas large sites (>80 hectares [198 acres]) have the highest occupancy rates. Optimal stands are larger than 80 hectares (198 acres) in extent and more than 600 meters (1,969 feet) wide; marginal stands are 20–40 hectares (49–99 acres) in extent and 100–200 meters (328–656 feet) wide; and unsuitable stands are less than 15 hectares (37 acres) in extent and less than 100 meters (328 feet) wide (Laymon and Halterman 1989). Typical nest sites in California (N = 18 nests) had moderately high canopy closure (79.6%) and low total ground cover (18.7%). The average distance from nests to water was 34.7 meters (114 feet) (Laymon and Halterman 1987). Along the Sacramento River in Glenn County, yellow-billed cuckoos have been documented nesting in walnut orchards adjacent to riparian habitats (Gaines and Laymon 1984).

Reproduction

Breeding season for yellow-billed cuckoos generally begins with pair formation in mid-June and lasts until mid-August. From June to August females lay one to five eggs, and both parents incubate them for 9–11 days. Both parents feed the nestlings until they fledge at approximately 7–9 days. This rapid rate of development allows for the species' short stay in California. Adult helpers at nests have been documented in southern California. Two females and two males were noted feeding nestlings at single nests. Because some females lay eggs in other nests than theirs, the two females were probably both parents of the young in the observed nest. At the South Fork of the Kern River, Kern County, four nests fledged young but only 57% of eggs produced young, and only 43% of those young survived for more than 1 week. The mean number of surviving young per nest was 1.5. There is no information on lifetime reproductive success. (Hughes 1999.) For information relating to brood parasitism see *Ecological Relationships* below.

Dispersal Patterns

There is no information on initial dispersal from natal or breeding sites. There is limited information on breeding site fidelity in Canada, but none from the western population (Hughes 1999). However, in California, western yellow-billed cuckoos return annually to nearly all of the few currently occupied breeding locations remaining in suitable condition. This suggests that they exhibit strong nest-site fidelity. A few western yellow-billed cuckoos have been found in eastern Californian desert oases and riparian areas during June (spring migration) (Sterling pers. comm.).

Longevity

There is very little information on survivorship or life span due to the small number of band recoveries. In Canada, three banded yellow-billed cuckoos were at least 4 years old. (Hughes 1999.)

Sources of Mortality

Predation is thought to cause 80% of nest failures. Blue jays (*Cyanocitta cristata*) and common grackles (*Quiscalus quiscula*) are known nest predators in the eastern United States. In California, the likely nest predators are snakes, mammals, American crow (*Corvus brachyrhynchos*), and jays (*Cyanocitta* sp., *Aphelocoma* sp.). Raptors may prey significantly on adults and have been observed attacking adults on nests. Many migrating yellow-billed cuckoos are in weakened condition and are easily killed by raptors over open terrain. (Hughes 1999.)

Behavior

Using radiotelemetry, Laymon and Halterman (1987) determined that yellow-billed cuckoos have large home ranges, averaging 17 hectares (42 acres) in southern California riparian habitats. However, there was no evidence of territoriality, prompting them to speculate that adjacent pairs nested asynchronously in order to reduce competition for food, thereby negating the necessity for territories (Laymon and Halterman 1989).

During the breeding season, yellow-billed cuckoos prey on caterpillars (especially Sphinx moth [Sphingidae] larvae) and grasshoppers. In southern California, katydids (Tettigoniidae) become an increasingly important

part of the diet during the latter part of summer. Also, small frogs and fruit such as blackberries (*Rubus* spp.), wild grapes (*Vitis* spp.) and elderberries (*Sambucus* spp.) may be important food sources. (Hughes 1999.)

Movement and Migratory Patterns

Yellow-billed cuckoos migrate annually to wintering grounds in South America, flying at night. The western subspecies migrates 4–8 weeks later in the spring and 2–3 weeks earlier in the fall than eastern populations. Spring migration begins in late May and lasts until late June, and fall migration begins in late August and lasts until mid-September. Locations of wintering grounds of western yellow-billed cuckoos in Latin America are not well known; specimens collected in the species' wintering range have not been identified to the subspecific or regional population level. (Hughes 1999.)

Ecological Relationships

Yellow-billed cuckoos are facultative brood parasites over much of their range. They will lay eggs in nests of conspecifics (recognized by clutches of five or more eggs) and occasionally in nests of other species such as black-billed cuckoo (*Coccyzus erythropthalmus*), American robin (*Turdus migratorius*), gray catbird (*Dumetella carolinensis*), and wood thrush (*Hylocichla mustelina*). Most hosts lay similar blue-green eggs. There are no data on nest parasitism from California. Brood parasitism rates may be higher during population spikes of insect prey such as cicadas and tent caterpillars. In the eastern United States, mobbing of females by individuals of other species may indicate their recognition of the threat of parasitism. Yellow-billed cuckoos are infrequently subject to brown-headed cowbird (*Molothrus ater*) brood parasitism. (Hughes 1999).

Population Threats

Yellow-billed cuckoos in California are threatened by the loss or degradation of suitable large tracts of riparian habitat, pesticide poisoning, and loss of prey base due to pesticides (Gaines and Laymon 1984).

Western Burrowing Owl (*Athene cunicularia hypugea*)

Status

FEDERAL: Species of Concern.

STATE: Species of Special Concern.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Burrowing owls (*A. cunicularia*) live and breed in the desert and grassland habitats from south central Canada through most of the Central United States and Central America to the southern end of South America (Rosenberg et al. 1998).

California

In California the range of western burrowing owl extends through the lowlands south and west from north central California to Mexico, with small, scattered populations occurring in the Great Basin and the desert regions of the southwestern part of the state (DeSante et al. 1996). Burrowing owls are absent from the coast north of Sonoma County and from high mountain areas such as the Sierra Nevada and the ranges extending east from Santa Barbara to San Bernardino. Burrowing owl populations have been greatly reduced or extirpated from the San Francisco Bay Area (Trulio 1997) along the coast to Los Angeles. They have also apparently disappeared from the Coachella Valley. The remaining major population densities of burrowing owls in California are in the Central and Imperial Valleys (DeSante et al. 1996).

Placer County Phase I Planning Area

Historical

There is no information on historic population size and distribution of burrowing owls that is specific to the Phase I Planning Area. Grinnell and Miller (1944) indicated that the species was "originally common" and even "abundant" in suitable habitat that includes the Phase I Planning Area.

Current

Burrowing owls are considered rare in Placer County (Webb 2003). They were only observed at three locations in the Phase I Planning Area in during the 2002 Lincoln Christmas Bird Count and during the 2003 watershed surveys of western Placer County Pandolfino pers. comm., Easterla pers. comm.; Figure 46).

Population Status and Trends

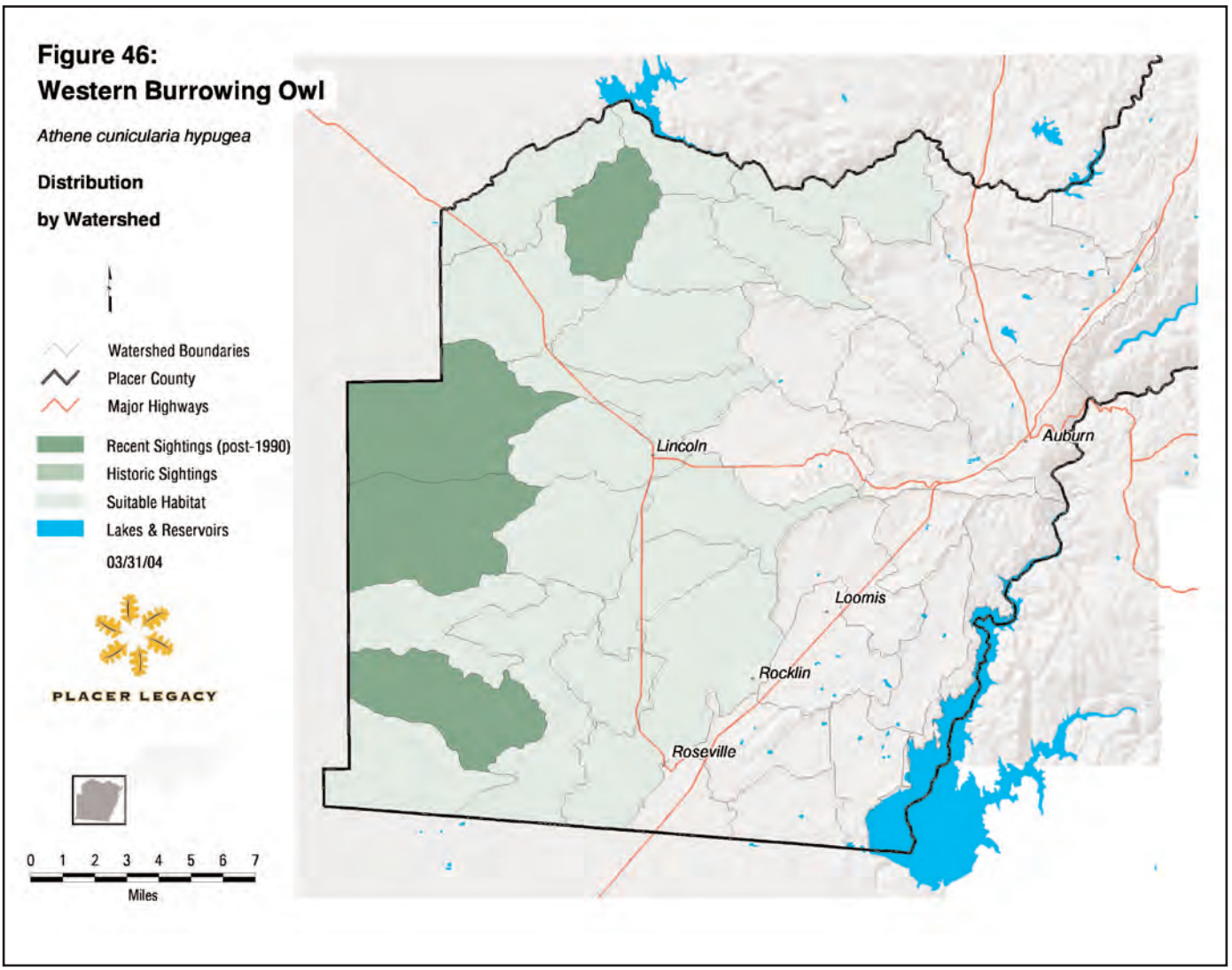
North America

Burrowing owls were once widespread and generally common over western North America. In recent decades a number of populations have declined or, in some cases, disappeared altogether. Burrowing owls are now endangered in Canada and have declined in many parts of the United States (DeSante et al. 1996, 1997; James and Espie 1997). In California, western burrowing owl is a species of special concern; it is listed as endangered or threatened in a number of other states.

California

In California, western burrowing owl is a species of special concern; it is listed as endangered or threatened in a number of other states. The California Department of

Western Burrowing Owl Jones & Stokes file photo



Fish and Game indicates that the California population of burrowing owls is between 1,000 and 10,000 pairs (James and Espie 1997; Rosenberg et al. 1998) with a declining trend. The primary factors cited in the decline are habitat loss, pesticides, predators, harassment, reduced burrow availability, and vehicle collisions.

Placer County Phase I Planning Area

There is no detailed information on population trends of burrowing owls in Placer County because of the lack of baseline data; however, Webb (2003) described the population as declining.

Natural History

Habitat Requirements

Burrowing owls are found in open, dry grasslands, agricultural and range lands, and desert habitats, often associated with burrowing animals. They can also inhabit grass, forb, and shrub stages of piñon and ponderosa pine habitats. They can be found at elevations ranging from 200 feet below sea level to 9,000 feet above. Burrowing owls commonly perch on fence posts or on mounds outside the burrow. They can be found at the margins of airports and golf courses and in vacant urban lots. (Rosenberg et al. 1998.)

Reproduction

The breeding season is March to late August; the season tends to last later in the northern part of the range. Clutch size ranges from one to twelve and averages about seven. The incubation period is 28–30 days. The female performs all the incubation and brooding and is believed to remain continually in the burrow while the male does all the hunting. The young fledge at 44 days but remain near the burrow and join the adults in foraging flights at dusk. (Rosenberg et al. 1998.)

Dispersal Patterns

Burrowing owls tend to be resident where food sources are stable and available year-round. They disperse or migrate south in areas where food becomes seasonally scarce. In resident populations, nest-site fidelity is common, with many adults renesting each year in their previous year's burrow; young from the previous year often establish nest sites near (<300 meters [<985 feet]) their natal sites (Rosenberg et al. 1998). Burrowing owls in migratory populations also often renest in the same burrow, particularly if the previous year's breeding was successful (Belthoff and King 1997). Other birds in the same population may move to burrows near their previous year's burrow.

Longevity

The maximum life span recorded for a banded bird in the wild is about 8.5 years (Rosenberg et al. 1998).

Sources of Mortality

Predators of burrowing owls include prairie falcon (*Falco mexicanus*), red-tailed hawk (*Buteo jamaicensis*), Swainson's hawk (*Buteo swainsoni*), ferruginous hawk (*Buteo regalis*), northern harrier (*Circus cyaneus*), golden eagle (*Aquila chrysaetos*), foxes (*Vulpes* spp.), coyote (*Canis latrans*), and domestic dogs and cats. Many owls are killed at night by traffic when flying low over roads. Attempts to exterminate rodents by the use of poisons may also kill burrowing owls. (Rosenberg et al. 1998.)

Behavior

Burrowing owls tend to be opportunistic feeders. Large arthropods, mainly beetles and grasshoppers, comprise a substantial portion of their diet. Small mammals, especially mice, rats, gophers, and ground squirrels, are also important food items. Other prey animals include reptiles and amphibians, scorpions, young cottontail rabbits, bats, and birds such as sparrows and horned larks (*Eremophila alpestris*). Consumption of insects increases during the breeding season. Burrowing owls may hover while hunting. After catching their prey, they return to perches on fence posts or the ground. Burrowing owls are primarily active at dusk and dawn, but if necessary will hunt at any time of day. (Rosenberg et al. 1998.)

Movement and Migratory Patterns

Northern populations of burrowing owls are usually migratory, while more southern populations may move short distances (Coulombe 1971; Martin 1973; Botelho 1996) or not at all (Brenckle 1936; Ligon 1961; Thomsen 1971; Haug et al. 1993). Little is known about the winter ranges of migratory populations (Haug et al 1993), although migratory burrowing owls are believed to mix with resident populations in California during the winter months (Coulombe 1971). Moreover, those burrowing owls breeding farthest north appear to migrate the farthest south (James and Ethier 1989).

Ecological Relationships

Burrowing owls in California are commensal with California ground squirrels (*Spermophilus beecheyi*) in rangeland and agricultural areas. They may compete incidentally with other predators such as coyote, other owls and hawks, skunks, weasels, and badgers for rodents and a variety of insects. (Rosenberg et al. 1998.)

Population Threats

Western burrowing owl is experiencing precipitous population declines throughout most of the western United States, and has disappeared from much of its historical range in California. Nearly 60% of California burrowing owl "colonies" that existed in the 1980s had disappeared by the early 1990s (DeSante et al. 1997). Conversion of grasslands to agriculture, other habitat destruction, and poisoning of ground squirrels have contributed to population reductions first noted in the 1940s. Within the past 20 years, however, and particularly within the past 5 years, the decline of burrowing owls in California appears to have greatly accelerated. This appears to be a result of habitat loss caused by increased residential and commercial development.

Bank Swallow (*Riparia riparia*)

Status

FEDERAL: None.

STATE: Threatened.

OTHER: Protected under the federal Migratory Bird Treaty Act (16 U.S.C. 703B711); California Partners in Flight: high-priority riparian obligate species (Evans 1997).

RECOVERY PLAN: State Recovery Plan was adopted in 1992.

PLACER NCCP/HCP CATEGORY: Class 2.

Distribution

North America

Bank swallows occur throughout the northern two-thirds of the United States, most of Canada, and into northern Alaska (Garrison 1999).

California

Bank swallow historically occurred along the larger lowland rivers throughout California, with the exception of southern California, where the species occurred principally along the coast and at the mouths of large rivers such as the Los Angeles River (Grinnell and Miller 1944; Humphrey and Garrison 1987). The California Department of Fish and Game (1995, 2000) describes the current breeding range as primarily confined to parts of the Sacramento Valley and northeastern California, including the banks of the Sacramento and Feather Rivers; a few scattered colonies persist along the central and northern coast.

Placer County Phase I Planning Area

Historical

There are no known historical breeding sites in Placer County (Grinnell and Miller 1944; Webb 2003; Figure 47).

Current

There are only a few spring and fall records of migrants in the Phase I Planning Area and no known breeding sites (Webb 2003). Bank swallows may breed if suitable nesting sites become available there in the future. There are records of breeding colonies from nearby Sutter and Yuba Counties, especially along the Feather River (California Department of Fish and Game 1995; California Natural Diversity Database 2003). Occasional migrants pass through the valley portion of the Phase I Planning Area during April, May, August, and September.

Population Status and Trends

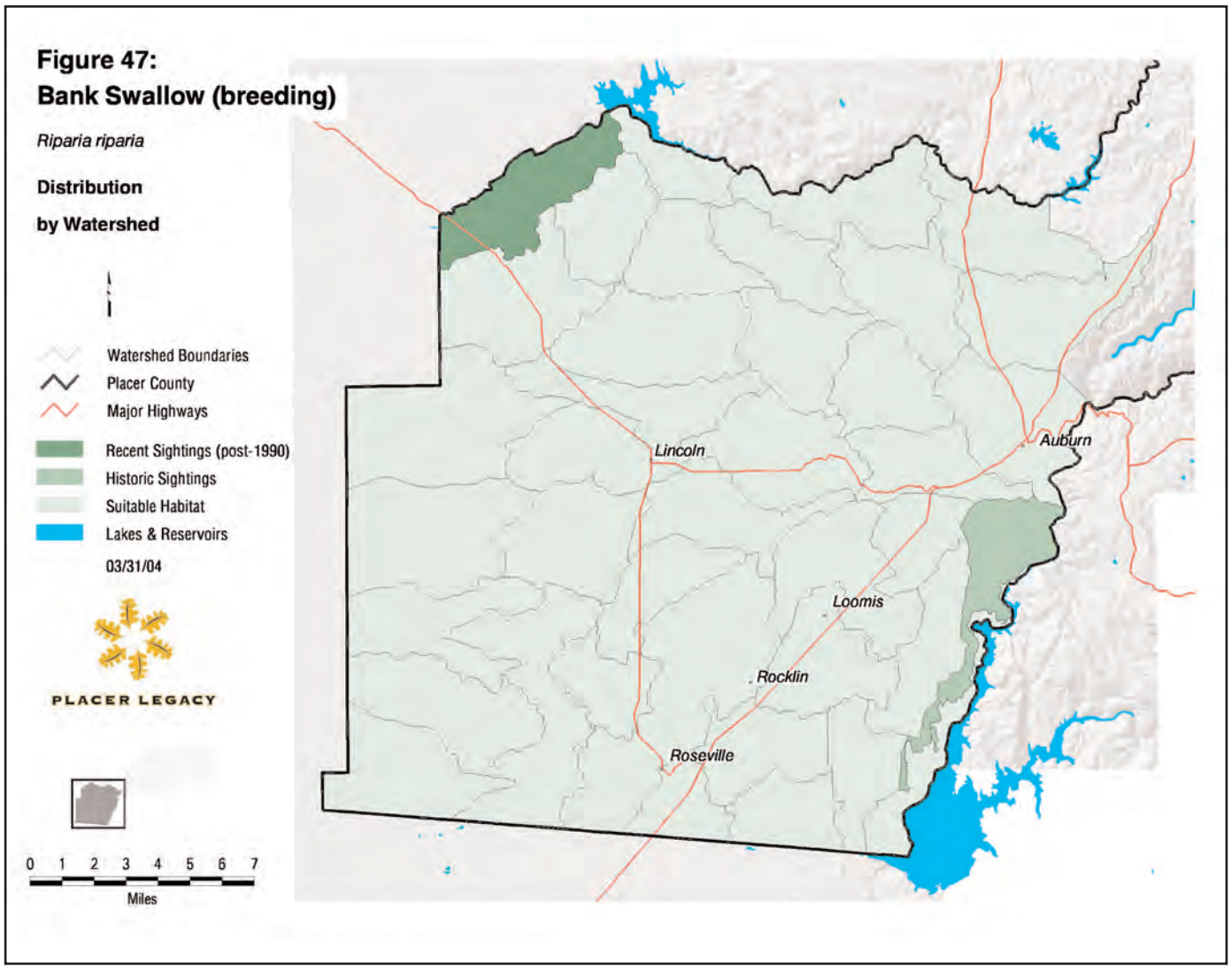
North America

Generally, populations in North America appear to be stable (Sauer et al. 2000). However, because the species breeds locally in colonies that often change locations between years, most population-monitoring programs may not measure population trends reliably. Breeding bird atlas data and focused surveys conducted specifically for bank swallow colonies provide the best available population trend data. (Garrison 1999.)

California

Bank swallow has been extirpated as a nesting species from southern California, and its range in northern California has been reduced by 50% since 1900 (Garrison 1999; California Department of Fish and Game 1995). Most colonies are along the Sacramento River north of the town of Colusa; some remain along the coast, along the Feather River in the Sacramento Valley, and in northeastern sections of northern California. Along the Sacramento River, an estimated population of 13,170 pairs in 1986 declined to 4,990 pairs in 1998 then rebounded to 8,210 pairs the following year (California Department of Fish and Game 2000d). These populations were estimated by applying

Bank Swallow Photo by Mike Danzenbaker



an average 45% occupancy rate of potential nest burrows in an active colony. Data from two colonies near Princeton show that this rate ranged from 31.6 to 56.1% (N = 6) (California Department of Fish and Game 1995).

Placer County Phase I Planning Area

Because there are no breeding records and only a few records of spring and fall migrants within the Phase I Planning Area, the issue of population status and trends is moot (Williams 1996; Webb 2003).

Natural History

Habitat Requirements

Foraging bank swallows take insects on the wing; accordingly, they are found over a wide variety of land cover types. However, bank swallows dig nest burrows in nearly vertical banks/cliff faces and require substrates comprised of soft soils such as fine sandy loam, loam, silt loam, and sand (Garrison et al. 1987). Suitable banks for nesting must be at least 1 meter (3.3 feet) above ground or water for predator avoidance (Garrison 2002). Suitable nest sites are few and are scattered throughout the species' remaining California range; they are most often found at coastal river mouths, along large rivers in the Sacramento Valley, on rivers and wildlife refuges in northeastern California, and occasionally in gravel and sand mines that provide and maintain nesting habitat (Grinnell and Miller 1944; California Department of Fish and Game 1995).

Reproduction

Bank swallows usually initiate a single breeding attempt in April. They lay clutches of one to nine eggs (usually four to five) and incubate them for 13–16 days. The young hatch in May and are fledged by July each year; there is no information on lifetime reproductive success (Garrison 1999).

Dispersal Patterns

Colony sites are often used in subsequent years as long as the substrate and burrows remain intact. There is no information on fidelity to colony or burrow sites from year to year, but it is likely that adults that breed successfully at a colony one year will return in subsequent years, especially considering the limited number of suitable colony sites. Return rate data are suspect because they do not include mortality rates of nonreturning birds. (Garrison 1999.)

Longevity

Bank swallows have been documented in excess of 9 years old. Average survival rates have been estimated at 34.9% and 53% in Wisconsin and New York, respectively. There are no data on survivorship from California populations (Garrison 1999).

Sources of Mortality

Bank swallow predators at nesting colonies include mammals, birds, and snakes. No information exists on predators during migration or on the wintering range. Documented predators in California include American kestrels (*Falco sparverius*) taking adults in flight, and gopher snakes (*Pituophis melanoleucus*) feeding on nestlings in burrows. Burrows in mining pits and along riverbanks sometimes collapse during erosive events, killing nestlings (Garrison 1999).

Behavior

A 1986 survey of bank swallows along the Sacramento River found colony sizes ranging from 12 to 1,784 pairs. Distances between burrows at a colony in California ranged from 1 to 59 centimeters (0.4 to 22.2 inches) (N = 32 colonies). (Humphrey and Garrison 1987.)

Bank swallows forage for aerial insects while flying, as do other swallow species. They 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% ants, bees, and wasps; 26.6% flies; 17.9% beetles; 10.5% mayflies; 8% bugs; and a few dragonflies, butterflies, and moths (Garrison 1999, 2002).

Movement and Migratory Patterns

Bank swallows migrate annually to South America. The first spring migrants arrive in California in mid-March with numbers peaking in May; the first fall migrants leave in late July, with a few birds remaining until mid-September (Humphrey and Garrison 1987; Garrison 1999; Garrison 2002). Bank swallows often join flocks of other species of swallows during migration (Garrison 1999).

Ecological Relationships

Limited data exist on interspecific competition, but there are a few cases where European starlings (*Sturnus vulgaris*) and house sparrows (*Passer domesticus*) have taken over nest burrows. Northern rough-winged swallows (*Stelgidopteryx serripennis*) also nest in burrows in similar substrates and sometimes in the vicinity of bank swallow colonies, but the extent of competition, if any, for those burrow sites is unknown. (Garrison 1999.)

Population Threats

Bank swallows have been eliminated from southern California because almost all the rivers and natural waterways have been converted into concrete-lined flood control channels. They have been eliminated from much of the Central Valley because of the application of riprap to natural riverbanks (California Department of Fish and Game 2000d). Other threats come from predators that have access to colonies, changes in gravel and sand mining operations that destroy or no longer create nesting habitat, and high spring floods that can scour out colonies along riverbanks (Garrison 1999).

Yellow Warbler (*Dendroica petechia*)

Status

FEDERAL: None.

STATE: Bird Species of Special Concern, Priority 3.

OTHER: California Partners in Flight: high-priority riparian obligate species (Evans 1997).

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

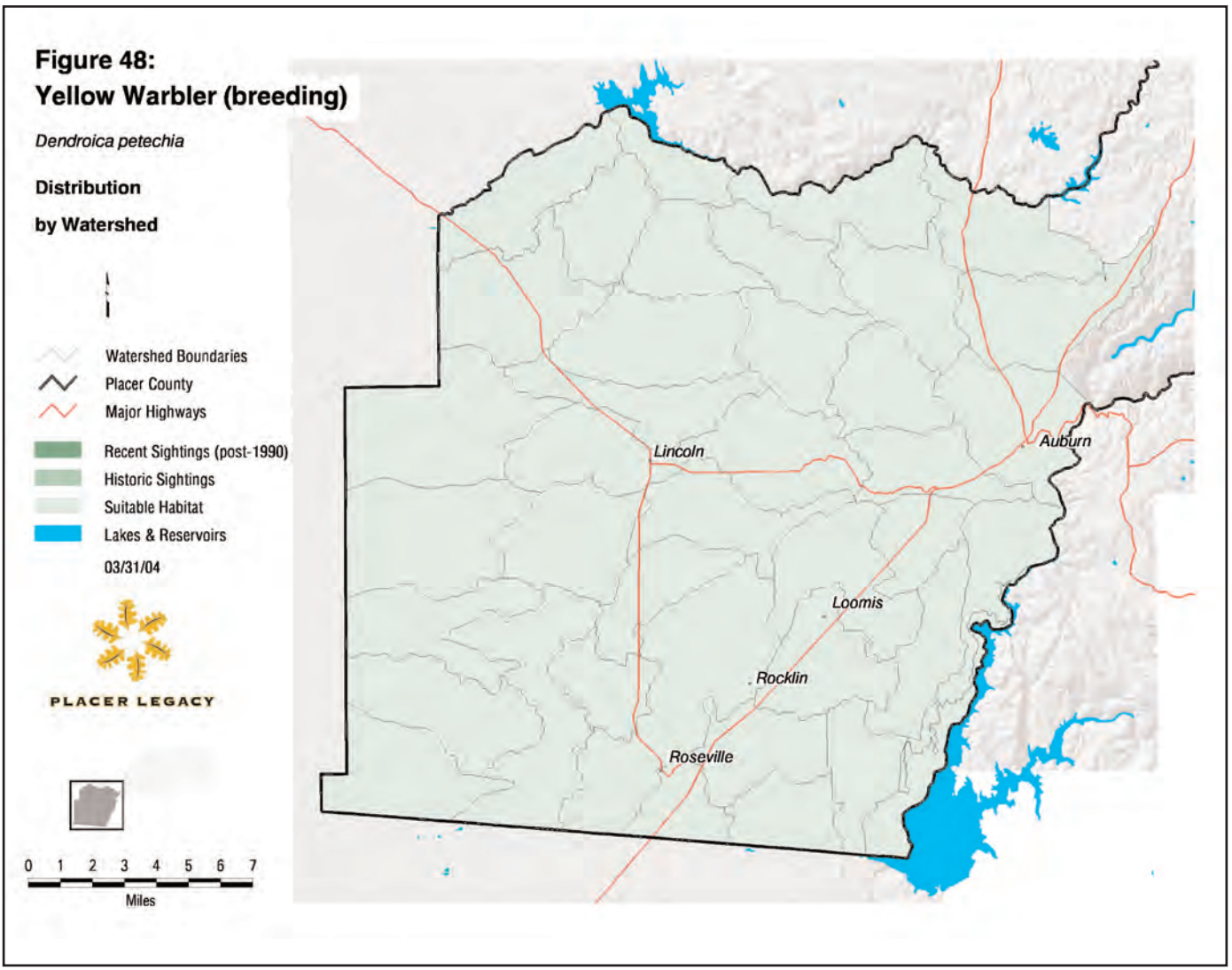
Yellow warblers of the "yellow" group of subspecies breed from the northern limits of the boreal forest in Alaska and Canada throughout most of the upper two-thirds of the continental United States. The breeding range also extends along the Pacific coast into Baja California, and there is a disjunct breeding population in the Sierra Madre Occidental of Mexico. (Lowther et. al. 1999.)

Yellow warblers winter from the coasts along northern Mexico south to northern South America (Lowther et. al. 1999).

California

Nests over all of California except the Central Valley, the Mojave Desert region, and high altitudes in the Sierra Nevada (Zeiner et al. 1990); winters along the Colorado River and in parts of Imperial and Riverside counties; two small permanent populations exist in San Diego and Santa Barbara counties (Garrett and Dunn 1981).

Yellow Warbler Photo by Mark Chainey



Placer County Phase I Planning Area

Historical

There is no specific historical information on the breeding status of yellow warblers within the Phase I Planning Area; however, yellow warblers were reported to nest throughout the lowlands of California, including the Central Valley, in the early to mid-twentieth century (Dawson 1923; Grinnell and Miller 1944).

Current

There is no information on the current breeding status of yellow warblers in the Phase I Planning Area. They are, however, common spring and fall migrants in the riparian corridors and other migration stopover sites (Webb 2003; Easterla pers. comm.; Figure 48).

Population Status and Trends

North America

Yellow warblers are common throughout much of their breeding range (Dunn and Garrett 1997). Breeding Bird Survey results indicate an overall North American population increase of 0.5% from 1966 to 2000 (Sauer et al. 2001).

California

Historically, yellow warblers were common breeders in lowland riparian habitats throughout California (Dawson 1923; Grinnell and Miller 1944). However, they have been largely extirpated from riparian areas in the Central Valley, as well as from lowland portions of southwestern California along the coastal plain (Heath 2002; Garrett and Dunn 1981). By 1973, yellow warblers had disappeared as a breeding species along the Feather and lower Sacramento Rivers, and they were

detected at only four of 20 sites along the upper Sacramento River (Gaines 1974). Breeding Bird Survey results suggest that the current decline of yellow warblers in California is centered mostly in the foothill areas where they still breed (Sauer et al. 1997).

Placer County Phase I Planning Area

Yellow warblers are not known to have nested on the Central Valley floor in western Placer County in recent years (Williams 1996), and no nesting pairs were detected during intensive riparian bird surveys during 1998 and 1999 in nearby Yolo, Sacramento, Sutter, and Colusa Counties (Ballard et al. 1999 cited in Heath 2002). Early descriptions of the species' breeding range in California encompassed most lowland areas including the Central Valley (Dawson 1923; Grinnell and Miller 1944). It is difficult to assess the extent of the historic abundance and consequent population decline because there are no quantitative data of confirmed breeding populations in the Central Valley (Heath 2002). However, with extensive riparian loss and the addition of brown-headed cowbird (*Molothrus ater*) brood parasitism during the twentieth century, it is reasonable to assume that local yellow warbler populations have experienced steep declines or have been extirpated in the Central Valley section of the Phase I Planning Area.

Yellow warblers may nest in the riparian habitats in the foothill region of the Phase I Planning Area (Williams 1996), but this has apparently not been confirmed.

Natural History

Habitat Requirements

In Colorado, Knopf and Sedgwick (1992) found that distance between willows (high density) was the best indicator of yellow warbler density and nest placement. They also found that the warblers selected nest sites based on nest concealment rather than on characteristics (e.g., vigor and structure) of individual shrubs.

In California, yellow warblers nest primarily in riparian habitats (Grinnell and Miller 1944), but in some montane areas they also nest in a variety of shrub habitats (e.g., manzanita, ceanothus) far removed from water (Grinnell et al. 1930; Beedy and Granholm 1985; Gaines 1992).

Migrants prefer forest edges to interiors and broad-leaf trees to conifers. They can be found in a variety of habitats, including riparian, oak woodland, and suburban parks and gardens (Dunn and Garrett 1997).

Reproduction

Females lay four to five eggs and incubate them for 11–13 days. Both parents feed young at the nest until the young fledge at 8–10 days. The daily survival rates for eggs/young in nests are 94–99%. (Lowther et. al. 1999.)

Dispersal Patterns

There is no information on dispersal of young or adults from breeding sites. There are data that show a strong tendency for breeding and wintering site fidelity with individuals returning in successive years. (Lowther et. al. 1999.)

Longevity

Survival rates for adults are difficult to measure; however, 53% of banded adults returned from their wintering grounds to their previous breeding locations. This is a conservative estimate of survivorship, because it presumes that nonreturning birds died rather than relocated to a different breeding area. The oldest known yellow warbler reached 8 years 11 months. (Lowther et. al. 1999.)

Sources of Mortality

There are few data on predation of adults; however, long-tailed weasel (*Mustela frenata*) is a known predator. Nest predation by various squirrels, snakes, jays, foxes, American crow (*Corvus branchrynchos*), striped skunk (*Mephitis mephitis*), raccoon (*Procyon lotor*), and domestic cat (*Felis domesticus*) has been documented. (Lowther et. al. 1999)

Behavior

Yellow warblers are highly territorial on both breeding and wintering grounds. Breeding territories in Utah averaged 0.14 hectare (0.35 acre), and winter territories in southern Mexico averaged 0.05 hectare (1.24 acre). Extra-pair copulation has been documented, so at least some males visit rival territories (Lowther et. al. 1999). Apparently, territory size has not been estimated in Central Valley riparian areas. Manolis (1973) reported a density of 34 territorial males/square kilometer (13/square mile) (four singing males on the study plot) in a disturbed riparian woodland west of Chico, Butte County. This was in contrast to none found by Gaines (1973) during a concurrent study in riparian woodland north of Glenn along the Sacramento River. Grinnell et al. (1930) reported eight singing males on a 4-hectare (10-acre) island in the Sacramento River near Red Bluff;

this report may pertain to migrants because the early May observation date was within the peak migration period. Yellow warbler populations may increase dramatically with the elimination of cattle grazing in willow riparian habitats (Taylor and Littlefield 1986).

Yellow warblers feed primarily on arthropods and rarely on wild fruit. A study of stomach samples of 98 yellow warblers from California found that their diet comprised 30% bees, wasps, and ants; 18% caterpillars; 19% bugs (Hemiptera); 16% beetles; 9% flies; and the balance of other arthropods and fruit (Beale 1907). Yellow warblers actively glean insects from leaves and occasionally sally to capture flying insects. Males often forage higher in trees than females (Lowther et. al. 1999). A study of foraging behavior in eastern songbirds showed that yellow warblers are very adaptable, allowing them to respond to local or changing environmental conditions in search of prey (Petit et al. 1990).

Movement and Migratory Patterns

Yellow warblers migrate to the neotropics annually. Spring migration in central California takes place from early April until late May, while fall migration begins in late July and lasts until mid-October (Dunn and Garret 1997).

Ecological Relationships

Yellow warblers often form flocks during migration and will also join mixed-species flocks with other warblers, vireos, and flycatchers. Territorial birds are highly aggressive towards other bird species and will often initiate chasing, especially near the nest or on winter territories. Nest parasitism by brown-headed cowbirds has greatly reduced some populations (Lowther et. al. 1999), including most former nesting populations in the Central Valley (Remsen 1978).

Population Threats

Loss and fragmentation of riparian habitat and brown-headed cowbird brood parasitism are the major factors contributing to the historic decline in the Central Valley (Gaines 1974, Heath 2002). These threats still exist and may impede any recolonizaton attempts in the Central Valley.

Yellow-breasted Chat (*Icteria virens*)

Status

FEDERAL: None.

STATE: Bird Species of Special Concern, Priority 3.

OTHER: California Partners in Flight: high-priority riparian obligate species (Evans 1997).

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Yellow-breasted chat is a widespread summer resident of the eastern United States; it has a much more fragmented distribution in the west, where its range includes the Cascade Range; central Oregon valleys; southern Idaho; northern Nevada; and portions of California, Utah, western Colorado, and central Arizona (Eckerle and Thompson 2001).

California

Yellow-breasted chats breed primarily in the northern third of the state and are currently scarce in central and southern California (Comrack 2002). However, a significant but declining population occurs along the lower Colorado River Valley in southern California (Rosenberg et al. 1991).

Placer County Phase I Planning Area

Historical

There is no information available on the historical distribution of yellow-breasted chats in Placer County, but they probably occurred historically in foothill riparian habitats in the Phase I Planning Area.

Current

There are no known populations in the Central Valley (western) portion of the Phase I Planning Area (Easterla pers. comm.). Yellow-breasted chats are uncommon nesting birds in foothill riparian areas within the Phase I Planning Area (Webb 2003; Figure 49).

Population Status and Trends

North America

There is little evidence of population decreases or increases over large sections of the species' range (Eckerle and Thompson 2001).

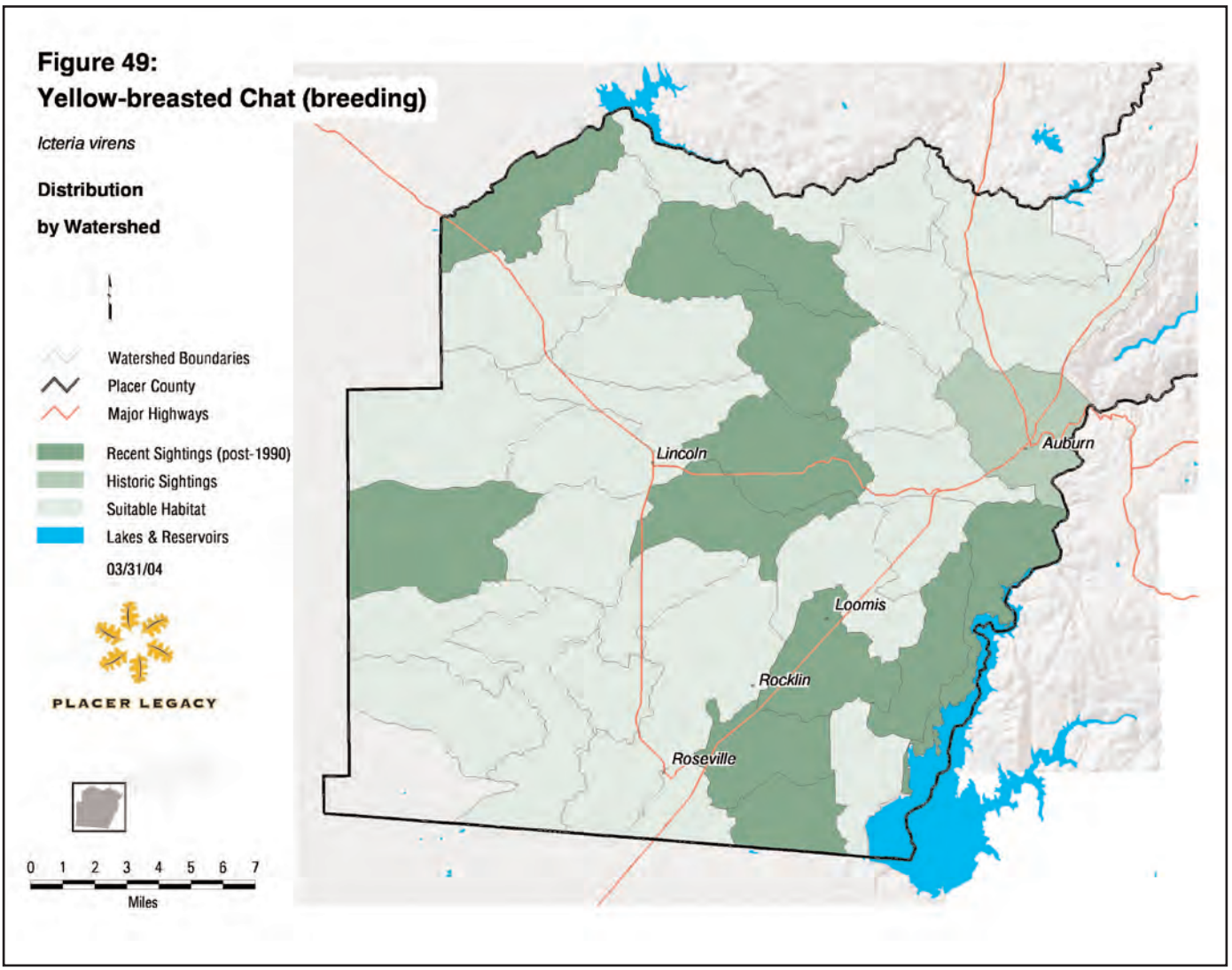
California

The southern California and Monterey County populations have declined in tandem with the destruction of much of the species' riparian habitat and increased brown-headed cowbird (*Molothrus ater*) nest parasitism (Garrett and Dunn 1981; Roberson and Tenney 1993). Although Gaines (1974) did not consider yellow-breasted chat to be decreasing in the Sacramento Valley, the species has since been reported to be rare or absent from the lower Sacramento Valley. Its statewide range has decreased by 35% during the twentieth century (Comrack 2002). There are no data on population trends in northern California, but local populations may be increasing (Comrack 2002).

Placer County Phase I Planning Area

There are no data on population trends in the Phase I Planning Area. However, because populations in the lower Sacramento Valley have been largely extirpated (Comrack 2002), it is reasonable to assume that, concomitant with habitat loss, populations in the Phase I Planning Area have also disappeared.

Yellow-breasted Chat Photo by Mike Danzenbaker



Natural History

Habitat Requirements

In northern and central California, yellow-breasted chats require riparian woodland or riparian shrub thickets with dense vegetation typically comprised of Himalayan blackberry (*Rubus discolor*), wild grape (*Vitis* sp.), and/or willows (*Salix* spp.) (Grinnell et al. 1930; Grinnell and Miller 1944; Comrack 2002). Tall willows, cottonwood (*Populus* sp.), and sycamore (*Platanus* sp.) are often used for song perches (Grinnell and Miller 1944; Dunn and Garrett 1997).

Reproduction

The breeding season generally begins in April or early May and can last until August. Males arrive on the breeding grounds shortly before females in April–late May. Little is known about pair formation and territory establishment for this species. Females initiate nest construction, which begins shortly after pair formation. Eggs are typically laid May–July. Females lay three to six eggs and incubate them for 11–12 days; both parents feed the nestlings until they fledge at approximately 9 days. No data are available on the reproductive success of the California population. (Eckerle and Thompson 2001.)

Dispersal Patterns

Little information is available on juvenile dispersal. Banding studies in Indiana showed that many juveniles moved away from the forests where they were born. Data on postbreeding dispersal are also scarce. Data from the eastern United States indicate an extremely low fidelity to breeding sites between years; however, in southern California the limited amount of available habitat may foster a higher level of breeding site fidelity. (Eckerle and Thompson 2001.)

Longevity

No data are available on survival rates of adults or fledglings. The oldest known individual was 8 years11 months (Eckerle and Thompson 2001).

Sources of Mortality

There is no information on sources of mortality in the western United States. Potential nest predators include western scrub-jay (*Aphelocoma californica*), snakes, and mammals such as raccoon (*Procyon lotor*), rats, and squirrels.

Behavior

In the eastern United States, yellow-breasted chat territory sizes ranged from 0.35 to 1.7 hectares (0.86 to 4.32 acres). Studies in the eastern United States found that effectiveness of territory defense decreased with increasing densities of males in a given area (Eckerle and Thompson 2001). There are no published data on territory sizes of yellow-breasted chats in California. Laymon (1984) estimated densities of 17–23 pairs/square kilometer (44–60 pairs/square mile) in riparian woodland on Dog Island in the Sacramento River near Red Bluff, Tehama County.

Yellow-breasted chats eat a variety of arthropods, including beetles and weevils, true bugs, ants, bees, caterpillars, and spiders; they forage in dense thickets, gleaning off leaves and twigs. They also eat fruit, especially blackberries (*Rubus* sp.), elderberries (*Sambucus* sp.), and wild grapes. (Eckerle and Thompson 2001.)

They do not join mixed-species flocks, even during postbreeding dispersal, migration, or on their wintering grounds where many migrant species join mixed-species flocks (Rappole and Warner 1980).

Movement and Migratory Patterns

Yellow-breasted chats migrate annually between their breeding grounds in North America and their wintering grounds in Mexico and Central America. For populations that breed in southern California, spring migration from the wintering grounds lasts from mid-April to late May; fall migration from the breeding grounds lasts from mid-July to mid-September (Dunn and Garrett 1997).

Ecological Relationships

Yellow-breasted chats are highly susceptible to brood parasitism by brown-headed cowbirds (Eckerle and Thompson 2001). A study of nesting chats in Missouri concluded that cowbird parasitism rates increased and nest predation decreased as habitat patch size increased (Burhans and Thompson 1999).

Population Threats

The loss and degradation of riparian woodland with dense understory is a continuing threat to the Central Valley population of yellow-breasted chat, as is brown-headed cowbird brood parasitism. Grazing that degrades suitable habitat has been implicated in local population declines (Sedgwick and Knopf 1987).

Grasshopper Sparrow (*Ammodramus savannarum*)

Status

FEDERAL: None.

STATE: Bird Species of Special Concern, Priority 2.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Grasshopper sparrows are primarily found from the Great Plains to the eastern seaboard, with a few isolated populations in the western United States, including some in California (Vickery 1996).

California

Grinnell and Miller (1944) described grasshopper sparrow's occurrence in California as "sparse and irregularly distributed" from Mendocino, Trinity, Shasta, and Lassen Counties south to San Diego County and west of the Sierra Nevada and desert regions. Grasshopper sparrows are now known from Del Norte and Siskiyou Counties and many additional areas that were unknown to Grinnell and Miller (Unitt 2002; Sterling pers. comm.). However, their statewide distribution is still best described as sparse and irregular.

Placer County Phase I Planning Area

Historical

There are no historical records for Placer County.

Current

One singing male was found in April 1999 in annual grassland east of Lincoln, but this individual was present only for a few days. A fall migrant was found in September 1999 along Brewer Road. (Easterla pers. comm.; Webb 2003; Figure 50.)

Population Status and Trends

North America

Breeding Bird Survey data are inadequate to assess population trends throughout the species' range (Sauer et al. 2001). Regional population trends are related to land use. For example, an 85% decline occurred in Illinois during the past 35 years due to conversion of pasture to row crops. A severe decline was also noted in Florida due to conversion of native prairie to agriculture, and an increase was observed in South Carolina, perhaps due to an increase in pasture. (Vickery 1996.)

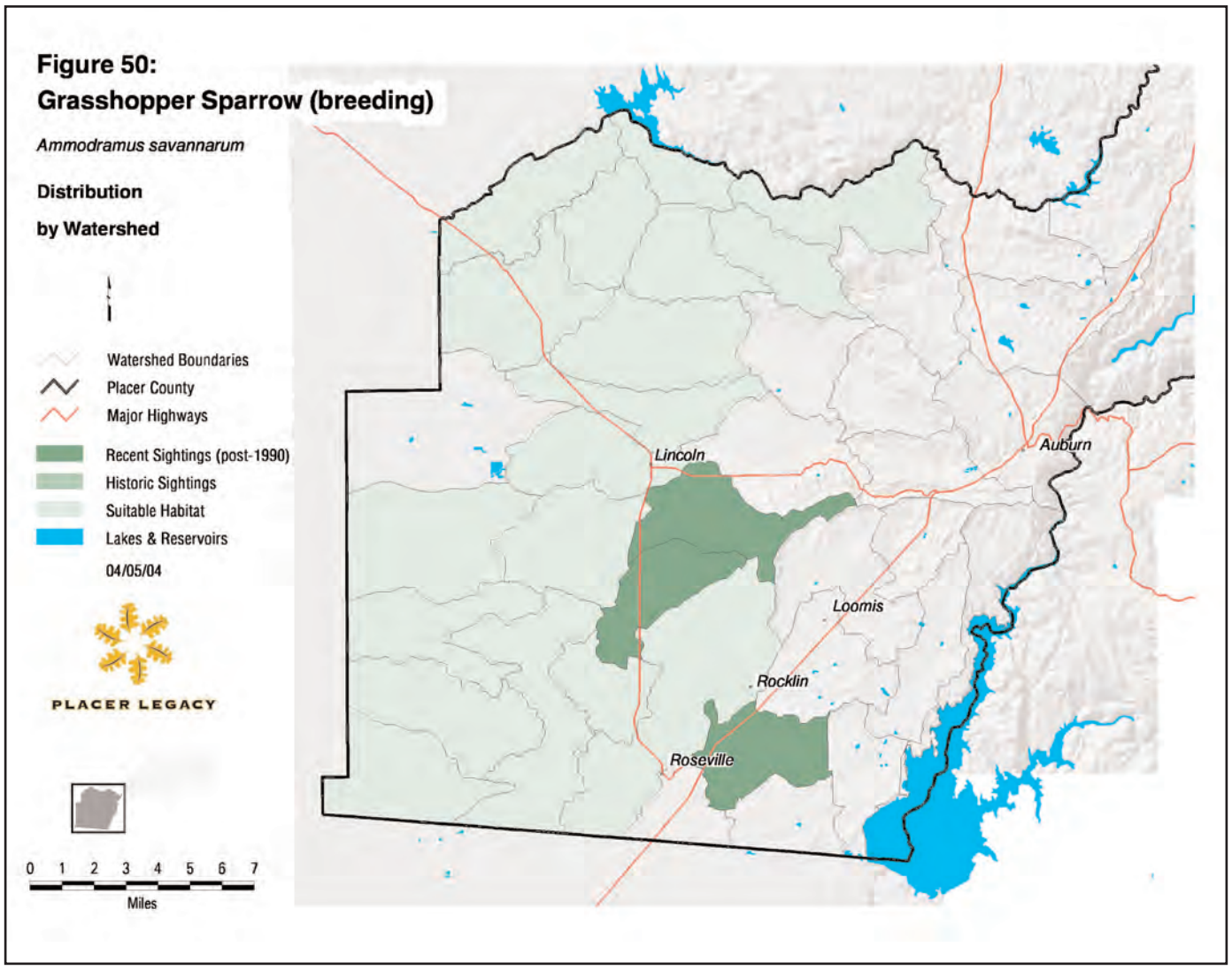
California

Breeding Bird Survey data are inadequate to assess population trends in California (Sauer et al. 2001). Many local populations in southern California have been extirpated due to habitat loss resulting from urbanization and from conversion of grasslands to cultivated agriculture. (Unitt 2002).

Placer County Phase I Planning Area

Because grasshopper sparrows have only been recorded twice in Placer County, any discussion of population trends would be unfounded. However, they are often (and easily) overlooked in grassland habitats even when present. Further searching of annual grasslands in the Phase I Planning Area could reveal that grasshopper sparrows are rare to uncommon breeding birds that may also be present in winter.

Grasshopper Sparrow Photo by Pete LaTourette



Natural History

Habitat Requirements

In California, grasshopper sparrows require dry, well-drained grasslands with patches of bare ground (Grinnell and Miller 1944). These grasslands often include scattered, taller shrubs or annuals that are used for song perches (Grinnell and Miller 1944; Vickery 1996; Unitt 2002). In Placer and adjacent counties, grasshopper sparrows have been found on rolling hills with extensive patches of rye grass (*Secale* spp.) along the western and eastern edges of the Central Valley (Easterla pers. comm.; Jones & Stokes file data; Sterling pers. comm.). There are few regional records of this species, most likely because of the lack of focused surveys in potential habitat areas. More field investigations are needed to provide data that describe the full range of suitable grassland habitats for grasshopper sparrows in the Sacramento Valley and adjacent foothills. Elsewhere in its range, native bunchgrasses are preferred over sod-forming grasses (Unitt 2002).

Reproduction

Females lay three to six eggs and incubate them for 11–13 days. Both parents and, occasionally, related adult and juvenile helpers feed the nestlings, which remain in the nest for 8–9 days. Grasshopper sparrows frequently renest in response to nest predation; rates may exceed 50%. Rates of brown-headed cowbird (*Molothrus ater*) brood parasitism are reportedly lower for grasshopper sparrow than for other grassland bird species, presumably because grasshopper sparrow nests are more difficult to find. (Vickery 1996.)

Dispersal Patterns

In San Diego County, 20% (N = 35) of male grasshopper sparrows returned to their successful breeding locations the following year (Collier 1994 *in* Vickery 1996). The patchy distribution of occupied habitat in California is conducive to high return rates because returning spring migrants have few alternatives to their former breeding sites. There are no data on dispersal to other locations, but anecdotal evidence suggests that grasshopper sparrows can colonize new locations that become suitable habitat as a result of reduction in grazing and/or higher seasonal rainfall (Sterling pers. comm.).

Longevity

A Florida banding study suggested that resident males have a mean lifespan of 2.9 years; one individual lived for at least 6.5 years (Delaney et al. 1993 *in* Vickery 1996). However, the northern and central California populations are likely to be highly migratory; consequently, their annual survival rates are probably considerably lower due to mortality during migration.

Sources of Mortality

Hay and grass mowing during the nesting season (conducted earlier in spring now than was done historically) has resulted in nest failure and mortality of young and/or eggs. Predation on adults by loggerhead shrikes (*Lanius ludovicianus*) and on nestlings by corvids, snakes, and a variety of mammals may significantly affect small populations. Nest predation rates are higher near woodlands and brush fields due increased to exposure to avian and mammalian predators. (Vickery 1996.)

Behavior

Grasshopper sparrows often nest semi-colonially in clusters of territories (Grinnell and Miller 1994; Vickery 1996). Reported mean territory sizes vary considerably throughout the species' distribution and range from 0.19 to 1.40 hectares (0.47 to 3.46 acres) (Vickery 1996). Populations in Maine require habitat patches greater than 100 hectares (247 acres) (Vickery et al. 1994 in Lyon 2000). Territory sizes and habitat patch requirements have not been studied in California. Potential territory sizes in the Phase I Planning Area probably depend on habitat quality as well as patch size.

Grasshopper sparrows forage on the ground primarily for grasshoppers, but other insects, including bees and wasps, beetles, and caterpillars, are also eaten. Studies have shown that insects account for 61% and 29% of the summer and fall diets, respectively. The remainder of the diet is comprised of seeds. Stomach analysis in California (N = 8) found seeds from knotweed (*Polygonum* sp.), campion (*Lychnis* sp.), oats (*Avena* sp.), and pigweed (*Amaranthus* sp.). (Vickery 1996).

Movement and Migratory Patterns

Grasshopper sparrow movement and migration patterns in California are not well known. Although migrating individuals have been documented in Placer County (Webb 2002) and elsewhere in California (Grinnell and Miller 1944; Garrett and Dunn 1981; Unitt 2002), wintering birds are occasionally found in the Central Valley (Easterla pers. comm.; Jones & Stokes file data). Due to their cryptic nature and the vast expanses of potential wintering habitat, this sparrow's local movements and wintering distributions are difficult to study.

Ecological Relationships

Grasshopper sparrows are not known to interact territorially with other grassland birds, although countersinging with savannah sparrows (*Passerculus sandwichensis*) has been documented (Vickery 1996). Their territories are known to overlap with those of savannah sparrows without any sign of aggressive interactions (Vickery 1996).

Grasshopper sparrows do not flock together nor do they join mixed-species flocks as social members (Vickery 1996), but they occasionally forage in association with other species in areas where seeds are concentrated (Sterling pers. comm.).

Population Threats

The primary population threats come from development of grasslands for housing and commercial buildings, overgrazing of occupied habitat, and mowing of hay and grass before the young have fledged (Unitt 2002; Vickery 1996).

Modesto Song Sparrow (*Melospiza melodia mailliardi*)

Status

FEDERAL: None.

STATE: Bird Species of Special Concern, Priority 3.

OTHER: California Partners in Flight: high-priority riparian obligate species by (Evans 1997).

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

California

The Modesto song sparrow is endemic to California, where it occurs in the Central Valley from Colusa to Stanislaus Counties and east of the Suisun Marshes (Grinnell and Miller 1944). Current population centers are in the Butte Sink, Butte County, and the Sacramento–San Joaquin Delta (Gardali 2002).

Placer County Phase I Planning Area

Historical

Although Grinnell and Miller (1944) did not cite specific records for Placer County, they described the general range of the subspecies to include the Phase I Planning Area.

Current

Modesto song sparrows are common residents of riparian and marsh habitats throughout the Phase I Planning Area (Webb 2003; Figure 51). This is the only subspecies that is known to occur in western Placer County and adjacent Sierra foothill counties (Grinnell and Miller 1944; Gardali 2002).

Population Status and Trends

California

Humple and Geupel (2000) reported that Modesto song sparrows were absent from most riparian habitat in the Sacramento Valley but were still common in the upper San Joaquin Valley. Because 90% of the subspecies' habitat has been destroyed, there is likely to have been a concomitant decline in its population (Gardali 2002).

Placer County Phase I Planning Area

There is no information on population trends for the Phase I Planning Area.

Natural History

Habitat Requirements

Modesto song sparrows are found in a variety of habitats including riparian willow (*Salix* spp.) thickets, valley oak (*Quercus lobata*) riparian with an understory of blackberry (*Rubus* spp.), ruderal areas along levees and irrigation canals, and cattail (*Typha* sp.) and tule (*Scirpus* sp.) marshes (Gardali 2002).

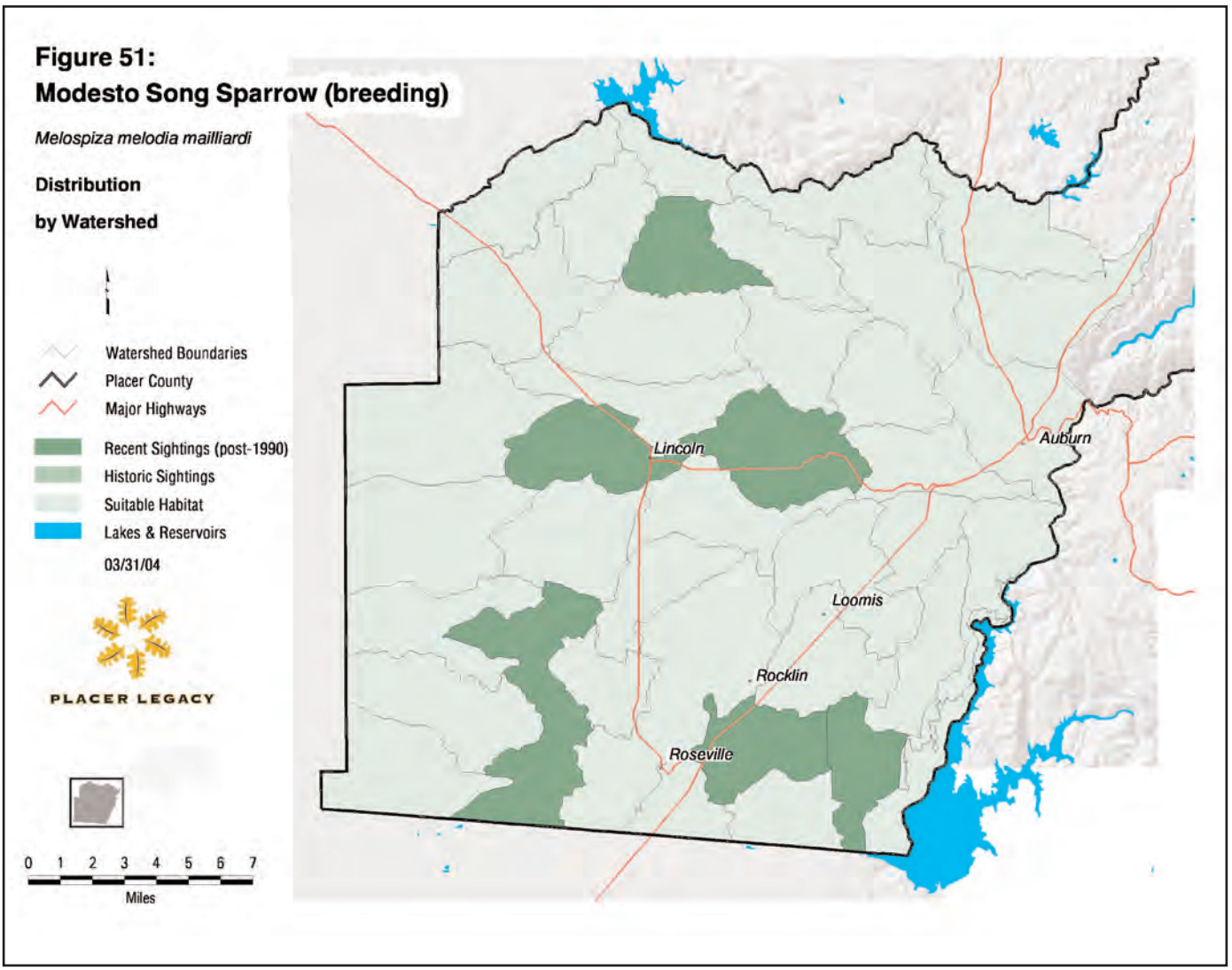
Reproduction

There is very little information for this subspecies. A study at the Cosumnes River Preserve found "low" reproductive success due to high levels of nest predation, but the study also found a relatively low rate of brown-headed cowbird (*Molothrus ater*) brood parasitism (18%) (N = 62) (DiGaudio and Geupel 1998 in Gardali 2002). Song sparrow females lay three to four eggs, incubate them for 12–14 days, and feed the young for 9–12 days until they leave the nests (Humple and Geupel 2000).

Dispersal Patterns

There is very little information on dispersal for California populations, and the subspecies' relatively sedentary habits may explain why large tracts of suitable habitat in the lower Sacramento Valley remain unoccupied. However, Modesto song sparrows colonized created wetlands at the Sacramento Wastewater Treatment Plant within 2 years of construction, suggesting that the relatively larger populations in the

Modesto Song Sparrow Photo by Pete LaTourette



northern San Joaquin Valley readily disperse and colonize habitats (Gardali 2002).

Longevity

There is no available information on longevity of Modesto song sparrow.

Sources of Mortality

Potential predators identified at the Cosumnes River Preserve that are also abundant in the Phase I Planning Area include feral cat, raccoon (*Procyon lotor*), skunks, Virginia opossum (*Didelphis virginiana*), rats, and corvids (DiGaudio and Geupel 1998 in Gardali 2002).

Behavior

There is no information on territory size or density of this subspecies. However, population densities of song sparrows in Marin County riparian habitats ranged from 4.4 to 8.1 territories/creek kilometer (7.1 to 13.1 territories/mile). Mean densities in wetlands in British Columbia ranged from 1.7 to 5.6 pairs/hectare (0.7 to 2.3 pairs/acre) (Rogers et al. 1997).

Analyses of song sparrow stomach contents from throughout California found that the diet varies considerably through the year. Overall diet consisted of 73% "weed" seeds and fruit (*Rubus* and elderberry) with the remainder consisting of beetles, ants, bees, wasps, flies, caterpillars, and true bugs. Insects comprised more than 71% of the diet during May (breeding season) but only 3% during September. (Beale 1910.)

Movement and Migratory Patterns

Modesto song sparrows do not migrate.

Ecological Relationships

Modesto song sparrows do not form conspecific flocks but associate with other sparrows at concentrations of seeds (Sterling pers. comm.). Nest parasitism by brown-headed cowbirds has contributed to a demographic sink in a wetland population in British Columbia (Rogers et al. 1997).

Population Threats

High predation associated with increased habitat fragmentation in rapidly developing areas of the Central Valley is considered the most serious population threat. Brown-headed cowbird parasitism rates are low and are not considered to be a significant factor in reproductive success. Destruction of habitat is more important now than in historical times because 95% of the habitat has already been destroyed (Humple and Geupel 2000; Gardali 2002.)

Tricolored Blackbird (*Agelaius tricolor*)

Status

FEDERAL: Species of Concern.

STATE: Species of Special Concern, Priority 1.

OTHER: None.

RECOVERY PLAN: None.

PLACER NCCP/HCP CATEGORY: Class 3.

Distribution

North America

Tricolored blackbirds are largely endemic to California, and more than 99% of the global population occurs in the state. In any given year, more than 75% of the breeding population can be found in the Central Valley (Hamilton 2000). Small breeding populations also exist at scattered sites in Oregon, Washington, Nevada, and western coastal Baja California (Beedy and Hamilton 1999).

California

The species' historical breeding range in California included the Sacramento and San Joaquin Valleys, lowlands of the Sierra Nevada south to Kern County, the coast region from Sonoma County to the Mexican border, and sporadically on the Modoc Plateau (Dawson 1923; Neff 1937; Grinnell and Miller 1944).

Population surveys and banding studies of tricolored blackbirds in the Central Valley from 1969 through 1972 concluded that their geographic range and major breeding areas were unchanged since the mid-1930s (DeHaven et al. 1975a). In recent decades, breeding colonies have been observed in all Central Valley counties (Beedy and Hamilton 1997, 1999; Hamilton 2000). The species also breeds locally along the California coast from Humboldt to San Diego Counties; on the Modoc Plateau and western edge of the Great Basin (mostly Klamath Basin); in lowlands surrounding the Central Valley; and in western portions of San Bernardino, Riverside, and San Diego Counties. During winter, virtually the entire population of the species withdraws from Washington, Oregon (although a few remain), Nevada, and Baja California, and wintering populations shift extensively within their breeding range in California (Beedy and Hamilton 1999).

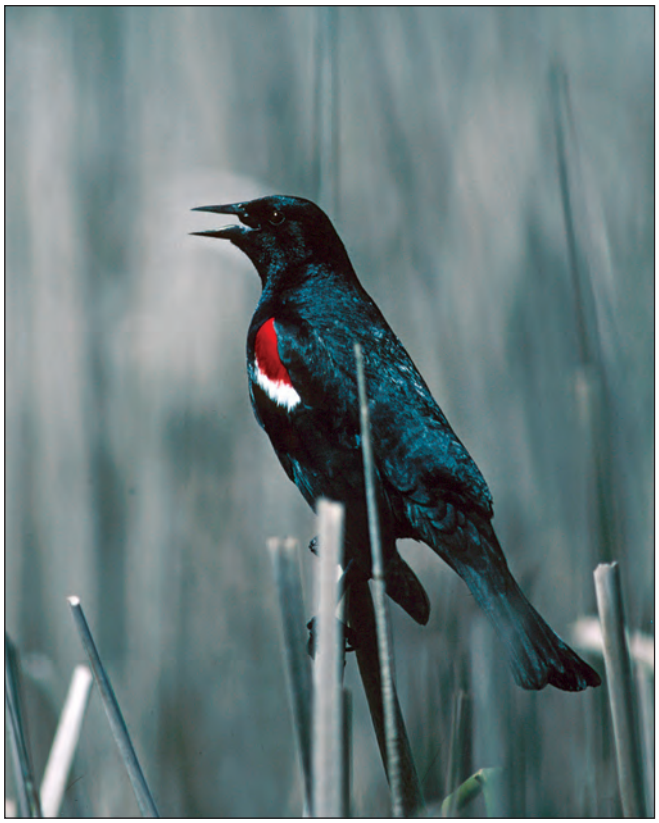
Placer County Phase I Planning Area

Historical

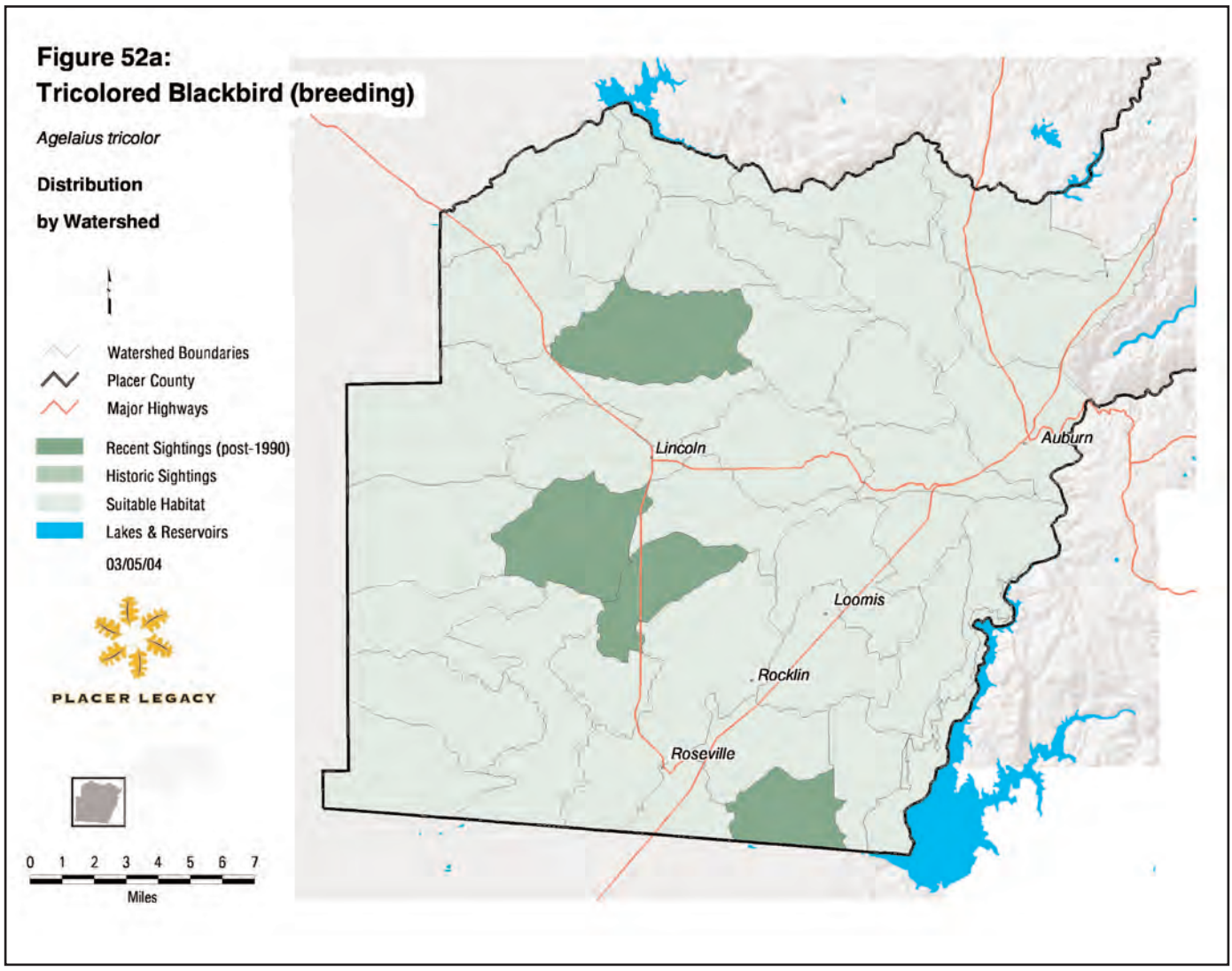
Neff (1937) found only two tricolored blackbird breeding colonies in Placer County. Both of these were in cattails (*Typha* sp.) along canals near the city of Lincoln in 1933 and 1936 and contained about 1,000 and 1,500 nests, respectively. A colony (unknown number of pairs) was found nesting in cattails in a wet pasture near Lincoln in 1971 (Hosea 1986; California Natural Diversity Database 2003). Another colony of about 2,000 pairs was found in cattails around a marsh on the Chamberlain Ranch (north-northwest of Lincoln) in 1971 (DeHaven pers. comm.).

Current

In the past 10 years, only two tricolored blackbird colonies in Placer County have been reported to the California Natural Diversity Database (2003): a 1994 colony in cattails at a farm pond near Roseville (unknown number of pairs); and a 1997 colony at a cattail marsh near Folsom Lake (unknown number of pairs). In April 2000, a colony of about 5,000 individuals was found in bulrushes adjacent to Industrial



Tricolored Blackbird Photo by Mike Danzenbaker



Boulevard, and a second colony of about 1,000 individuals was observed in Himalayan blackberry (*Rubus discolor*) on Riosa Road (Hamilton 2000). In March 2003, a colony of about 500–1,000 individuals was found in a marsh adjacent to Twelve Bridges Road and a housing development east of Highway 65 (Jones & Stokes data). All these colony locations are in rapidly developing areas of Placer County, and their current suitability for breeding and foraging tricolored blackbirds has not been assessed (Figure 52a).

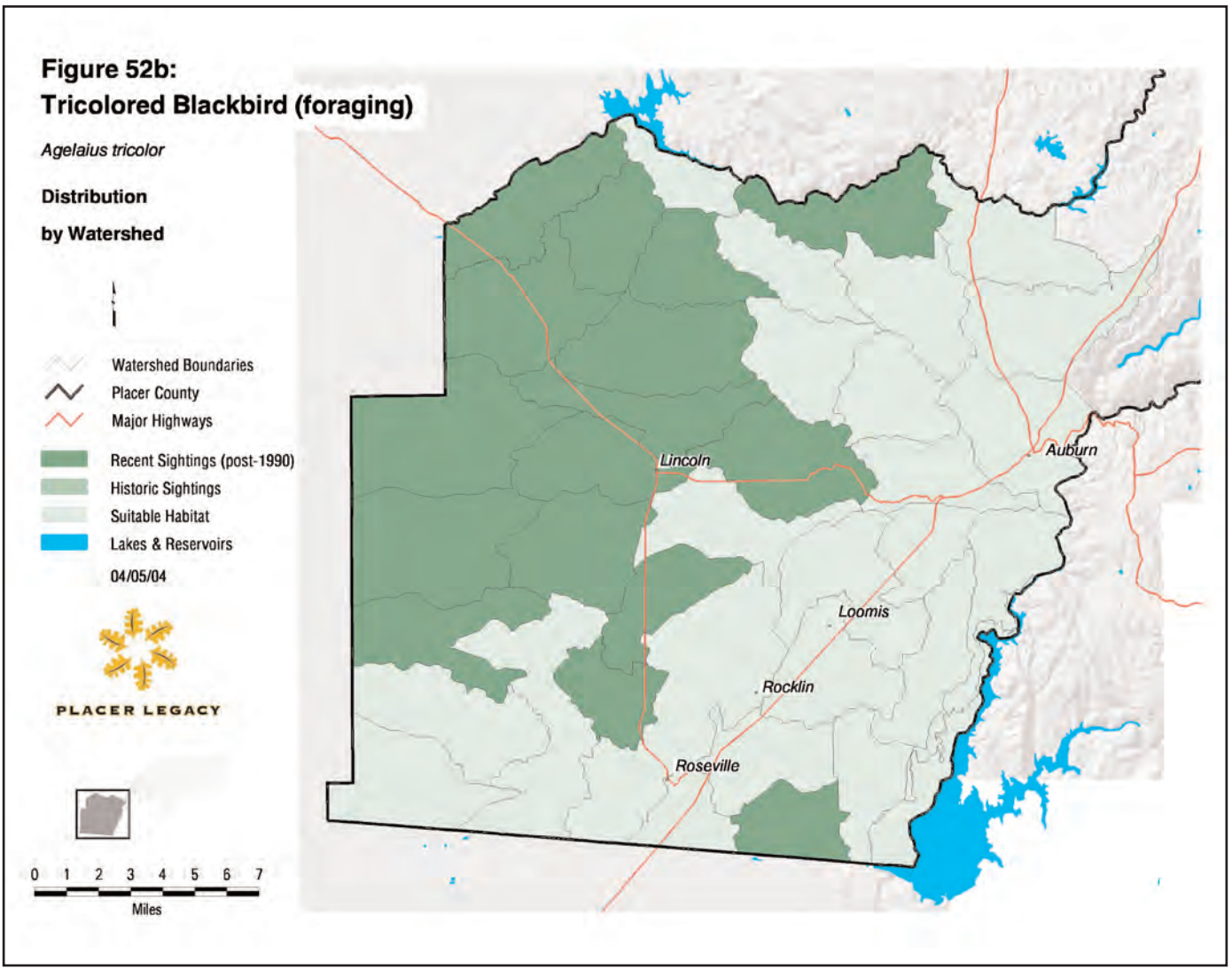
Tricolored blackbirds are common nonbreeding visitors to lowlands from early October to early March. They are often found in mixed flocks with other species of blackbirds, or they may occur as single-species flocks in annual grasslands, wetlands, and agricultural areas. Suitable foraging habitat for this species exists in most watersheds below about 150 meters (500 feet) in the Phase I Planning Area that have not experienced extensive urbanization (Figure 52b).

Population Trends

North America/California

The first systematic surveys of tricolored blackbird's population status and distribution were conducted by Neff (1937, 1942). During a 5-year interval, he found 252 breeding colonies in 26 California counties; the largest colonies were in rice-growing areas of the Central Valley. He observed as many as 736,500 adults per year (1934) in just eight Central Valley counties. The largest colony he observed was in Glenn County; it contained more than 200,000 nests (about 300,000 adults) and covered almost 24 hectares (60 acres). Several other colonies in Sacramento and Butte Counties contained more than 100,000 nests (about 150,000 adults).

DeHaven et al. (1975a) estimated that the overall population size in the Sacramento and northern San Joaquin valleys had declined by more than 50% since the mid-1930s. They performed intensive surveys and



banding studies in the areas surveyed by Neff (1937) and observed significant declines in tricolored blackbird numbers and the extent of suitable habitat in the period since Neff's surveys. Orians (1961a) and Payne (1969) 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.

The U.S. Fish and Wildlife Service, the California Department of Fish and Game, and California Audubon cosponsored intensive, volunteer tricolored blackbird surveys in suitable habitats throughout California in 1994, 1997, 1999, and 2000 (Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000). Local, regional, and statewide tricolored blackbird populations have experienced major declines since 1994. Statewide totals of adults in four late-April surveys covering all recently known colony sites were: 369,359 (1994); 237,928 (1997); 104,786 (1999); and 162,508 (2000). These surveys also identified several important distribution and population trends for tricolored blackbirds.

- Local, regional, and statewide populations and distributions vary from year to year.
- Sixty percent of all tricolored blackbirds located in all years were found in the 10 largest colonies.
- Seventy percent of all tricolored blackbird nests and 86% of all foraging by nesting birds were on private agricultural lands.
- In some portions of their range, tricolored blackbirds have declined or been eliminated; the species has been subject to local extirpation in most of Yolo County and portions of southern Sacramento County.

Placer County Phase I Planning Area

Existing data are insufficient to estimate local population trends for tricolored blackbirds in Placer County. However, all available data suggest that their breeding population is very small (i.e., <3,000 pairs) or absent in most recent years. Similarly, most of their historical nesting and foraging habitats are near the cities of Lincoln and Roseville; the current status of these colony sites has not been determined. Rapid development in these areas may reduce their overall suitability for nesting tricolored blackbirds by conversion of existing freshwater marshes, agricultural lands, and pastures to other land uses.

Natural History

Habitat Requirements

Tricolored blackbirds have three basic requirements for selecting their breeding colony sites: open accessible water; a protected nesting substrate, including either flooded or thorny or spiny vegetation; and a suitable foraging space providing adequate insect prey within a few miles of the nesting colony (Hamilton et al. 1995; Beedy and Hamilton 1997, 1999). Almost 93% of the 252 breeding colonies reported by Neff (1937) were in freshwater marshes dominated by cattails and bulrushes (*Schoenoplectus* sp.). The remaining colonies in Neff's study were in willows (*Salix* spp.), blackberries (*Rubus* sp.), thistles (*Cirsium* and *Centaurea* spp.), or nettles (*Urtica* sp.). In contrast, only 53% of the colonies reported during the 1970s were in cattails and bulrushes (DeHaven et al. 1975a).

An increasing percentage of tricolored blackbird colonies in the 1980s and 1990s were reported in Himalayan blackberries (Cook 1996), and some of the largest recent colonies have been in silage and grain fields (Hamilton et al. 1995, Beedy and Hamilton 1997, Hamilton 2000). Other substrates where tricolored blackbirds have been observed nesting include giant cane (*Arundo donax*); safflower (*Carthamus tinctorius*) (DeHaven et al. 1975a); tamarisk trees (*Tamarix* spp.); elderberry/poison-oak (*Sambucus* spp. and *Toxicodendron diversilobum*); and riparian scrublands and forests (e.g., *Salix, Populus, Fraxinus*) (Beedy and Hamilton 1999).

Foraging habitats in all seasons include annual grasslands; wet and dry vernal pools and other seasonal wetlands; agricultural fields (e.g., large tracts of alfalfa with continuous mowing schedules and recently tilled fields); cattle feedlots; and dairies. Tricolored blackbirds also forage occasionally in riparian scrub habitats and along marsh borders. Weed-free row crops and intensively managed vineyards and orchards do not serve as regular foraging sites (Beedy and Hamilton 1997, 1999).

Grassland/vernal pool complexes and rice fields characterize the landscape in much of the species' breeding range and preferred foraging habitats in western Placer County. Ungrazed grasslands containing tall grasses (>15 centimeters [6 inches] tall) and vernal pools are preferred over dry, grazed grasslands containing short grasses. Foraging birds often congregate at the margins of wet vernal pools and within their interiors once they dry (Cook 1996).

Foraging

Foods delivered to tricolored blackbird nestlings include beetles and weevils; grasshoppers; caddisfly larvae; moth and butterfly larvae (Orians 1961a; Crase and DeHaven 1977; Skorupa et al. 1980); and, especially in rice-growing areas, dragonfly larvae (Beedy and Hamilton 1999). Breeding-season foraging studies in Merced County showed that animal matter makes up about 91% of the food volume of nestlings and fledglings, 56% of the food volume of adult females, and 28% of the food volume of adult males (Skorupa et al. 1980).

Adults may continue to consume plant foods throughout the nesting cycle but also forage on insects and other animal foods. Immediately before and during nesting, adult tricolored blackbirds are often attracted to the vicinity of dairies, where they take high-energy items from livestock feed rations. Adults with access to livestock feed, such as cracked corn, begin providing it to nestlings when they are about 10 days old (Hamilton et al. 1995). More than 88% of all winter food in the Sacramento Valley is plant material, primarily seeds of rice and other grains but also weed seeds (Crase and DeHaven 1978). In winter, tricolored blackbirds often associate with other blackbirds, but flocks as large as 15,000 individuals (almost all tricolored blackbirds) may congregate at one location and disperse to foraging sites (Beedy and Hamilton 1999).

Reproduction

Tricolored blackbirds are closely related to red-winged blackbirds (*Agelaius phoeniceus*), but the two species differ substantially in their breeding ecology. Red-winged blackbird pairs defend individual territories, while tricolored blackbirds are among the most colonial of North American passerine birds (Bent 1958; Orians 1961a, 1961b, 1980; Orians and Collier 1963; Payne 1969; Beedy and Hamilton 1999). As many as 20,000 or 30,000 tricolored blackbird nests have been recorded in cattail marshes of 4 hectares (9 acres) or less (Neff 1937; DeHaven et al. 1975a), and individual nests may be built less than 0.5 meter (1.5 feet) apart (Neff 1937). Tricolored blackbird's colonial breeding system may have adapted to exploit a rapidly changing environment where the locations of secure nesting habitat and rich insect food supplies were ephemeral and likely to change each year (Orians 1961a; Orians and Collier 1963; Collier 1968; Payne 1969).

Tricolored blackbird nests are bound to upright plant stems from a few centimeters to about 1.8 meters (6 feet) above water or ground (Baicich and Harrison 1997); however, nests in the canopies of willows and ashes may be more than 3.7 meters (12 feet) high (Hamilton pers. comm.). Their nests are rarely built on the ground (Neff 1937). Deep cup nests are constructed with outer layers of long leaves (e.g., cattail thatch, annual grasses, or forbs) woven tightly around supporting stems. The inner layers are coiled stems of grasses lined with soft plant down, mud, or algal fibers. Nest building takes about 4 days (Payne 1969).

Egg laying can begin as early as the second day after nest initiation but ordinarily starts about 4 days after the local arrival of tricolored blackbirds at breeding sites (Payne 1969). One egg is laid per day, and clutch size is typically three to four eggs (Payne 1969; Hamilton et al. 1995). Emlen (1941) and Orians (1961b) estimated the incubation period at 11 or 12 days, while Payne (1969) estimated it to be 11–14 days. About 9 days generally elapse from hatching until the oldest nestling is willing to jump from the nest when disturbed. Young require about 15 days from this prefledging date until they are independent of their parents. Thus, one successful nesting effort for a reproductive pair takes about 45 days (Hamilton et al. 1995).

Dispersal Patterns

DeHaven et al. (1975b) found that tricolored blackbirds were unlikely to nest at the sites where they hatched or where they had nested the year before (N = 298 recoveries from 45,660 banded birds). However, breeding colonies often exhibit site fidelity and traditionally use many of the same areas year after year if these sites continue to provide essential resources such as secure nesting substrates, water, and suitable foraging habitats (Beedy et al. 1991; Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000). As discussed in *Movement and Migratory Patterns* below, the distribution of tricolored blackbirds in the Central Valley varies according to relatively predictable, seasonal movements.

Longevity

Banding studies, summarized by Neff (1942) and DeHaven and Neff (1973), indicated that tricolored blackbirds can live for at least 13 years, but most live for much shorter periods. There are no annual survivorship studies of tricolored blackbird, and available banding data are inadequate to provide this information (Beedy and Hamilton 1999).

Sources of Mortality

Entire colonies (up to tens of thousands of nests) in cereal crops and silage are often destroyed by harvesting and plowing of agricultural lands in the San Joaquin Valley (Beedy and Hamilton 1999). Concentration of a high proportion of the known population in a few breeding colonies increases the risk of major reproductive failures, especially in vulnerable habitats such as active agricultural fields.

Historical accounts documented the destruction of nesting colonies by a diversity of avian, mammalian, and reptilian predators (Beedy 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*). Some large colonies (up to 100,000 adults) may lose more than 50% of nests to coyotes (*Canis latrans*), especially in silage fields, but also in freshwater marshes when water is withdrawn (Hamilton et al. 1995).

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 in Merced County; selenium toxicosis was diagnosed as the primary cause of death. At a Kern County colony, 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.

Behavior

Males defend only the immediate areas around the nests. Male territory size ranges from 1.8 square meters (19.38 square feet) (Lack and Emlen 1939) to 3.25 square meters (35 square feet) (Orians 1961b). Average size of recently established territories of six banded males at two different colonies was 3.25 square meters (35 square feet); volumetric territories in willows were calculated to be 8.5–11.3 cubic meters (300–400 cubic feet) (Collier 1968). Some Himalayan blackberry colonies have nesting densities up to six nests/square meter (0.56 nest/square foot) (Cook and Hamilton pers. comms.). After 1 week of nest-building and egg-laying, males may cease territorial defense (Orians 1961b).

Most tricolored blackbirds forage within 5 kilometers (3.1 miles) of their colony sites (Orians 1961a), but commute distances of up to 15 kilometers (9.3 miles) have been reported (Beedy and Hamilton 1999). Short-distance foraging (i.e., within sight of the colony) for nestling provisioning also is common. Both sexes are known to provision the nestlings (Beedy and Hamilton 1999).

Proximity to suitable foraging habitat appears to be extremely important for the establishment of colony sites, as tricolored blackbirds always forage, at least initially, in the field containing the colony site (Cook 1996). However, usually only a minor fraction of the area within the commuting range of a colony provides suitable foraging habitat. For example, within a 5-kilomeer (3-mile) radius there may be low-quality foraging habitats such as cultivated row crops, orchards, vineyards, and heavily grazed rangelands in association with high-quality foraging areas such as irrigated pastures, lightly grazed rangelands, vernal pools, and recently mowed alfalfa fields (Beedy and Hamilton 1999; Cook 1999).

Movement and Migratory Patterns

During the breeding season, tricolored blackbirds exhibit itinerant breeding (Hamilton 1998). In the north Central Valley and northeastern California, individuals move after first nesting attempts, both successful and unsuccessful (Beedy and Hamilton 1997). Banding studies indicate that significant movement into the Sacramento Valley occurs during the post-breeding period (DeHaven et al. 1975b).

In winter, numbers of tricolored blackbirds decrease in the Sacramento Valley and increase in the Sacramento–San Joaquin River Delta and north San Joaquin Valley (Neff 1937; Orians 1961a; Payne 1969; DeHaven et al. 1975b). By late October, large flocks also congregate in pasturelands in southern Solano County and near dairies on Point Reyes Peninsula in Marin County (Beedy and Hamilton 1999). Other birds winter in the central and southern San Joaquin Valley. Concentrations of more than 15,000 wintering tricolored blackbirds may gather at one location and disperse up to 32 kilometers (20 miles) to forage (Neff 1937; Beedy and Hamilton 1999). Individual birds may leave winter roost sites after less than 3 weeks and move to other locations (Collier 1968), suggesting winter turnover and mobility. In early March/April, most birds vacate the wintering areas in the Central Valley and along the coast and move to breeding locations in the Sacramento and San Joaquin Valleys (DeHaven et al. 1975b).

Ecological Relationships

Tricolored blackbirds occupy a unique niche in the Central Valley/coastal marshland ecosystems. In areas where numbers are high, they both aggressively and passively dominate and often displace sympatric marsh-nesting species, including red-winged blackbirds and yellow-headed blackbirds (*Xanthocephalus xanthocephalus*), through sheer numbers (Orians and Collier 1963; Payne 1969).

Population Threats

The greatest threats to this species are the direct loss and alteration of habitat, but other human activities and predation also threaten tricolored blackbird populations in the Central Valley (Beedy and Hamilton 1999).

Habitat Loss and Alteration

Most native habitats that once supported nesting and foraging tricolored blackbirds in the Central Valley have been altered by urbanization and unsuitable agricultural uses, including vineyards, orchards, and row crops (Frayer et al. 1989; Wilen and Frayer 1990). In Sacramento County, a historic breeding center of the species, the conversion of grassland and pastures to vineyards expanded from 3,050 hectares (7,536 acres) in 1996 to 5,330 hectares (13,171 acres) in 1998 (DeHaven 2000). Many former agricultural areas within the historical range of tricolored blackbird are now being urbanized; in western Placer County, where tricolored blackbirds forage in the ungrazed annual grasslands associated with rural subdivisions, suitable habitat will be largely eliminated as current land conversion patterns continue.

In some places, most historical tricolored blackbird breeding and foraging habitats have been eliminated and there is currently little or no breeding effort where there once were large colonies (Orians 1961a; Beedy et al. 1991). Elsewhere, tricolored blackbirds have shifted from cattails as a primary nesting substrate (Neff 1937) to Himalayan blackberries (DeHaven et al. 1975a), and more recently to cereal crops and barley silage (Hamilton et al. 1995).

Other Human Activities

Nests and nest contents in cereal crops and silage are often destroyed by agricultural operations (Hamilton et al. 1995; Beedy and Hamilton 1997). Harvesting of silage and plowing of weedy fields are currently the most common reasons tricolored blackbird nesting colonies are destroyed on agricultural lands. Other factors that may affect the nesting success of colonies in agricultural areas include herbicide and pesticide applications and spraying for mosquito abatement (Beedy and Hamilton 1999).

Predation

Predation is at present (i.e., 1985–2003) a major cause of complete nesting failure at some tricolored blackbird colonies in the Central Valley. Historical accounts documented the destruction of nesting colonies by a diversity of avian, mammalian, and reptilian predators. Recently, especially in permanent freshwater marshes of the Central Valley, entire colonies (>50,000 nests) have been lost to black-crowned night-herons, common ravens, coyotes, and other predators (Beedy and Hamilton 1999).