Appendix X
Species Profiles

Note to Reviewers: These species profiles will likely be an appendix to the HCP/NCCP. The profiles included are dependent on the covered species list. Because this list is preliminary, the species profiles may change. In addition to the range maps provided in these profiles, we will be adding a map illustrating the potential suitable habitat for each covered species within the inventory area based on habitat models we are currently developing. These maps will have all California Department of Fish and Game NDDB records superimposed on them for reference. Please note that these profiles are not intended to be treatises on each covered species. Rather, they are intended to summarize ecological information most relevant to this HCP/NCCP. If you have suggestions to cite additional literature or unpublished reports, please provide us with copies of these documents if possible.

Townsend's Western Big-Eared Bat (*Corynorhinus townsendii townsendii*)

Status

State: Species of Concern
Federal: Species of Concern
Other: Western Bat Working Group High Priority Species

Population Trend

Global: Declining (Pierson et al. 1999)
State: Declining (Pierson 1988, Pierson and Rainey 1996)
Within Inventory Area: Unknown

Data Characterization

The location database for the Townsend's western big-eared bat (*Corynorhinus townsendii townsendii)* within its known range in California includes 20 data records dated from 1987 to 2000. Of these records, 6 were documented within the past 10 years; of these, 1 was of high precision and can be accurately located within the its survey area. None of these records are located within the inventory area.

A moderate amount of literature is available for the Townsend's western big-eared bat because of its rare and declining status. Most of the information available is on the natural history, distribution, population status, and threats to this species. A conservation assessment and conservation strategy has been published.

Range

Townsend's big-eared bats occur throughout most of western North America from British Columbia to central Mexico, east to the Black Hills of South Dakota, and across Texas to the Edwards Plateau (Hall 1981, Kunz and Martin 1982). Isolated, relictual populations of this species are found in the southern Great Plains and the Ozark and Appalacian Mountains (Hall 1981, Kunz and Martin 1982). The subspecies *pallescens* occurs in Washington, Oregon, California, Nevada, Idaho, Arizona, Colorado, New Mexico, Texas, and Wyoming. The subspecies *townsendii* occurs in Washington, Oregon, California, Nevada, Idaho, and possibly southwestern Montana and northwestern Utah (Hadley 1959, Hall 1981). In California, the boundary between *pallescens* and *townsendii* runs north-south approximately through the center of the Central Valley, with *C. t. townsendii* on the west side (Hall 1981). This species occurs from near sea level to well above 3,160 meters above sea level (Pearson et al. 1952, Nagorsen and Brigham 1993).

Occurrences within the ECCC HCP/NCCP Inventory Area

Townsend's big-eared bat is found throughout California, but specific details on its distribution within the central Coast Ranges are not well known. Records of this species include sites in the coastal lowlands and agricultural areas of Marin, Napa, Alameda, and San Mateo Counties and nearby hills (Pierson 1988). However, there are no published records of Townsend's big eared bat within Contra Costa County. Because of the scarcity of suitable habitat including mines and caves, it is unlikely that significant maternity roosts of this species occur in the county. However, future research may show that small numbers of individual bats roost in buildings, bridges, or other structures within the inventory area.

Biology

Habitat

Townsend's big-eared bats can be found in a variety of habitats throughout California, but they are most commonly associated with desert scrub, mixed conifer forest, and pinon-juniper or pine forest habitat. Within these communities, they are specifically associated with limestone caves, mines, lava tubes, and buildings (Dalquest, 1947, 1948; Graham 1966; Pearson et al. 1952; Kunz and Martin 1982; Pierson et al. 1991; Dobkin et al. 1995).

During hibernation, Townsend's big-eared bats typically prefer habitats with relatively cold (but above freezing) temperatures in quiet, undisturbed places. These areas are often in the more interior, thermally stable portions of caves and mines (Barbour and Davis 1969, Dalquest 1947, Humphrey and Kunz 1976, Pearson et al. 1952, Zeiner et al. 1990). Hibernating bats are often found in ceiling pockets (Pierson et al. 1991). In central California, solitary males and small clusters of females are also known to hibernate in buildings (Pearson et al. 1952, Kunz and Martin 1982). Females may roost in colder places than males during these periods (Pearson et al. 1952).

During spring and summer, females establish maternity colonies in the warm parts of caves, mines, and buildings (Dalquest 1948, Pearson et al. 1952, Twente 1955, Pierson et al. 1991). In California, some maternity roost may reach 30ºC (86ºF) (Pierson et al. 1991). Favored roost locations for females and young are often in a ceiling pocket or along the walls just inside the roost entrance (Pierson et al. 1991). Night roosts may include buildings or other structures, such as bridges (Pierson et al. 1996, Rainy and Pierson unpublished manuscript).

Foraging

Townsend's big eared bats feed primarily on small moths, but also take other insects, including flies, lacewings, dung beetles, and sawflies (Whitaker et al. 1977; Kunz and Martin 1982;). Radio-tracking studies in northern California have found Townsend's big-eared bats foraging within forested habitats and along heavily vegetated stream corridors, avoiding open, grazed pasture land (Pierson and Fellers 1998, Pierson et al. 1999). Individuals may travel up to 13 kilometers from their day roost (Pierson et al. 1999).

Reproduction

Female Townsend's big-eared bats arrive at maternity roost sites in early spring and give birth to a single offspring in late spring or early summer after an approximately 3-month gestation period (Pearson et al. 1952). In California, young are born over a 3- to 5-week period beginning in late May. Maternity colonies disperse in fall, and mating occurs in fall and winter. The peak of copulations occurs from November through February, although some females apparently mate before arriving at hibernacula (Kunz and Martin 1982). Females are sexually mature and mate in their first autumn. However, as in most bats, females store sperm, and ovulation does not occur until early spring (Pearson et al. 1952). Ovulation may occur either before or after females leave hibernation. Townsend's big-eared bats are large at birth, weighing approximately 25% of the mother's postpartum mass (Kunz and Martin 1982). Young grow rapidly, reaching adult size in approximately 1 month, and capable of flight in 2.5 to 3 weeks. They are fully weaned by 6 weeks (Pearson et al. 1952).

Demography

Band recoveries show longevity records of up to 16 years, 5 months (Paradiso and Greenhall 1967) and 21 years, 2 months (Perkins 1994). Pearson et al. (1952) estimated the annual survivorship for Townsend's big-eared bats was about 50% for young and 80% for adults.

Behavior

Townsend's big-eared bat is a relatively sedentary species for which no long-distance migrations have been documented (Pearson et al. 1952, Barbour and Davis 1969, Humphrey and Kunz 1976). The longest seasonal movements recorded for this species are 32.2 kilometers in California (Pearson et al. 1952) and 39.7 kilometers in Kansas (Humphrey and Kunz 1976).

Townsend's big-eared bats hibernate in mixed-sex aggregations of 100 to 500 individuals. They periodically arouse during winter and move to alternate roosts. Individuals actively forage and drink throughout winter (Brown et al. 1994). Hibernation is prolonged in colder areas and intermittent where climate is predominately not freezing (Kunz and Martin 1982).

Ecological Relationships

Townsend's big-eared bat is a lepidopteran specialist, with a diet consisting of more than 90% moths (Pierson et al. 1999). Night roosts of this species often include other bat species, including pallid bat (*Antrozous pallidus*), big brown bat (*Eptesicus fuscus*), California myotis (*Myotis californicus*), small-footed myotis (*M. ciliolabrum*), long-eared myotis (*M. evotis*), little brown bat (*M. lucifugus*), fringed bat (*M. thysanodes*), long-legged bat (*M. volans*), and Yuma myotis (*M. yumanensis*).

Threats

Townsend's big-eared bats are highly sensitive to roost disturbance. Activities that can result in significant disturbance or loss of habitat include mine reclamation, renewed mining, water impoundments, recreational caving, loss of building roosts, and bridge replacement (Kunz and Martin 1982, Pierson et al. 1999). Pesticide contamination may also threaten this species in agricultural areas (Geluso et al. 1976).

Conservation and Management

Townsend's big-eared bat has been classified as a High Priority species by the Western Bat Working Group for all populations throughout its range. This classification indicates that this species is imperiled or is at high risk of imperilment. In 1994, a Townsend's big-eared bat conservation strategy was initiated as part of the Idaho Conservation Effort. This strategy was prepared by a team of experts from 8 participating states and resulted in the publication of the Species Conservation Assessment and Conservation Strategy for the Townsend's big-eared bat (Pierson et al. 1999). The species conservation assessment summarizes the life history and habitat requirements, historical and current distribution and abundance of this species throughout its range, its current status, and threats to the species in each state. The conservation strategy is a plan that, if successful, will remove or minimize identified threats and promote restoration or recovery of the species.

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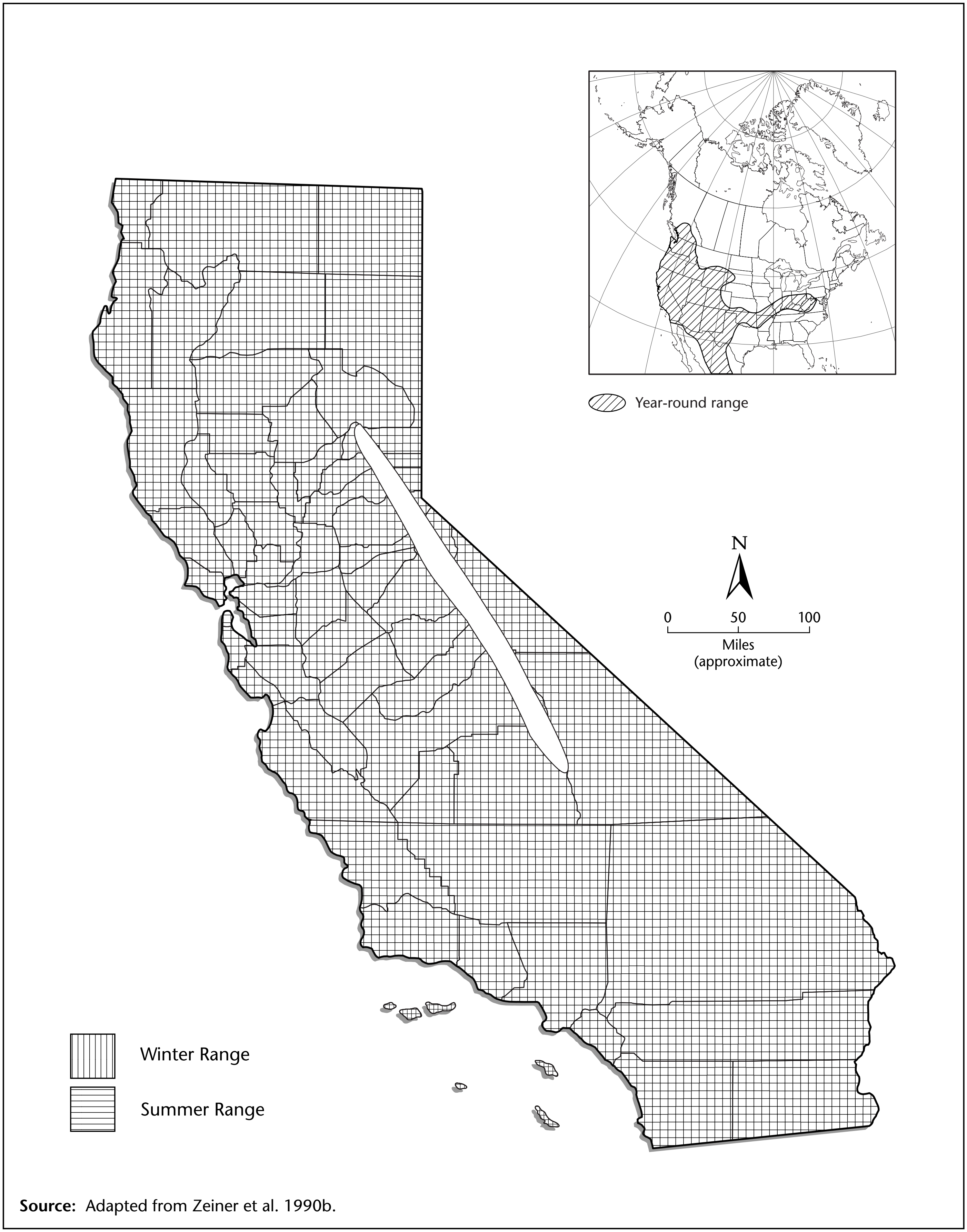

Source: Adapted from Zeiner et al. 1990b.

Jones & Stokes

Townsend's Big-Eared Bat *(Corynorhinus townsendii townsendii)* Distribution

San Joaquin Kit Fox (*Vulpes macrotus mutica*)

Status

State: Threatened
Federal: Endangered

Population Trend

Global: Declining
State: As above
Within Inventory Area: Unknown

Data Characterization

The location database for the San Joaquin kit fox (*Vulpes macrotus mutica)* within its known range in California includes 22 data records from 1975 to 1999. Of these records, none of the 7 documented within the past 10 years were of sufficient precision to be accurately located within the survey areas. Three of these 7 records are located within the ECCC HCP/NCCP inventory area. These records represent sighting within non-native grassland, grazed, and agricultural habitat. This database includes records of individual sightings and locations of occupied, vacant, and natal dens.

A moderate amount of literature is available for the San Joaquin kit fox because of its endangered status. Long-term studies have been conducted on the ecology and population dynamics of this species in core population centers at the Elk Hills and Buena Vista Naval Petroleum Reserves in Kern County and on the Carrizo Plain Natural Area in San Luis Obispo County. Numerous surveys have been conducted in the northern portion of the range, including Contra Costa County. Quantitative data are available on population size, reproductive capacity, mortality, dispersal, home-range movement patterns, and habitat characteristics and requirements. A number of models have been developed to describe the species' population dynamics. A recovery plan for the San Joaquin kit fox has been published.

Range

The San Joaquin kit fox is found only in the Central Valley area of California. Kit foxes currently inhabit suitable habitat in the San Joaquin valley and in surrounding foothills of the Coast Ranges, Sierra Nevada, and Tehachapi Mountains, from southern Kern County north to Contra Costa; Alameda and San Joaquin counties on the west; and near La Grange, Stanislaus County on the east.

Occurrences within the ECCC HCP/NCCP Inventory Area

Fifty-three occurrences of San Joaquin kit foxes have been documented within the inventory area since 1967 (Duke et al. 1997). These records were located from the Black Diamond Mines area and Lone Tree Valley in the north to Round Valley, Los Vaqueros Reservoir, and Brushy Creek in the south (Duke et al. 1997). Fifteen of these records were documented since 1986. The greatest density of occurrences is located in the southern portion of the inventory area.

Biology

Habitat

San Joaquin kit foxes occur in a variety of habitats, including grasslands, scrublands, vernal pool areas, alkali meadows and playas, and an agricultural matrix of row crops, irrigated pastures, orchards, vineyards, and grazed annual grasslands (Williams et al. 1998). They prefer habitats with loose-textured soils (Grinnell et al. 1937, Hall 1946, Egoscue 1962) that are suitable for digging, but they occur on virtually every soil type. Dens are generally located in open areas with grass or grass and scattered brush, and seldom occur in areas with thick brush. Preferred sites are relatively flat, well-drained terrain (Williams et al. 1998, Roderick and Mathews 1999). They are seldom found in areas with shallow soils due to high water tables (McCue et al. 1981) or impenetrable bedrock or hardpan layers (O'Farrell and Gilbertson 1979, O'Farrell et al. 1980). However, kit foxes may occupy soils with a high clay content where they can modify burrow dug by other animals, such as ground squirrels (*Spermophilus beecheyi*) (Orloff et al. 1986).

In the northern part of its range (including San Joaquin, Alameda and Contra Costa Counties) where most habitat on the valley floor has been eliminated, kit foxes now occur primarily in foothill grasslands (Swick 1973, Hall 1983, Williams et al. 1998), valley oak savanna and alkali grasslands Bell 1994). Less frequently they occur adjacent to and forage in tilled and fallow fields and irrigated row crops (Bell 1994). These foxes will den within small parcels of native habitat that is surrounded by intensively maintained agricultural lands (Knapp 1978) and adjacent to dryland farms (Jensen 1972, Orloff et al. 1986, Williams et al. 1998).

Foraging Requirements

The diet of kit foxes varies, with season and geographic locality based on local availability of potential prey. In the northern portion of their range, kit foxes most commonly prey on California ground squirrels, cottontails (*Sylvilagus audubonii*), black-tail jackrabbits (*Lepus californicus*), pocket mice (*Perognathus* spp.), and kangaroo rats (*Dipodomys* spp.) (Hall 1983, Orloff et al. 1986, Williams et al. 1998). Secondary prey taken opportunistically may include ground-nesting birds, reptiles, and insects (Laughrin 1970).

Reproduction

Kit foxes can, but do not necessarily, breed their first year. Sometime between February and late March, 2 to 6 pups are born per litter (Egoscue 1956, Zoellick et al. 1987a, Cypher et al. 2000). The annual reproductive success for adults can range between 20 and 100% (mean: 61%;) and 0 and 100% for juveniles (mean: 18%) (Cypher et al. 2000). Population growth rates generally vary positively with reproductive success and kit fox density is often positively related to both current and the previous year's prey availability (Cypher et al. 2000). Prey abundance is generally strongly related to the previous year's effective (October to May) precipitation.

Longevity

Kit foxes in the wild can live as long as 8 years, but such longevity is rare (Williams et al. 1998). Annual survival rates of juvenile foxes generally range between 21 and 41% (Berry et al. 1987, Ralls and White 1995). In captivity, kit foxes can live up to 10 years (McGrew 1979). The annual natural mortality rate of adults is approximately 50% (Berry et al. 1987, Ralls and White 1995), but is closer to 70% for juveniles (Berry et al. 1987). Coyotes (*Canis latrans*)and other predators (red foxes [*Vulpes vulpes*], domestic dogs, bobcats [*Lynx rufus*] and large raptors) are the primary sources of mortality for adult and juvenile foxes (Hall 1983, Betty et al. 1987b, Ralls and White 1995, Warrick et al. 1999, White et al. 2000, Cypher et al. 2000), and vehicles are usually the secondary cause (Cypher et al. 2000).

Population Dynamics

In a long-term study of kit fox population dynamics at the Naval Petroleum Reserves in California, Cypher et al. (2000), showed that population growth rates vary positively with reproductive success, and population density is positively related to both the current and the previous year's prey availability. Prey abundance was strongly related to the previous year's effective precipitation (October to May).

White and Garrott (1999) note that 2 density-dependent mechanisms appear to regulate kit fox population patterns. The first, the rate of juvenile recruitment, is inversely related to the density of adult foxes because higher proportions of juveniles are generally killed by coyotes at higher fox densities. The mortality rates of adult foxes are apparently independent of population density. The second is that populations of kit foxes are bounded by their territorial spacing behavior, which limits recruitment at higher densities. These mechanisms, therefore, may act together to curtail population growth at high densities, whereas decreased juvenile mortality by coyotes can act independently to increase population growth at low densities.

Density-independent factors, particularly unpredictable fluctuations in precipitation that contribute to high-frequency, high-amplitude fluctuations in the abundance of kit fox prey, can also result in variations in reproductive rates that cause population crashes or eruptions (White and Garrott 1999). Unpredictable short-term fluctuations in precipitation, and in turn, prey abundance could therefore generate longer-term, aperiodic fluctuations in the density of foxes that are independent of special or persistent causes such as predation or disease.

Dispersal

The pups emerge above ground at approximately 1 month of age and some disperse after 4 to 5 months, usually between July and September. In a study of 209 dispersing juveniles, Koopman et al. (2000) found that 33% dispersed from their natal territory, significantly more males (49%) than females (24%). The percentage of male dispersal was weakly related to mean annual litter size, and the percentage of female dispersal was weakly and inversely related to annual small-mammal prey abundance. Most of the dispersing juveniles (65%) died within 10 days of leaving their natal range. However, survival tended to be higher for dispersing males than for males that remained within their natal area. There was no difference in survival for dispersing and philopatric females. Non-dispersing offspring of both sexes may remain with their parents through the following year and help raise the next litter (White and Ralls 1993…), but this behavior is not always observed (Koopman et al. 2000).

Behavior

Den Use

San Joaquin kit foxes use numerous dens throughout the year. They are used for temperature regulation, shelter from inclement weather, reproduction, and escape from predators. Hall (1983) documented a family of 7 kit foxes that used 43 dens in 1 year, while 1 other individual used 70 dens (K Ralls, pers. comm. in Williams et al. 1998). Koopman et al. (1998) found that individual foxes within the Naval Petroleum Reserves use an average of 11.8 different dens each year, and den use does not differentiate between sexes. The number of dens used varied among seasons, with more dens used during the dispersal season than during the breeding or pup-rearing seasons. On average, kit foxes used an individual den 10% of the time throughout the year, but favored dens were used 32% of the time. Approximately 50% of the dens used by a kit fox in a 1-year period had not been used by that fox during the previous year. Male and female pups up to 18 months of age denned equally with either adult parent. They denned with siblings for up to 21 months of age. Radio telemetry studies of kit fox movement on the Carrizo Plain Natural Area (White and Ralls 1993) indicate that foxes use individual dens for an average of 3.5 days before moving to a different den. Den changes are believed to be primarily in response to a need to avoid coyotes, although local depletion of prey and increases in external parasites in the dens may also influence this behavior (Egoscue 1956 *in* Williams et al. 1998).

Movement

Kit foxes may range up to 20 miles at night (Girard 2001) during the breeding season and somewhat less (6 miles) during the pup-rearing season. Home ranges vary from less than 1 square mile up to approximately 12 square miles (Knapp 1978, Spiegel and Bradbury 1992, White and Ralls 1993). The home ranges of pairs or family groups of kit foxes generally do not overlap (White and Ralls 1993). This behavior may be an adaptation to periodic drought-induced scarcity in prey abundance.

Social Structure

Genetic and field studies of kit foxes on the Carrizo Plains Natural Area (Ralls et al. 2001) showed that foxes living in adjacent home ranges tended to be more closely related than foxes from more distant home ranges. This pattern emerged largely because females on adjacent home ranges were often closely related. Foxes that shared the same den were usually members of the same social group, but occasionally foxes from different social groups shared dens, possibly during pair formation. San Joaquin kit foxes can maintain enduring social relationships with adult offspring or siblings that have dispersed to new home ranges and found a mate.

Ecological Relationships

San Joaquin kit foxes prey upon a variety of small mammals, ground-nesting birds, and insects. They are in turn subject to predation or killing by such species as coyote, non-native red foxes, domestic dog, eagles, and large hawks (Hall 1983, Berry et al. 1987, Ralls and White 1995). White et al. (2000) determined that coyotes were responsible for 59% of kit fox deaths during a 4-year telemetry study at Camp Roberts in southern Monterey County.

Threats

Loss, fragmentation, and degradation of habitat by agricultural, urban, and industrial development continues to decrease the remaining habitat and carrying capacity of San Joaquin kit foxes throughout its range. Livestock grazing is not thought to be detrimental to kit foxes (Morrell 1975, Orloff et al. 1986), but it may affect the number of prey species available, depending on the intensity of grazing (Williams et al. 1998). In some areas, livestock grazing may benefit kit foxes by reducing shrub cover and maintaining grassland habitat.

Continued fragmentation of habitat is a serious threat to this species. Increasing isolation of populations and social groups through habitat degradation and barriers to movement, such as aqueducts and busy highways, can limit dispersal to and habitation of existing and former lands. This isolation also favors inbreeding depression in populations, as well as making the smaller populations susceptible to extinction from stochastic environmental events such as droughts, flooding, fire, and periodic declines in prey abundance.

The use of pesticides to control rodents and other pests also threatens kit fox in some areas, either directly through poisoning or indirectly through reduction of prey abundance. Invasion of fragmented, occupied kit fox habitat by coyotes, red foxes, and feral dogs can contribute to increased mortality of kit foxes.

Conservation and Management

The San Joaquin kit fox is listed as both state and federally endangered. A recovery plan for this species was completed in 1983 that outlines objectives to halt the decline of the species and increase population sizes above the 1981 level (Williams et al. 1998). Subsequent conservation actions have included acquisition of important habitat by the U.S. Bureau of Land Management (BLM), California Department of Fish and Game (CDFG), California Energy Commission, Bureau of Reclamation, U.S. Fish and Wildlife Service (USFWS), and Nature Conservancy. Substantial long-term research has been conducted on populations in the Naval Petroleum Reserves and in the Carrizo Natural Area in southern California. These studies have provided important information on kit fox habitat requirements, behavior, demographics, and threats.

In 1998, a recovery plan for upland species of the San Joaquin Valley was completed (Williams et al. 1998), which included a revised recovery strategy for the San Joaquin kit fox. The goal of this recovery plan is to maintain a viable metapopulation of kit foxes on private and public lands throughout it's the plan's geographic range. This will include preservation of existing core and satellite populations. Areas where core populations are found include the Carrizo Plain Natural Area in San Luis Obispo County; the natural lands of western Kern County, including the Naval Petroleum Reserves, the Lokern Natural Area, and adjacent natural lands inhabited by kit foxes; and the Ciervo-Panoche Natural Area of western Fresno and eastern San Benito Counties. Camp Roberts and Fort Hunter Liggett also provide important habitat for kit foxes in the Salinas and Pajaro river watersheds. Additional lands in the San Joaquin Valley that have kit foxes or the potential to have them include refuges and other lands managed by the CDFG, California Department of Water Resources, Center for Natural Lands Management, Lemoore Naval Air Station, Bureau of Reclamation, and USFWS, as well as various private lands in these areas. While kit foxes have been documented in numerous locations in East Contra Costa County, no conservation areas were identified for this species in the 1998 recovery plan. However, the recovery plan identifies the protection of existing kit fox habitat in the northern portion of its range and protection of existing connections between habitat in Contra Costa County and habitat farther south as primary recovery actions.

Status Assessment

San Joaquin kit foxes are known to occur within the ECCC HCP/NCCP inventory area, with greater numbers occurring in the southern portion of the area. However, compared with populations in southern California, little is known about the ecology and habitat needs of kit foxes in the northern part of their range. Researchers have consistently indicated that the behavioral ecology of kit foxes in this region is poorly known and may be different from the ecology of foxes in the southern part of their range (Laughrin 1970, Swick 1973, Morrell 1975, Orloff et al. 1986, Sproul and Flett 1993, Bell 1994). The northern populations of kit foxes appear to use different prey (ground squirrels instead of kangaroo rats), and their denning habitat appears different (Orloff et al. 1986). In addition, habitat (ground cover, dominant vegetation, land use practices, rainfall, and in some cases relief) is substantially different in the north than in the south, where kit foxes are more abundant and well studied. Because of these differences, some geographic differences may exist in the demographic characteristics of these populations. However, the threats of habitat loss; degradation and fragmentation; predation by coyotes, red foxes, feral dogs, and other predators; and vehicular mortality are likely to be comparable in both regions of their range.

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Tricolored Blackbird (*Agelaius tricolor*)

Status

State: Bird Species of Special Concern, Priority 1
Federal: Species of Concern

Population Trend

Global: Declining
State: Declining (Beedy and Hamilton 1997, 1999)
Within Inventory Area: Possibly declining (Beedy and Hamilton 1997).

Data Characterization

Statewide surveys were conducted for tricolored blackbirds (*Agelaius tricolor)* in California during 1994 and 1997 (Beedy and Hamilton 1999). Additional surveys include data on local distribution and population trends (Neff 1937, DeHaven et al. 1975a,). Because this species is nomadic with erratic movement behavior, local occurrence data provides only limited information on long-term small area use patterns. This species forages and breeds in specific locations the inventory area with freshwater marshes dominated by cattails of bulrushes, or in areas with suitable willow, blackberry, thistle, or nettle habitat.

A moderate amount of literature is available for the tricolored blackbird because it is a highly visible colonial bird species commonly associated with wetland habitat. Beedy and Hamilton (1999) provide a comprehensive review of information available on general natural history, behavior, distribution and population changes, known demographics and population regulation, and conservation and management. No rangewide management plan has been developed.

Range

The current breeding range of the tricolored blackbird includes California and scattered local occurrences in Oregon, Washington, Nevada, and Baja California (Beedy and Hamilton 1999). Since 1980, active breeding colonies have been observed in 46 California counties, including Contra Costa County. It breeds locally west of the Cascade Range, Sierra Nevada, and southeastern deserts from Humboldt and Shasta Counties south to San Diego County. In the Central Valley, breeding colonies have been located east into the foothills of the Sierra Nevada. The species also breeds in marshes of Klamath Basin in Siskiyou and Modoc Counties and Honey Lake Basin in Lassen County.

Occurrences within the ECCC HCP/NCCP Inventory Area

The tricolored blackbird is a sporadic resident within the inventory area. California Natural Diversity Database records document 2 breeding colony occurrences along the northern border of the Los Vaqueros watershed. The Contra Costa County Breeding Bird Atlas shows additional breeding locations east and north of these areas (http://www.flyingemu.com/ccosta).

Biology

Habitat

The primary historic breeding habitat of tricolored blackbirds in the Central Valley was freshwater marshes dominated by cattails or bulrushes, with some colonies occurring in willows, blackberries, thistles, and nettles (Neff 1937). More recent colonies have been observed in a diversity of upland and agricultural areas, including silage and grainfields (Collier 1968, Cook 1996), arundo, safflower, mustard, stinging nettles, riparian scrublands and forests, desert olive, and spiny field plants such as wheat, barley and thistles (Orians 1961a; DeHaven et al 1975a; Beedy et al. 1991; Hamilton et al. 1995; Beedy and Hamilton 1999).

Some small breeding colonies at public and private lakes, reservoirs, and parks are surrounded by shopping centers, subdivisions, and other urban development. Adults from these colonies generally forage in nearby undeveloped upland areas. Beedy and Hamilton (1999) predict that these small, urban wetlands and upland foraging habitats may continue to accommodate tricolored blackbirds in the future unless they are eliminated entirely by development. High-quality foraging areas include irrigated pastures, lightly grazed rangelands, dry seasonal pools, mowed alfalfa fields feedlots, and dairies (Beedy and Hamilton 1999). Lower quality foraging habitats include cultivated row crops, orchards, vineyards, and heavily grazed rangelands.

Habitat requirements for breeding colony sites include accessible water, protected nesting sites (flooded or surrounded by thorny or spiny vegetation), and suitable foraging area within a few kilometers of the nesting colony (Beedy and Hamilton 1997). The nests are attached to upright plant stems from a few centimeters to about 1.5 meters above water or the ground. Some nests have been found up 3 meters and higher in willow canopies, in valley oak saplings and in canopies of small ashes (Beedy and Hamilton 1999). Nests are usually located under dense overhanging vegetation, which provides protection from direct sunlight and rain. Breeding males defend only the immediate nesting area. Male territories range in size from 1.8 to 3.25 square meters.

Foraging Requirements

Tricolored blackbirds are opportunistic foragers and will consume any locally abundant insect resource including grasshoppers, beetles, weevils, caddis fly larvae, and butterfly larvae, as well as grains, snails, and small clams (Orians 1961a, Collier 1968, Payne 1969, Crase and DeHaven 1977, Martin et al. 1951, Skorupa et al. 1980). Concentrated agricultural food resources available at dairies and feedlots are also consumed when available (Skorupa et al. 1980).

Reproduction

Male tricolored blackbirds begin territorial behavior and courtship as early as late February (Beedy and Hamilton 1999). Nesting is generally synchronous, with the first eggs being laid within 1 week (Orians 1961a), even in colonies of up to 100,000 nests. The first eggs can be laid at early colonies by mid-March or early April. Clutches typically contain 3 to 4 eggs (range 1 to 5) (Payne 1969). Eggs take approximately 11 days to hatch; all eggs hatch within 24 hours. Fledging takes 12 to 14 days. Synchronized second broods within a colony may be initiated as little as 30 days after the first brood. Individual pairs may nest 2 or more times per year.

Demography

Banding studies show the lifespan of tricolored blackbirds to be at least 12 years (Neff 1942, DehHaven and Neff 1973, Kennard 1975). No annual survivorship studies have been conducted.

Behavior

During the breeding season, tricolored blackbirds exhibit itinerant breeding, commonly moving to different breeding sites each season. In the north Central Valley and northeast California, individuals move after first nesting (Beedy and Hamilton 1997). Banding studies indicate that significant movement into the Sacramento Valley occurs during the postbreeding 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 northern San Joaquin Valley (Neff 1937, Payne 1969, DeHaven et al. 1975b). Large flocks also congregate in pasturelands in southern Solalno County and near dairies on the Point Reyes Peninsula by late October (Beedy and Hamilton 1999). Other birds winter in the central and southern San Joaquin Valley. Concentrations of over 15,000 wintering tricolors may gather at 1 location and disperse up to 32 kilometers to forage (Neff 1937, Beedy and Hamilton 1999). Individual birds may leave winter roost site 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 Sacramento County and throughout San Joaquin Valley (DeHaven et al. 1975b).

Tricolored blackbirds are highly colonial and sometimes polygynous, with 1 to 4 females pairing with 1 male (Payne 1969). Historic colonies of over 200,000 pairs have been documented occupying a 24 hectares of cattail marsh (Neff 1937). This social cohesion is retained during the nonbreeding season with birds forming large foraging and roosting flocks. These flocks may be all tricolors, or mixed flocks with red-winged blackbirds, Brewer's blackbirds, brown-headed cowbirds, and European Starlings (Beedy and Hamilton 1999).

Ecological Relationships

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

Threats

Habitat loss, including development and conversion to agriculture of marshlands in the Central Valley, poses the greatest threat to tricolored blackbird populations. Many historical breeding and foraging habitats have been eliminated, and currently there are limited locations where large colonies can exist (Beedy et al. 1991; Beedy and Hamilton 1997, 1999). Pesticides and other toxic contaminants can also have a negative effect on existing tricolor populations. Beedy and Hayworth (1992) documented an almost complete failure of a large colony (approximately 47,000 adults) at Kesterson Reservoir in Merced County, an area contaminated by selenium deposited from agricultural drainage water. Exposure to spraying of mosquito abatement oil has also eliminated breeding in other populations (Hosea 1986).

Conservation and Management

The tricolored blackbird is a bird species of special concern in California (California Department of Fish and Game and Point Reyes Bird Observatory 2001). Management goals that have been proposed include maintaining a viable self-sustaining population throughout the species' current geographic range, avoiding losses of colonies and their associated habitats, increasing breeding populations on suitable public and private lands managed for this species, and enhancing public awareness and support for protection of habitat and active colonies. A California Department of Fish and Game and U.S. Fish and Wildlife Service program for purchasing portions of crops to preserve several large colonies of tricolors in Kings, Fresno, and Tulare Counties was implemented in 1993 and 1994 with significant conservation results. These actions and participation by landowners in delaying harvest to protect active nesting colonies resulted in an addition of an estimated 37,000 and 44,000 first-year added to the 1994 and 1995 breeding seasons (Beedy and Hamilton 1999). Similar conservation measures could be used in the inventory area to enhance populations.

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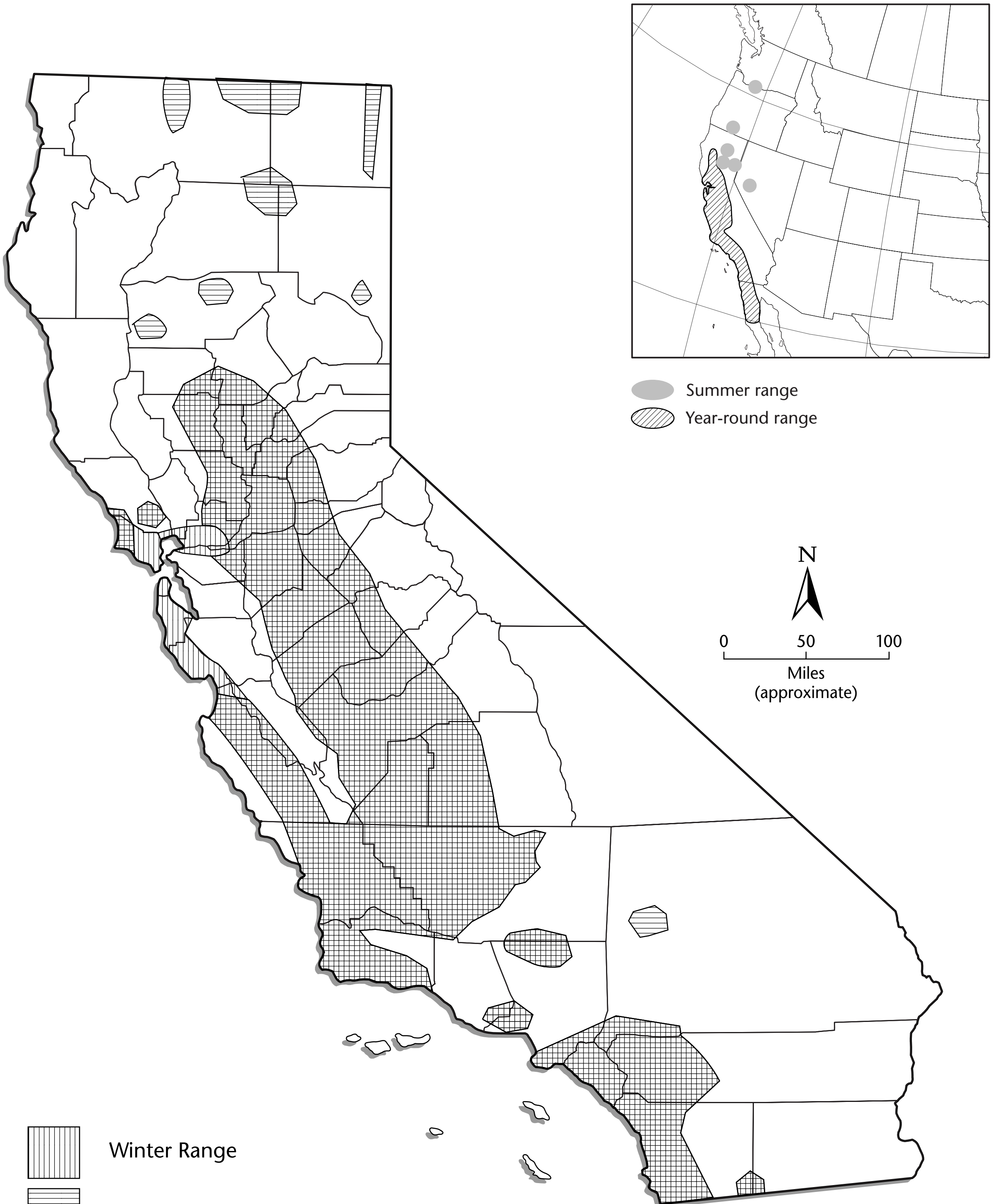

Source: Adapted from Zeiner et al. 1990a, and Sibley 2000.

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Jones & Stokes

Tricolored Blackbird *(Agelaius tricolor)* Distribution

Golden Eagle (*Aquila chrysaetos*)

Status

State: Fully Protected species
Federal: Protected under the Bald Eagle and Golden Eagle Protection Act

Population Trend

Global: Apparently stable in most areas of western U.S.; unknown elsewhere
State: Declining in southern California; common and presumably stable elsewhere in California
Within Inventory Area: Unknown

Data Characterization

Extensive long-term studies have been conducted on the distribution, demographics, and general biology of Golden Eagles (*Aquila chrysaetos)* in the vicinity of the ECCC HCP/NCCP inventory area as part of investigations on the impact of wind turbine operation on this species (see Hunt et al. 1998). These studies provide detailed information on the distribution and habitat-use patterns of resident and non-resident Golden Eagles, population structure, reproductive rates, survival rates, and population equilibrium dynamics.

A moderate amount of additional literature is available for the Golden Eagle outside the inventory area because it is a highly visible, fully protected bird of prey and top avian predator within its range. Most of the literature pertains to general natural history, behavior, distribution, and population changes in the past 30 to 40 years. Some information is available on demographics and population trends. Limited species-specific management information is available.

Range

The Golden Eagle is Holarctic in distribution. In North America, it breeds from northern and western Alaska east to the Northwest Territories, Canada, and south to southern Alaska, Baja California, the highlands of northern Mexico, west-central Texas, western portions of Oklahoma, Nebraska, and the Dakotas, and irregularly in eastern North America. The Golden Eagle winters in North America from south-central Alaska and the southern portions of the Canadian provinces south throughout the western breeding range and more rarely eastward (Johnsgard 1990).

Occurrences within the ECCC HCP/NCCP Inventory Area

The Golden Eagle is a resident breeder and migrant within the ECCC HCP/NCCP inventory area. The reproductive status of numerous nesting pairs has been monitored regularly within this general area (Hunt et al. 1998). The

Contra Costa County Breeding Bird Atlas (http://www.flyingemu.com/ccosta/) shows additional breeding locations east and north of these areas.

Biology

Habitat

Golden eagles use nearly all terrestrial habitats of the western states except densely forested areas. In the interior central Coast Ranges of California, Golden Eagles favor open grasslands and oak savanna, with lesser numbers in oak woodland and open shrublands (Hunt et al. 1998). Secluded cliffs with overhanging ledges and large trees are used for nesting and cover. Nest trees include several species of oak (*Quercus* spp.), foothill pine (*Pinus sabianiana* and *P. coulteri*), California bay laurel (*Umbellularia californica*), eucalyptus (*Eucalyptus* spp.), and western sycamore (*Plantanus racemosa*) (Hunt et al. 1998). Preferred territory sites include those that have a favorable nest site, a dependable food supply (medium to large mammals and birds), and broad expanses of open country for foraging. Hilly or mountainous country where takeoff and soaring are supported by updrafts is generally preferred to flat habitats (Johnsgard 1990). Deeply cut canyons rising to open mountain slopes and crags are ideal habitat (Beebe 1974).

Breeding densities are directly related to territorial spacing and foraging requirements for the species. Territory size has been estimated to average 124 square kilometers (sq km) in northern California (Smith and Murphy 1973), but can vary largely with habitat conditions. Hunt et al. (1998) report an 820-sq-km area near Livermore supported at least 44 pairs of Golden Eagles in 1997, with a density of 1 pair per 19 sq km. This density is among the highest reported for the species.

Foraging

Golden eagles prey mostly on rabbits, hares, and rodents, but also take other mammals, birds, reptiles, and some carrion (Olendorff 1976, Hunt et al. 1998). California ground squirrels (*Spermophilus beecheyii*) and black-tailed jackrabbits (*Lepus californicus*) are the 2 most important prey species for the Golden Eagle within the inventory area (Hunt et al. 1998). Eagles typically hunt by using favorite perches located near areas that have regular updrafts to facilitate soaring to heights from which they can scan their hunting areas (Johnsgard 1990).

Reproduction

Nest building can occur almost any time of year (Brown 1976). Golden eagles prefer to locate their nests on cliffs or trees near forest edges or in small stands near open fields (Bruce et al. 1982, Hunt et al. 1995, 1998). Mating occurs from late January through August, with peak activity in March through July. Eggs are laid from early February to mid-May. Clutch size varies from 1 to 4 eggs, but 2 is the most common size (Brown 1976, Johnsgard 1990, Hunt et al. 1995). Incubation lasts 43–45 days (Beebe 1974), and the fledging period is about 72–84 days (Johnsgard 1990). The young usually remain dependent on their parents for as long as 11 weeks afterward. Breeding success tends to be variable depending upon local prey abundance. In a 15-year study of Golden Eagles in Oregon, Thompson et al. (1982) calculated a mean of 1.08 young fledged per breeding territory, 1.7 young fledged per successful nest, and 51% overall nesting success. Beecham and Kochert (1975) showed a similar average of 1.1 young fledged per nesting attempt, 1.8 young fledged per successful nest, and 65% overall nesting success in Idaho. More recently, Hunt et al. (1998) reported natality estimates of 0.64 and 0.58 young per pair for 57 and 59 pairs, respectively, in 1996 and 1997, within a 190-sq km wind resource area, a portion of which is within the ECCC HCP/NCCP inventory area. Brood sizes for this study varied from 1.44 to 1.62 fledglings per nest.

Demography

There are no published reports of the longevity of Golden Eagles in the wild. Captive Golden Eagles have lived to 48 years, but it is not likely that they live that long in the wild (Brown and Amadon 1968).

Behavior

Movement and Dispersal

Breeding Golden Eagles in the central Coast Ranges of California are mostly resident; juveniles may remain in the vicinity of their natal area until evicted by the parents. Floater non-breeding birds (adults without breeding territories) commonly move about regionally until they find a suitable vacant territory or are able to evict a territorial owner (Brown 1969, Hunt et al. 1995, 1998). Some migrants may temporarily move into areas used by resident birds during the winter.

Social

Healthy Golden Eagle populations include 4 population segments: breeders, juveniles, subadults, and floaters (Hunt et al. 1998). Breeders are individuals 4 years old or older that defend territories containing a potentially successful nest. Breeding pairs partition the landscape into a mosaic of territories that define the population density and size. Territorial boundaries tend to remain fairly stable from year to year (Marzluff et al. 1997). The size and density of territories is a function of either food or nest-site availability. During years of low prey availability, eagles may forgo breeding but still occupy and maintain their territories.

Juveniles are eagles less than 1 year old; subadults are 1, 2, and 3 years of age. The existence of floaters is an indication that all habitat suitable for breeding is occupied by territorial pairs (Hunt et a1. 995, 1998). Floaters act to maintain the breeding segment of the population by replacing breeders that have died. However, if the number of floaters is large relative to the number of breeders, floater competition for nesting territories may reduce the reproductive rate (Hansen 1987).

Ecological Relationships

Golden eagles are the top avian predator in the grassland/savanna ecosystem of the central Coast Range in California. They may directly compete with ferruginous and other smaller hawks for small mammals, and with California condors (*Gymnogyps californianus*) for carrion. Territorial interactions with other Golden Eagles may result in some fatalities.

Threats

Existing threats to Golden Eagle survival in the central Coast Ranges of California include both foraging- and nesting-habitat loss; human disturbance of nesting birds; and direct fatalities from wind turbine strikes, electrocution, and poisoning. An analysis of the causes of fatalities of 61 Golden Eagles radio-tagged and recovered in the Diablo Range from January 1994 to December 1997 (Hunt et al. 1998) showed that 37% were killed by turbine strikes, 16% by electrocution, and 5% by lead poisoning (Hunt et al. 1998). The remaining birds were lost due to shootings (2%), car strikes (5%), botulism (2%), territorial fights with other eagles (5%), collision with fences (3%), fledging mishaps (10%), and other unknown factors (15%)

Conservation and Management

Golden eagle management and conservation generally includes habitat management, population enhancement, hazard management, controlling human activity in sensitive raptor areas, and education. Cattle ranching throughout the central Coastal Ranges can benefit and be beneficial to the Golden Eagle if grazing is maintained at moderate levels that stimulate growth of herbaceous foods used by primary prey species, including ground squirrels and rabbits (Hunt et al. 1995). In this area, ground squirrel populations are reported to reach their highest densities in areas of low grass height typical of grazed lands. Cattle ranching also provides eagles a source of carrion from dead cows, stillborn calves, and placentas.

Hazard management efforts that are being implemented to reduce wind turbine strikes include replacement of turbine models with fewer larger, but slower, ones that are less likely to strike soaring or hunting eagles.

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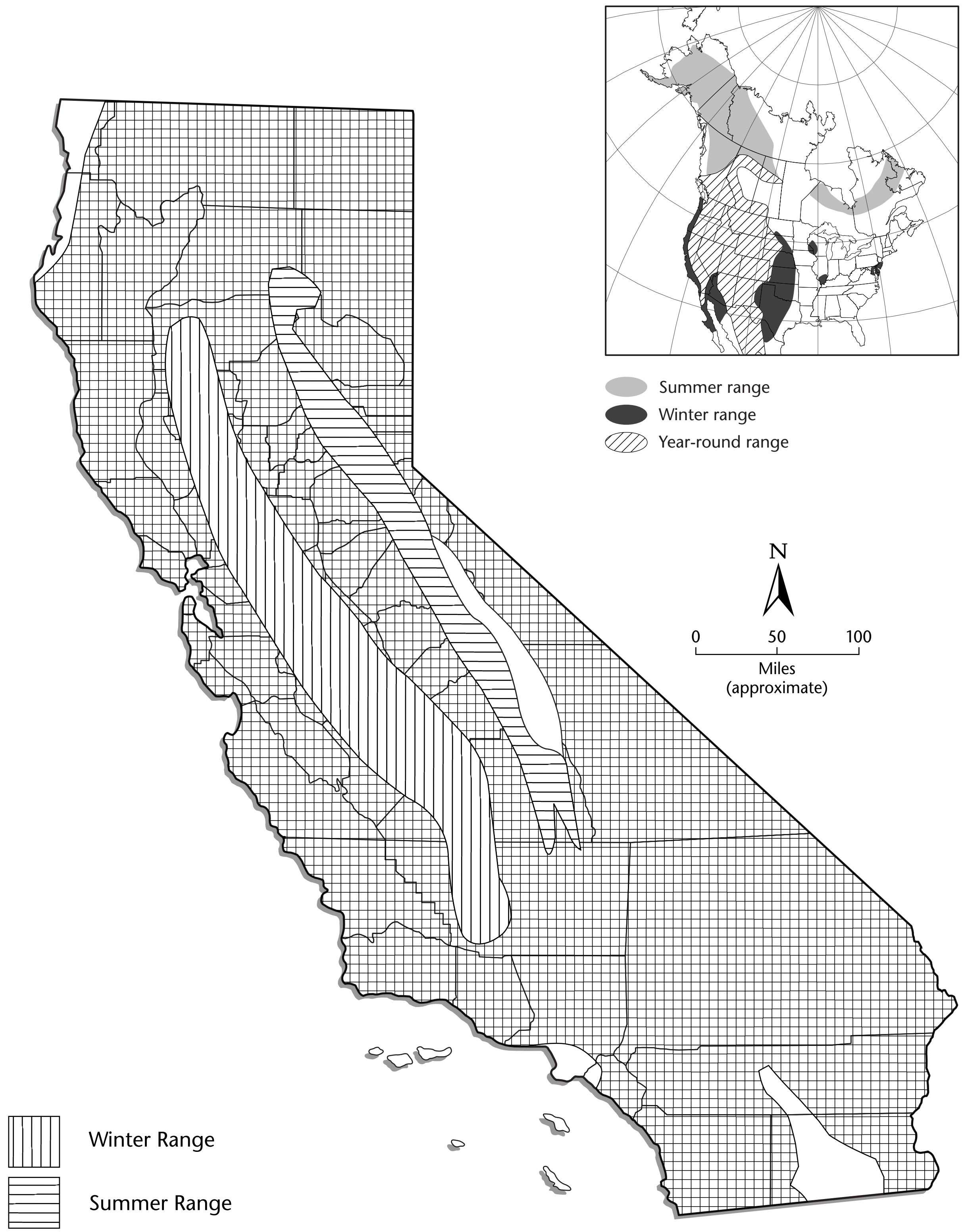

Source: Adapted from Zeiner et al. 1990a, and Silbey 2000.

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Jones & Stokes

Golden Eagle *(Aquila chrysaetos)* Distribution

Western Burrowing Owl (*Athene cunicularia hypugea*)

Status

State: Species of Special Concern
Federal: Species of Concern

Population Trend

Global: Declining
State: Declining
Within Inventory Area: Unknown

Data Characterization

The location database for the Western Burrowing Owl within the inventory area includes 17 data records dated from 1989 to 2000. Of these records, 13 were documented within the past 10 years; of these, 5 are of high precision and may be accurately located within the inventory area. Approximately 1 of these high precision records is located within developed areas. The remainder of the records occur within nonnative annual grassland habitats, or adjacent to roads or irrigation canals in agricultural fields.

A large amount of peer-reviewed literature is available for the Western Burrowing Owl. This species is declining throughout its range; therefore, most of the research studies emphasize nest site selection, passive relocation, use of artificial burrows, reproductive success, dispersal, and foraging behavior.

Range

The Western Burrowing Owl, the western race of the burrowing owl, is found throughout western North America, west of the Mississippi River and south into Mexico. Other burrowing owl races occur in arid, open habitats from the provinces of southern and southwestern Canada to southern Florida and South America (Haug et al. 1993).

In California, the Western Burrowing Owl is a year-round resident. It was once widespread, but its range has contracted significantly, particularly in coastal grasslands. Since the 1940s, numbers have declined in most areas of the state except the Imperial Valley (DeSante and Ruhlen 1995).

The Western Burrowing Owl is distributed over most of the Central Valley. Suitable foraging and breeding habitat for burrowing owl, such as grasslands, vernal pool grasslands, fallow agricultural fields, and open oak woodlands occur throughout most of the Central Valley and are represented in the inventory area. The potential to extend owl habitat use into suitable areas is limited by land management practices that reduce ground squirrel populations, thereby limiting the number of suitable owl nesting burrows.

Occurrences within the ECCC HCP/NCCP Inventory Area

The Western Burrowing Owl occurs in the southeast portion of the inventory area (Glover pers. comm.) and likely occurs in potential habitat throughout other portions of the inventory area. Potential habitat is defined as habitat that could support burrowing owls, based on a general classification of land cover types (e.g., grassland, vernal pool grassland, grassland pasture) developed for the HCP/NCCP. Because a comprehensive survey for the burrowing owl has not been conducted in the inventory area, neither the current population size nor the locations of all occurrences are known.

Biology

Habitat

Burrowing owls require habitat with 3 basic attributes: open, well-drained terrain; short, sparse vegetation; and underground burrows or burrow facsimiles. During the breeding season, they may also need enough permanent cover and taller vegetation within their foraging range to provide them with sufficient prey, such as small mammals (Wellicome 1994). Burrowing owls occupy grasslands, deserts, sagebrush scrub, agricultural areas (including pastures and untilled margins of cropland), earthen levees and berms, coastal uplands, and urban vacant lots, as well as the margins of airports, golf courses, and roads.

Burrowing owls select sites that support short vegetation, even bare soil, presumably because they can easily see over it. However, they will tolerate tall vegetation if it is sparse. Owls will perch on raised burrow mounds or other topographic relief, such as rocks, tall plants, fence posts, and debris piles, to attain good visibility (Haug et al. 1993).

The most important habitat consideration for the Western Burrowing Owl is the availability of underground burrows throughout their life cycle. Although the owls nest and roost in these burrows, they do not (contrary to their name) create them. Rather, the owls rely on other animals to dig their burrows. Throughout their range, they use burrows excavated by fossorial (i.e., digging) mammals or reptiles, including prairie dogs, ground squirrels, badgers, skunks, armadillos, woodchucks, foxes, coyotes, and gopher tortoises (Karalus and Eckert 1987). Where the number and availability of natural burrows is limited (e.g., where burrows have been destroyed or ground squirrels eradicated), owls will occupy drainage culverts, cavities under piles of rubble, discarded pipe, and other tunnel-like structures (Haug et al. 1993).

For Western Burrowing Owls, what constitutes an isolated habitat patch and the minimum size of a viable patch of habitat (i.e., habitat capable of sustaining a population over a long time period) are not well documented. These parameters are affected by habitat quality, the juxtaposition of the site relative to other suitable habitat, surrounding land uses, and prey availability. Burrowing owls have been observed in small (i.e., 1 acre) lots nearly surrounded by development, and owls will fly through urban areas to forage in nearby areas. However, the type and minimum extent of development that constitutes a movement barrier between occupied patches and nearby foraging areas are not known.

It is assumed that corridors between small habitats and other suitable areas would partly offset the insular effects of small or isolated habitats on owl populations by increasing foraging potential and facilitating dispersal or colonization. The size and dimensions of corridors that would be adequate to facilitate movements of burrowing owls between suitable habitats has not been studied. Also, these requirements probably vary with the distance between suitable habitats, surrounding land uses, and the type and quality of habitat within the corridor.

Breeding habitat requirements

Like other owls, Western Burrowing Owls breed once per year in an extended reproductive period, during which most adults mate monogamously. Both sexes reach sexual maturity at 1 year of age. Clutch sizes vary, and the number of eggs laid is proportionate to prey abundance (the more prey that is available, the more eggs owls tend to lay). Clutches in museum collections in the western United States contain 1 to 11 eggs (Murray 1976).

There is little information on lifetime reproductive success (Haug 1993). Females supplemented with food will have higher reproductive success than females without supplemented food, which may explain poor reproductive success in areas with low-quality foraging habitat (Wellicome 1992). Depending on assumptions about migration, the probability that juvenile burrowing owls will survive to 1 year of age (the age of first breeding) has been estimated between 0.23 and 0.93, and annual adult survivorship between 0.42 and 0.93 (Johnson 1997).

During the breeding season, burrowing owls spend most of their time within 50 to 100 meters (162 to 325 feet) of their nest or satellite burrows (Haug and Oliphant 1990). During the day, they forage in the vicinity of the natal burrow, where they find it easy to prey on insects in low, open vegetation. Burrowing owls will nest in loose colonies, although owls display intraspecific territoriality immediately around nest burrow (Haug et al. 1993).

Foraging Requirements

This opportunistic feeder will consume arthropods, small mammals, birds, amphibians, and reptiles. Insects are often taken during the day, while small mammals are taken at night. In California, crickets and meadow voles were found to be the most common food items (Thomsen 1971). In urban areas, burrowing owls are often attracted to street lights, where insect prey congregates.

Owls have been detected foraging out to 1 mile from their burrows (Johnson pers. comm.). Inter-nest distances, which indicate the limit of an owl's territory, have been found to average between 61 and 214 meters (198 and 695 feet) (Thomsen 1971, Haug and Oliphant 1990). Nocturnal foraging can occur up to several kilometers away from the burrow, and owls concentrate their hunting uncultivated fields, ungrazed areas, and other habitats with an abundance of small mammals (Haug and Oliphant 1990).

Demography

The oldest recorded age of a burrowing owl is 8 years, 8 months (USGS records: http://www.pwrc.usgs.gov/bbl/homepage/long2890.htm). Collisions with vehicles are the most common cause of mortality in this species (Haug et al. 1993). Other sources of owl mortality include disease, exposure, and human activity around nests (digging or disking). Disturbance from dogs is another potential source of mortality (Thomsen 1971 *in* Haug et al. 1993).

Dispersal

Most of the North American populations migrate or disperse to wintering areas. In northern California, owls migrate south during September and October. Southern California populations are not migratory.

The spatial requirements of burrowing owls are not well understood. Breeding pairs of Western Burrowing Owls may require a minimum of 6.5 acres of contiguous grassland of high foraging quality to persist (California Department of Fish and Game 1995). However, burrowing owl pairs have been observed in isolated habitat patches as small as 1 acre. An area this size does not support the foraging requirements of most burrowing owls, and individuals occurring at sites this small must forage offsite. Reproductive success and long-term persistence in small and isolated habitats are unknown. Although the relationship between habitat area and population viability of this species is not well documented, small and isolated habitat patches are not likely to sustain high reproductive success or long-term persistence (see “Threats” below).

Behavior

Burrowing owls in California typically begin pair formation and courtship in February or early March, when adult males attempt to attract a mate. Loud “coo-cooing” at dusk indicates that this stage of the breeding cycle has begun. Beginning in April, eggs are laid at least 1 day apart and are incubated by both adults for about 3 to 4 weeks. Young owlets are brooded underground for another 3 to 4 weeks, at the end of which they may sometimes be seen at the burrow entrance in their natal-down plumage. Nestlings emerge asynchronously and tentatively in early June. They gradually become bolder, eventually spending more time outside, near the burrow entrance. During this period, nestlings can range widely on foot, even before they can fly. The adults guard their brood tenaciously, attacking intruders if provoked. Older nestlings or fledglings may move to nearby satellite burrows as the natal burrow becomes crowded.

Ecological Relationships

Western burrowing owls most commonly live in burrows created by California ground squirrels (*Spermophilis beecheyi*). Accordingly, the quality of burrowing owl habitat in the inventory area is closely and positively related to the occurrence and population health of ground squirrels in an area. Burrowing owls and ground squirrels can co-inhabit the same burrow system (Johnson pers. comm.), but the frequency with which this occurs has not been measured, and underground interactions have not been studied.

Threats

An immediate threat to the burrowing owl is the conversion of grassland habitat to urban and agricultural uses, and the loss of suitable agricultural lands to development. Equally important is the loss of fossorial rodents, such as prairie dogs and ground squirrels, across much of the owl's historical habitat. Eradication programs have decimated populations of these rodents and have in turn disrupted the ecological relationships on which owls depend—because Western Burrowing Owls need other animals to dig their burrows, the loss of fossorial rodents limits the extent of year-round owl habitat.

Another cause of population declines is thought to be pesticide use (especially organophosphates in southern Canada), but evidence does not clearly indicate that other contaminants are reducing populations (Gervais et al. 1997). Habitat fragmentation (Remsen 1978) probably increases foraging distances, making hunting less efficient and potentially reducing reproductive success. Fragmentation may reduce the chances that a male owl will attract a mate and could decrease reproductive success.

The population of Western Burrowing Owls in the Central Valley is threatened by conversion of habitat to urban uses and agriculture, particularly the conversion of grasslands to vineyards. Agricultural lands provide much lower quality habitat for burrowing owls than grasslands. Suitable habitat in agricultural areas is usually restricted to peripheral bands along the edges of plowed fields. These areas are often frequently disturbed and subject to loss from agricultural activities. Also, the loss of suitable agricultural land to development has reduced the extent of suitable habitat. Control of ground squirrels has reduced the extent and quality of potentially suitable burrowing owl habitat by reducing the number of suitable nesting burrows. The use of rodenticides and insecticides may have reduced prey populations, resulting in lowered survivorship and reproductive success.

In urban settings, owls occurring in isolated habitats may experience frequent disturbances from adjacent land uses (e.g., habitat degradation, predation) and barriers to foraging areas. Important biotic interactions between owls and rodent populations may be disrupted because some rodent populations are sensitive to habitat area and surrounding land uses as well. For example, the availability of rodent prey may be limited in isolated habitats, and ground squirrels may abandon or be eradicated from small parcels of habitat in urban settings. Also, small and isolated occurrences are more likely to experience random local extirpation as a result of natural disturbances (Goodman 1987), and recolonization of small or isolated habitat patches is less likely than recolonization of large habitat areas.

Conservation and Management

The burrowing owl is experiencing precipitous population declines throughout North America. In Canada, its numbers are rapidly declining, and, in 1995, the Committee on the Status of Endangered Wildlife in Canada listed it as endangered. In Mexico, it is officially considered threatened. The burrowing owl is also declining 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 and Ruhlen 1995, DeSante et al.1997). In the San Francisco Bay Area and the central portion of the Central Valley (from Yolo and Sacramento Counties to Merced County), the burrowing owl population has declined by at least 65% since 1986 (DeSante pers. comm.).

Common management efforts employed to conserve existing burrowing owl colonies include prevention of all disturbance during the nesting season, installation of permanent artificial burrows, and management of the vegetation around the burrows by mowing or controlled grazing.

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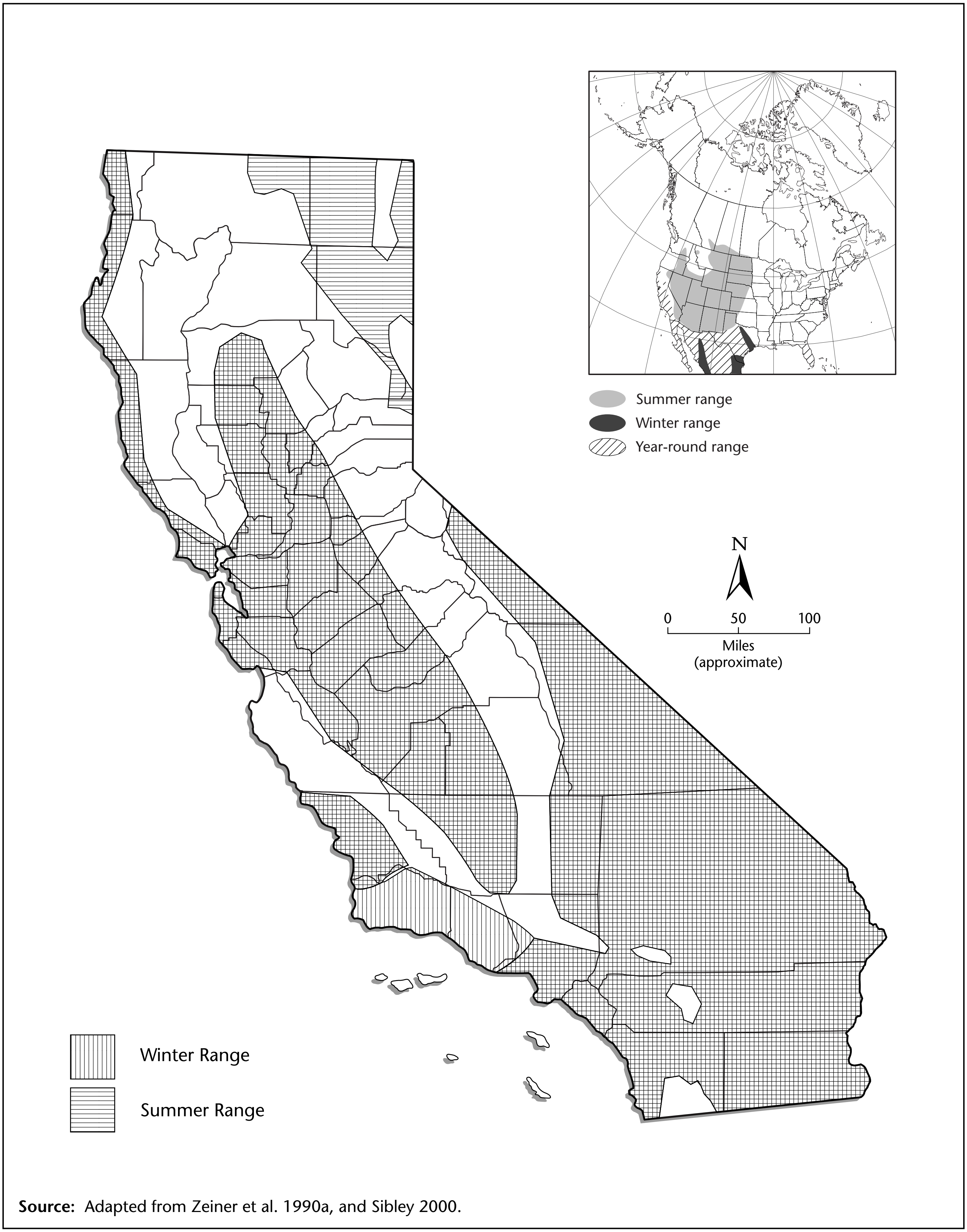
Glover

Johnson



Source: Adapted from Zeiner et al. 1990a, and Sibley 2000.

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Jones & Stokes

Western Burrowing Owl *(Athene cunicularia hypugea)* Distribution

Swainson's Hawk (*Buteo swainsoni*)

Status

State: Threatened
Federal: None

Population Trend

Global: Declining
State: Declining
Within Inventory Area: Unknown

Data Characterization

The location database for the Swainson's hawk (*Buteo swainsoni)* within the inventory area includes 4 data records from the California Natural Diversity Database (CNDDB) (2001), dated 1987 to 2000, and approximately 7 nest site locations within the last 4 years (Steve Glover pers. comm). Of these 11 records, 8 were documented within the last 10 years. All records are considered extant and mapped at high precision (nest may be accurately located within 80 meters).

A considerable amount of literature is available for the Swainson's hawk. Most of the literature pertains to habitat requirements, niche determination, competition with congeners, population trends, migration, and mortality from insecticide use on the wintering grounds.

Range

This diurnal raptor is a complete migrant, highly mobile, and has a large home range. Swainson's hawks breed in desert, shrubsteppe, grassland, and agricultural habitats in areas throughout most of the western U.S. and Canada, and in northern Mexico (England et al. 1995). They are locally common to rare breeders in California. Historically, breeding populations probably occurred throughout the state of California, except in bioregions characterized by mountainous forested terrain (Bloom 1980). Breeding populations in California currently occur in 2 locations, the Great Basin and the Central Valley. The largest population of breeding Swainson's hawks in California is located in the middle of the Central Valley between Sacramento and Modesto, and in the northern San Joaquin Valley. Swainson's hawks arrive on the breeding grounds in late February and early March in the Central Valley and in mid-April in the Great Basin. In September, most Swainson's hawks migrate to the Pampas of southern South America. However, the Central Valley population winters in Central Mexico and to a lesser extent throughout Central and South America (Bradbury et al. in prep.).

Occurrences within the ECCC HCP Inventory Area

During the breeding season, Swainson's hawks are found nesting throughout the inventory area. Most pairs have been observed nesting in small clumps of eucalyptus trees (Glover pers. comm.). There are 4 CNDDB (2001) records of Swainson's hawk nesting in the northeast section of the ECCC HCP/NCCP inventory area.

Biology

Habitat

In general, Swainson's hawks inhabit a wide variety of open habitats. In California's Central Valley, suitable habitat consists of 2 primary elements, suitable nest trees and proximity to high-quality foraging habitat. This species nests within riparian forest or in remnant riparian trees and forages in agricultural lands (such as fallow fields and alfalfa fields) (Estep 1989, Babcock 1995). Agricultural patterns and cover types influence suitability of foraging and home-range habitat. Habitat with the highest foraging value includes ruderal fields, fallow fields, grain crops, and safflower fields. In the Central Valley, extensive areas of unsuitable agricultural cover types may be the reason Swainson's hawks have large home-range sizes (mean 40.4 sq km) in this region (Babcock 1995).

Breeding

In the Central Valley, nest trees commonly used by Swainson's hawk include Fremont cottonwood (*Populus fremontia*), willow (*Salix* spp.), sycamore (*Plantanus racemosa*), Valley Oak (*Quercus lobata*), and walnut (*Juglans* spp.). Occasionally planted trees, such as eucalyptus (*Eucalyptus* spp.), pine (*Pinus* spp.) and (*Sequoia sempervirens*), are also used for nesting. Most of the known nests occur in stringers of remnant riparian forest along drainages (England et al. 1997).

Density of Swainson's hawks within their breeding territories is influenced by land use and availability of nest trees (Estep 1989). Nest trees may be isolated or in a riparian forest (England et al. 1997). Breeding habitat suitability is also dependent on surrounding landscape and abundance of prey. Nest placement tends to be in the upper canopy and semi-exposed, which may provide birds with a panoramic view of the territory. Tree and nest heights are higher in the Central Valley compared to nest trees in the western United States (Estep 1989).

Foraging

Historically, the Swainson's hawk probably foraged in upland and seasonally flooded perennial grasslands (Woodbridge 1998). Currently, Swainson's hawks forage in low-growing crops and are more abundant in areas of moderate cultivation than in either grassland areas or areas of extensive cultivation (Schmutz 1987). When ranking various habitats used by Swainson's hawks in the Central Valley, Estep (1989) found that perennial grassland and alfalfa fields ranked highest for foraging habitat suitability.

Central Valley Swainson's hawks prey on small mammals, birds, toads, crayfish, and insects. California voles (*Microtus* spp.), pocket gophers (*Thomomys bottae*), and deer mice (*Peromyscus maniculatus*) account for the majority of the mammalian prey species during the breeding season. Immediately after the breeding season and prior to migration, the majority of the diet consists of grasshoppers and crickets (Estep 1989). There is no data on diet for wintering Swainson's hawks (for the Central Valley population), but diet composition is probably made up of insects and to a lesser degree small mammals (Bradbury et al. in prep.).

Reproductive Capacity

During the breeding season, Swainson's hawks form monogamous pairs and will defend territories against conspecifics (Estep 1989). A clutch size is typically 1 to 4 eggs (Fitzner 1980, England et al. 1997). In general, Central Valley Swainson's hawks will have a single clutch, which will be completed by mid-April (Estep 1989). Rarely does this species attempt to renest if first nest attempt fails. The female does the majority of incubating, and the incubation period lasts 34 to 35 days (Fitzner 1980). In addition, the female does most of the brooding and shading of nestlings, while the male feeds the young for their first 2 to 3 weeks (England et al. 1997). Young fledge at approximately 38 to 46 days (England et al. 1997). The Central Valley population exhibits low reproductive success compared to populations in other areas. This is probably due to the complete alteration of native foraging habitat into cultivated fields and urban development (Estep 1989).

Breeding density is influenced by availability of nest trees and land use. High densities of breeding birds are associated with alfalfa fields, while low densities are associated with irrigated pasture and weedy fields (Woodbridge 1991). A mean breeding density of 30.23 pair/100 sq km was recorded in the Central Valley (Estep 1989).

Demography

There is little information on survival rates or longevity in this species (England et al. 1997). In Washington State, Swainson's hawks are thought to be long-lived (15–20 years) (Fitzner 1980). Mortality in nestlings is primarily due to starvation and predation from nest predators (England et al. 1997). Adult mortality results from human-caused sources, such as collisions with vehicles, gunshot, and pesticide application used to control grasshopper outbreaks (especially in South America) (England et al. 1997).

Dispersal

Juveniles remain with adults for 2 to 4 weeks after fledging, at which point they depart parental territories and form groups in areas where food is abundant. Adults also congregate at this time (in August) and forage on insects in fields (Fitzner 1980, Estep 1989). Juveniles and adults leave the breeding ground in September (Bradbury et al. in prep.).

Behavior

Swainson's hawks build nests out of sticks, plant parts, and other weeds. Woodbridge (1998) found that some nests appeared flimsy and might not last the winter. Courtship displays occur near the nest site. They involve circling and steep dives (England et al. 1997).

During the breeding season, Swainson's hawks travel up to 29 km in search of prey (Estep 1989, Woodbridge 1991). This species spends large amounts of time foraging while soaring over open habitats. Foraging behavior in the Central Valley is associated with cultivation activities that expose prey (e.g. flood irrigation, burning, and disking). Large flocks of non-breeding individuals will forage and roost communally during the breeding season, eating a variety of prey that ranges from bats to flying insects (England et al. 1997, Woodbridge 1998).

Home-range size is dependent on proximity to foraging sites and the distribution of high-quality foraging habitat. The home-range size for pairs nesting in the Central Valley ranged from 336 to 8,718 hectares (Estep 1989) in one study, and from 724 to 7,659 hectares (Babcock 1995) in another study. The smallest home ranges were observed in areas where nest sites in riparian forest habitat were close to alfalfa or similar, recently harvested row crops (Estep 1989).

Ecological Relationships

There is no information on predation of adults (England et al. 1997). Researchers have observed egg and nestling predation by American crows (*Corvus brachynchos*), great horned owls (*Bubo virginianus*), and golden eagles (*Aquila chrysaetos)*.

Threats

Loss of high-quality foraging habitat is probably the most significant threat to the species' population within the inventory area. Loss of nesting habitat (remnant riparian) may be a threat to this species statewide. In addition, nest sites on private lands are vulnerable to changes in development and agricultural practices.

Swainson's hawks show a strong association with riparian forests. Protection and restoration of these habitats may therefore be important to the recovery of the species. As mentioned above, presence of suitable nest trees combined with proximity to high-quality foraging habitat is necessary for the reproduction of this species.

Current DFG guidelines for mitigation of loss of foraging habitat are not sufficient because the guidelines allow for losses of foraging habitat throughout the remainder of the region (Estep pers. comm.). The guidelines do not consider cumulative effects of agricultural intensification and conversion of crops that provide high-quality foraging habitat to crops that provide low-quality foraging habitat (e.g. alfalfa to vineyards).

Conservation and Management

The majority of the state's breeding sites are located in 2 disjunct populations: 1 in the Great Basin in the northeast corner of the state, and the other in the Central Valley. The largest population of this species is located within the inventory area between Sacramento and Modesto. Estep (pers comm.) estimates that this population includes approximately 900 breeding pairs.

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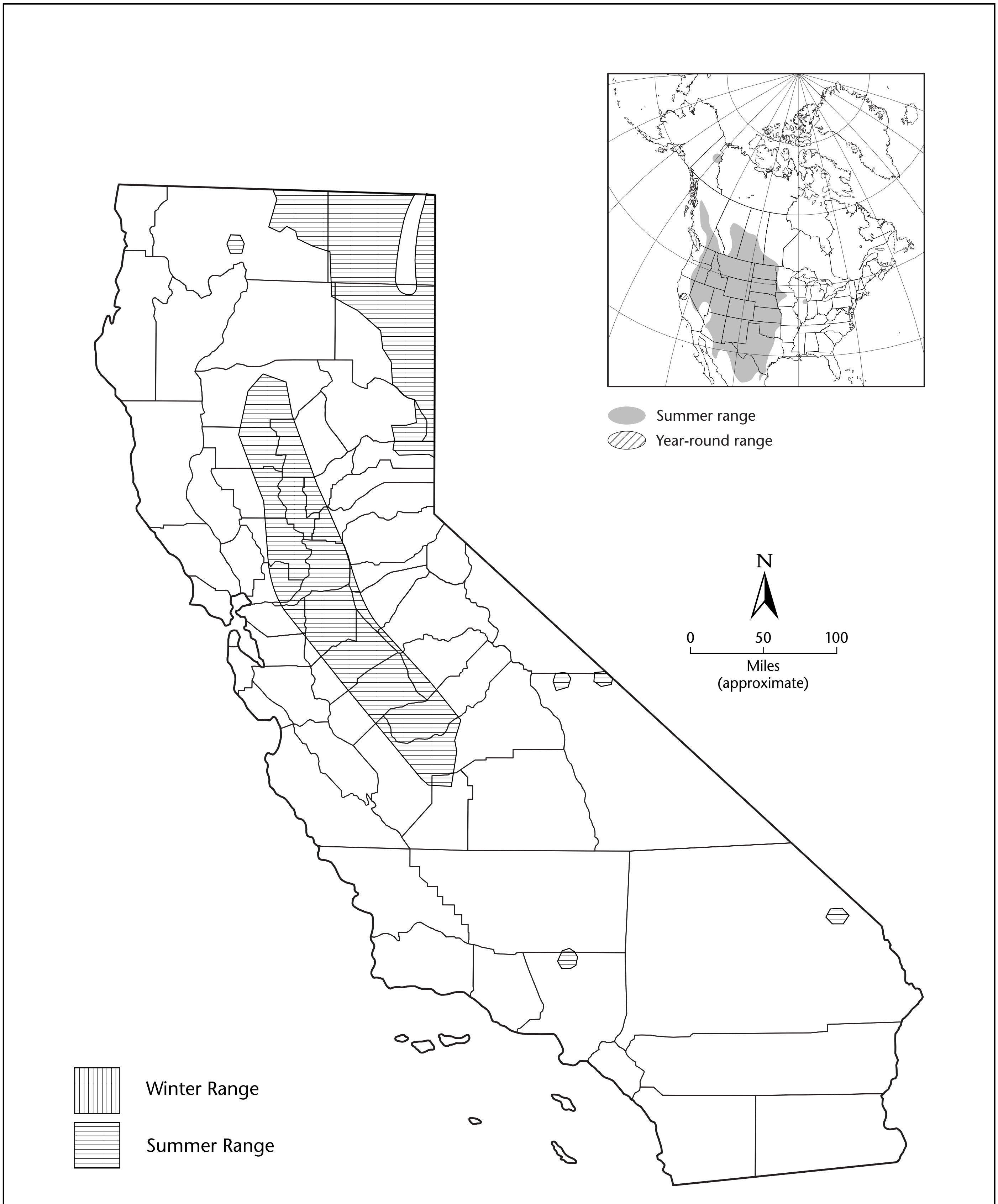

Source: Adapted from Zeiner et.al. 1990a, and Sibley 2000.

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Jones & Stokes

Swainson's Hawk *(Buteo swainsoni)* Distribution

Silvery Legless Lizard (*Anniella pulchra pulchra*)

Status

State: Species of Concern
Federal: Species of Concern

Population Trend

Global: Declining
State: Declining
Within Inventory Area: Unknown

Data Characterization

The location database for the silvery legless lizard (*Anniella pulchra pulchra)* within its known range in California includes 14 data records dated from 1988 to 2000. Of these records, 12 were documented within the past 10 years; of these, 9 are of high precision and may be accurately located. One of these records is located within the inventory area, at the East Bay Regional Park District Legless Lizard Preserve.

A small amount of literature is available for the silvery legless lizard because of its cryptic behavior and general difficulty to find. Most of the available literature pertains to natural history and reproductive patterns.

Range

The silvery legless lizard is nearly endemic to California. It ranges from Antioch in Contra Costa County south through the Coast, Transverse, and Peninsular Ranges, along the western edge of the Sierra Nevada Mountains and parts of the San Joaquin Valley and Mojave Desert to El Consuelo in Baja California (Hunt 1983, Jennings and Hayes 1994). Its elevation range extends from near sea level on the Monterey Peninsula to approximately 1,800 meters above sea level in the Sierra Nevada foothills.

Occurrences within the ECC HCP Inventory Area

The East Bay Regional Park District Legless Lizard Preserve is located east of the intersection of Highway 4 and Big Break Road north of Oakely. This is the only California Natural Diversity Database record for this species in the inventory area, but other occurrences are likely to exist within the inventory area due to the presence of suitable habitat.

Biology

Habitat

Silvery legless lizards occur primarily in areas with sandy or loose loamy soils such as under sparse vegetation of beaches, chaparral, or pine-oak woodland; or near sycamores, cottonwoods, or oaks that grow on stream terraces (Gorman 1957, Cunnignham 1959), Banta and Morafka 1968, Stebbins 1985, Jennings and Hayes 1994). The sandy loam soils of stabilized dunes seem to be especially favorable habitat (Grinnel and Camp 1917, Miller 1944, Smith 1946, Bury 1985). The species is often found under or in the close vicinity of logs, rocks, old boards, and the compacted debris of woodrat nests (Jennings and Hayes 1994). Rocky soils or areas disturbed by agriculture, sand mining, or other human uses is not suitable for legless lizards (Miller 1944, Bury 1972, Hunt 1983, Stebbins 1985). Soil moisture is essential for legless lizards to conserve energy at high temperatures; it also allows shedding to occur (Jennings and Hayes 1994).

Foraging Requirements

Adult and juvenile lizards are insectivorous and subsist largely on larval insects (especially moths and beetles), adult beetles, termites, and spiders (Jennings and Hayes 1994).

Reproduction

Silvery legless lizards are live-bearing and are believed to breed between early spring and July (Goldberg and Miller 1985). Oviductal eggs are observed in females from July through October (Goldberg and Miller 1985), and litters of 1 to 4 (normally 2) young are born from September to November (Miller 1944). Gestation lasts about 4 months (Goldberg and Miller 1985). Young lizards typically reach sexual maturity in 2 to 3 years (for males and females, respectively).

Demography

The longevity of the silvery legless lizard populations in the wild is unknown. However, sexually mature adults have lived for almost 6 years under laboratory conditions (Jennings and Hayes 1994).

Behavior

Legless lizards are fossorial animals that construct burrows in loose sandy soil (Miller 1944, Stebbins 1954). They appear to be active mostly during the morning and evening, when they rest just beneath the surface of sunlight-warmed substrate. They may also be active on the surface at night when substrate temperatures remain warm for extended intervals.

Ecological Relationships

Known predators of legless lizards include ring neck snakes (*Diadophis punctatus*), common king snakes (*Lampropeltis getulus*), deer mice (*Peromyscus maniculatus*), long-tailed weasels (*Mustela frenata*), domestic cats (*Felis sylvestris*), California thrashers (*Toxostomea redivivum*), American robins (*Turdus migratorius*), and loggerhead shrikes (*Lanius ludovicianus*) (Jennings and Hayes 1994).

Threats

The legless lizard's specialization for a fossorial existence in substrates with a high sand fraction makes it vulnerable to many types of habitat loss and disturbance. Legless lizards cannot survive in urbanized, agricultural, or other areas where a loose substrate in which to burrow has been removed or altered (e.g., disturbed by blowing or bulldozing) (Jennings and Hayes 1994). Other factors can alter the substrate such that the species cannot survive in the area any longer. These factors include livestock grazing, off-road vehicles activities, sand mining, beach erosion, excessive recreational use of coastal dunes, and the introduction of exotic plant species, such as ice plants (*Carpobrotus edulis* and *Mesembryanthemum crystallinum*), Marram grass (*Ammophila arenaria*), veldt grass (*Ehrharata calycina*) and eucalyptus trees (*Eucalyptus* spp.). These factors decrease soil moisture or alter the conformation of the substrate, which may act to limit the food base or make the substrate physically unsuitable for legless lizards (Jennings and Hayes 1994). Pesticides may also threaten legless lizards because of the species' insectivorous diet (Honegger 1975). Increasing numbers of feral cats associated with residential areas also threaten extant populations of this species (Miller 1944, Jennings and Hayes 1994).

Conservation and Management

Detailed studies of legless lizard habitat requirements need to be conducted to determine the distribution and ecological needs of this species more precisely.

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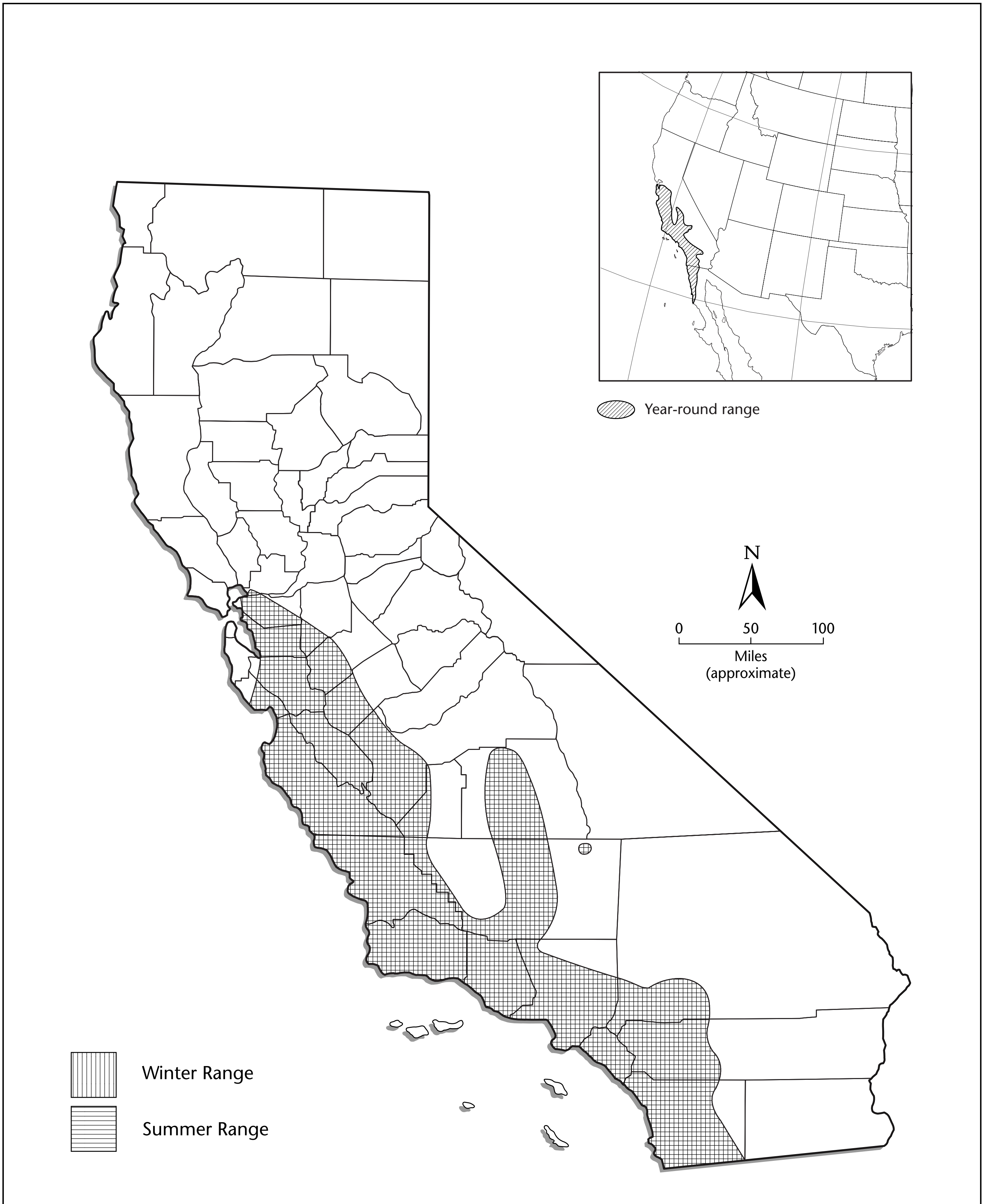

Source: Adapted from Zeiner et al. 1988.

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Jones & Stokes

Silvery Legless Lizard *(Anniella pulchra pulchra)* Distribution

Alameda Whipsnake (*Masticophis lateralis euryxanthus*)

Status

State: None
Federal: Threatened

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

There are 19 California Natural Diversity Database (CNDDB) records within the inventory area. The precision of these records ranges from an 80-meter circle to a 1-mile-radius circle (Darlene McGriff pers comm.). Of these 19 records, only 5 were recorded within the last 10 years, and the remaining 14 were documented as early as 1980. All of these CNDDB records are considered extant.

Alameda whipsnake (*Masticophis lateralis euryxanthus*) literature is scarce, probably because of the restricted range of this species. Available published literature consists of 1 master's of science (M.S.) thesis, 3 species accounts, and 1 survey. The U.S. Fish and Wildlife Service (USFWS) is in the process of developing a draft recovery plan for this species that will be available spring of 2002 (Heather Bell pers. comm). Ecological information available for this species is limited to a single live trapping and radio telemetry study conducted by Karen Swaim in 1994.

Range

The Alameda whipsnake is a subspecies of the California whipsnake (*Masticophis lateralis*). The North American distribution for the California whipsnake includes Northern California west of the Sierran Crest and desert to central Baja California. This species is absent from the floor of the Central Valley, and its California distribution parallels that of chaparral habitat (Stebbins 1985). The Alameda whipsnake's range is restricted to the inner Coast Range in western and central Contra Costa and Alameda Counties (U.S. Fish and Wildlife Service 2000). The historical range of the Alameda whipsnake has been fragmented into 5 disjunct populations (U.S. Fish and Wildlife Service 1997): Tilden–Briones, Oakland–Las Trampas, Hayward–Pleasanton Ridge, Sunol–Cedar Mountain, and the Mount Diablo–Black Hills (U.S. Fish and Wildlife Service 1997).

Occurrences within the ECCC HCP/NCCP Inventory Area

Of the 48 CNDDB (2001) records for the Alameda whipsnake in the state, 19 records occur within the ECCC HCP/NCCP inventory area. A large portion of the Mount Diablo–Black Hills population of the Alameda whipsnake occurs within the ECCC HCP/NCCP inventory area.

Biology

Habitat

The Alameda whipsnake occurs primarily in coastal scrub and chaparral communities, but also forages in a variety of other communities in the inner Coast Range, including grasslands and open woodlands (Swaim 1994). Rock outcrops with deep crevices or abundant rodent burrows are important habitat components for overnight dens, refuges from predators and excessive heat, and foraging (Swaim 1994). According to USFWS (2000), suitable habitat for this species includes communities that support mixed chaparral, coastal scrub, and annual grassland and oak woodlands that are adjacent to scrub habitats. Grassland areas that are linked to scrub by rock outcrops or river corridors are also considered primary constituent elements (U.S. Fish and Wildlife Service 2000).

The Alameda whipsnake requires open and partially open, low-growing shrub communities for many of its biological needs. This habitat provides cover for snakes during dispersal, cover from predators, and a variety of microhabitats where whipsnakes can move to regulate their body temperature (Swaim 1994). Whipsnakes exhibit a high degree of stability and a high mean activity in body temperature (33.4 degrees centigrade). Whipsnake habitat must consist of a mix of sunny and shadey sites in order to provide a range of temperatures for the snake's activities (Swaim 1994, U.S. Fish and Wildlife Service 2000). A sparse shrub canopy is ideal because it also provides a visual barrier from avian predators (Swaim 1994).

Other important habitat features include small mammal burrows, rock outcrops, talus, and other forms of shelter that provide snakes with alternative habitats for temperature regulation, protection from predators, egg-laying sites, and winter hibernaculum (winter residence where the snakes hibernate). Alameda whipsnakes spend November through March in a winter hibernaculum (U.S. Fish and Wildlife Service 2000).

Home-range size for male snakes in Alameda and Contra Costa counties (Tilden Park and Moller Ranch) varies in size from 1.9 to 8.7 hectares (ha) (mean = 5.5 ha). Home-range size for female snakes was 3.9 and 2.9 hectares (Swaim 1994). When movements of individual snakes were monitored (2 males and 1 female) in these areas, results indicated that most of the home range was not used. Both male and female snakes repeatedly returned to core retreat areas within their home range after intervals of non-use. These snakes did exhibit overlap in use of these relatively large home ranges, and there was no evidence of territorial behavior in this species (Swaim 1994).

Breeding Habitat Requirements

Mating occurs from late March through mid-June (U.S. Fish and Wildlife Service 2000). Whipsnakes lay a clutch of 6 to 11 eggs (Stebbins 1985), probably in loose soil or under logs or rocks (Zeiner et al. 1988). According to Swaim (1994), female Alameda whipsnakes will use grassland habitat for egg laying. Little else is known about habitat requirements for breeding and egg laying (Zeiner et al. 1988). Swaim (1994) documented that courtship and mating occur near the female's hibernaculum. During the breeding season, male snakes exhibit more movement throughout their home range, while female snakes remain sedentary from March until egg laying (Swaim 1994).

Foraging Requirements

Whipsnakes prey on a variety of vertebrate species, including frogs, lizards, nestling birds, and rodents (Zeiner et al. 1988). Rock outcrops are particularly important for the Alameda whipsnake because they support many of the species' prey (U.S. Fish and Wildlife Service 2000). Additionally, the Alameda whipsnake has been observed foraging in grassland habitats adjacent to native Diablan sage scrub habitats (Swaim 1994).

Occupied areas usually support a prey base of at least 2 lizard species, especially the western fence lizard (*Sceloporus occidentalis*) (Stebbins 1985).

Demography

There have been no studies of the demography or longevity of Alameda whipsnakes.

Dispersal

The Alameda whipsnake is non-migratory. There is little information on site fidelity and patterns of dispersal in this species; however, Swaim (1994) observed evidence of individual snakes using the same home range in successive years.

Behavior

The Alameda whipsnake is a fast moving, diurnal predator that forages actively on the surface (Zeiner et al. 1988). Alameda whipsnakes have 2 seasonal peaks in activity, 1 during the spring mating season and the other during late summer/early fall. During the first peak in activity males will move throughout their home range, while females remain close to their hibernaculum. Male movement appears to be associated with foraging and searching for mates. Females exhibit a peak in activity only for a few days during the spring when they move to an area outside their normal range, presumably to find egg-laying sites (Swaim 1994). After reproductive activities are completed, male and female movements resume similar patterns. In mid-June, both males and females exhibit decreased activity levels, though evidently this species does not estivate during the summer months (Swaim 1994). The second peak in seasonal activity occurs in late summer/fall. During this time, Swaim (1994) recorded activity in both hatchling and adult snakes, possibly in response to an increase in the availability of prey (hatchling lizards).

Ecological Relationships

Diurnal predators, especially raptors, prey on adult Alameda whipsnakes. Nocturnal mammals likely prey on Alameda whipsnake eggs (Zeiner et al. 1988). Basking in open terrain may expose snakes to predators such as red-tailed hawks (Fitch 1949 in Swaim 1994).

Threats

Alameda whipsnake populations have declined from loss of habitat resulting from urban expansion (U.S. Fish and Wildlife Service 2000). Urban development, particularly road and highway construction, has also fragmented Alameda whipsnake populations and made them more vulnerable to extinction (U.S. Fish and Wildlife Service 1997). Urban development adjacent to whipsnake habitat increases the likelihood of predation from feral cats and injury or death from public recreational use. Other significant threats to this species' recovery include inappropriate grazing practices and alteration of habitat through fire suppression (U.S. Fish and Wildlife Service 1997).

Fire suppression alters suitable Alameda whipsnake habitat in 2 important ways. First, fire suppression increases the chances of large catastrophic fires occurring in areas where vegetation has become overgrown. A buildup of flammable fuel loads in Alameda whipsnake habitat can lead to high intensity fire events that may be detrimental to this species. Second, fire suppression leads to a closed scrub canopy which tends to reduce the diversity of microhabitats that whipsnakes require(Swaim 1994).

Conservation and Management

The USFWS lists the Mount Diablo–Black Hills population of the Alameda whipsnake as having a high potential for recovery if threats from urban development, catastrophic wildfire, and grazing practices can be managed well (U.S. Fish and Wildlife Service 2000). As of October 2000, there had been no approved HCPs that cover the Alameda whipsnake or its habitat (U.S. Fish and Wildlife Service 2000).

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Source: BioSystems 1999.

Jones & Stokes

Alameda Whipsnake (*Masticophis lateralis euryxanthus*)
Known Occurrences

Giant Garter Snake (*Thamnopsis gigas*)

Status

State: Threatened
Federal: Threatened

Population Trend

Global: Declining
State: Declining
Within Inventory Area: Unknown

Data Characterization

The location database for the giant garter snake (*Thamnopsis gigas)* within its known range in California includes 142 data records from 1908 to 2000. Of these, 30 were documented within the past 10 years, 12 or which are of high precision and may be accurately located. Two of these records are located outside but near the ECCC HCP/NCCP inventory area. This database includes records of individual sightings and locations of occupied, vacant, and natal dens.

A moderate amount of literature is available for the giant garter snake because of its threatened status. Most of the literature pertains to habitat requirements, distribution, population demographics, threats, and management activities. A recovery plan for the giant garter snake has been published (U.S. Fish and Wildlife Service 1999).

Range

The giant garter snake is endemic to the valley floor of the Sacramento and San Joaquin Valleys of California. Records coincide with the historical distribution of large flood basins, freshwater marshes, and tributary streams of the Central Valley of California (Hansen and Brode 1980). The historic distribution of the giant garter snake extended from Sacramento and Contra Costa Counties southward to Buena Vista Lake near Bakersfield in Kern County.

Occurrences within the ECCC HCP/NCCP Inventory Area

No records of giant garter snake have been documented within the ECCC HCP/NCCP inventory area. However, suitable habitat occurs in the slough areas and drainage network associated with agricultural fields in the northeast section of the County (U.S. Fish and Wildlife Service 1999). The lack of records from this area may be due to a lack of survey effort.

Biology

Habitat

The giant garter snake inhabits agricultural wetlands and associated waterways, including irrigation and drainage canals, rice fields, marshes, sloughs, ponds, small lakes, low-gradient streams, and adjacent uplands (U.S. Fish and Wildlife Service 1999). Important features of these habitats include: 1) sufficient water during the snake's active season (early spring through mid–fall) to maintain an adequate prey base; 2) emergent vegetation, such as cattails (*Typha* spp.) and bulrushes (*Scirpus* spp.), for escape cover and foraging habitat; 3) upland habitat with grassy banks and openings to waterside vegetation for basking; and 4) higher elevation upland areas for cover and refuge from flood waters during the snake's inactive season (Hansen 1980, 1988, Brode and Hansen 1992, Hansen and Brode 1993). Giant garter snakes are absent from the larger rivers; wetlands with sand, gravel, or rock substrates; and riparian 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).

Foraging Requirements

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

Reproduction

The breeding season for the giant garter snake extends from March through May and resumes briefly during September (G. Hansen pers. comm. *in* 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 to 46 young (mean = 23) from late July through early September (Hansen and Hansen 1990). The young immediately disperse to dense cover where they absorb their yolk sac, then start feeding independently. The young will typically have doubled in size by 1 year of age (G. Hansen pers. comm. *in* U.S. Fish and Wildlife Service 1999), and sexual maturity usually takes 3 years in males and 5 years in females.

Demography

No studies of the longevity of giant garter snakes have been conducted.

Behavior

Giant garter snakes are most active from early spring through mid-fall; activity being dependent on local weather conditions (Brode 1990, Hansen and Brode 1993). During the 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 in close proximity to wetland habitats but can move over 800 feet from the water (G. 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 (Wylie et al. 1997).

Ecological Relationships

Giant garter snakes prey on a variety of fish and amphibians available within their habitat and are in turn prey for raccoons (*Procyon lotor*), striped skinks (*Mephitis mephitis*), opossum (*Didelphis virginiana*), red foxes (*Vulpes vulpes*), gray foxes (*Urocyon cinereoargenteus*), hawks (*Buteo* spp.), northern harriers (*Circus cyaneus*), great egret (*Ardea alba*), snowy egret (*Egretta thula*), American bittern (*Botaurus lentiginosus*), and great blue herons (*Ardea herodias*). Giant garter snakes may coexist with 2 other species of garter snake: the valley garter snake (*T. sirtalis fitichi*) and the western terrestrial garter snake (*T. elegans*) (R. Hansen 1980, G. Hansen 1986). This coexistence may be possible because of differences in foraging behavior (U.S. Fish and Wildlife Service 1999).

Threats

Habitat loss, degradation, and fragmentation are the primary threats to giant garter snake population viability (U.S. Fish and Wildlife Service 1999). Conversion of wetlands for agricultural, urban, and industrial development has resulted in the loss of over 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 (Brode and Hansen 1992, California Department of Fish and Game 1992, G. Hansen 1988, Hansen and Brode 1993).

Introduction of non-native predators, including the 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 probably had detrimental effects on the giant garter snake through direct predation (sensu Bury and Whelan 1984, Treanor 1993) and competition for smaller forage fish (California Department of Fish and Game 1992, G. Hansen 1986, Schwalbe and Rosen 1989).

Toxic contamination, particularly from selenium, and impaired water quality have also been identified as threats to some populations of the giant garter snake (Ohlendorf et al. 1988, Saiki and Lowe 1987, U.S. Fish and Wildlife Service 1993). Preliminary studies have documented potential bioaccumulative effects of agriculturally derived contaminants on giant garter snakes or their prey species (see Saiki et al. 1992, 1993). Disease and parasitism, (potentially related to reduced immune response ability from contaminants), may also pose a threat to this species (U.S. Fish and Wildlife Service 1999).

Conservation and Management

The giant garter snake was listed as threatened in California in 1971 and federally in 1993. Subsequent conservation actions have included the establishment of guidelines and mechanisms to minimize and mitigate take (U.S. Fish and Wildlife Service 1999), habitat and population surveys (G. Hansen 1982,1986, 1996, Hansen and Brode 1980), and development of management plans for public lands and land acquisitions (U.S. Fish and Wildlife Service 1999). A draft recovery plan for the giant garter snake was completed in 1999 (U.S. Fish and Wildlife Service 1999).

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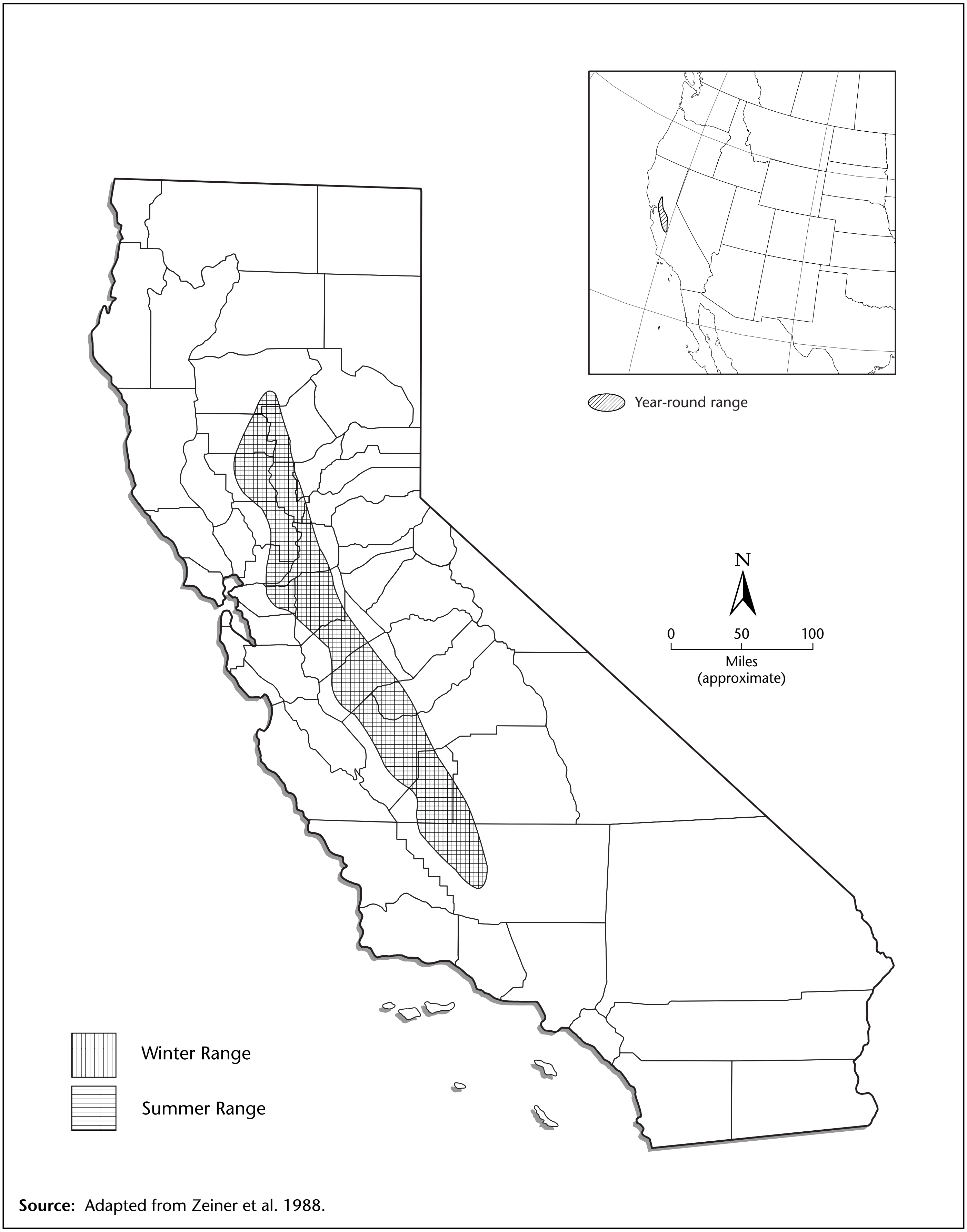

Jones & Stokes

Giant Garter Snake *(Thamnophis gigas)* Distribution

California Tiger Salamander (*Ambystoma californiense*)

Status

State: Species of Special Concern
Federal: Federal Candidate Species; Endangered (Santa Barbara population only); Petitioned for listing as endangered (Sonoma County population)

Population Trend

Global: California State endemic; declining (Jennings and Hayes 1994)
State: Declining (Jennings and Hayes 1994)
Within Inventory Area: Unknown

Data Characterization

The location database for the California tiger salamander (*Ambystoma californiense*) within the inventory area includes 96 data records dated from 1920 to 1999. Of these records, 45 were documented within the past 10 years. Of the 45 records, all are considered extant, and 37 are mapped at a "specific" precision level (within 80 meters).

There is very little general information on the ecology of the California tiger salamander and few peer-reviewed research studies. Available literature includes research on reproductive ecology, burrowing ability, dispersal from breeding area, habitat use and migratory behavior. The lack of data may be due to the fact that this species spends most of its life underground in small mammal burrows (U.S. Fish and Wildlife Service 2000). There are many gaps in data for the California tiger salamander, including habitat and population distribution, and differentiating between introduced tiger salamanders and California tiger salamanders. The California tiger salamander was not recognized as distinct species until 1991 (U.S. Fish and Wildlife Service 2000).

Range

The California tiger salamander is endemic to California. Historically, the California tiger salamander probably occurred in grassland habitats throughout much of the state. Habitat conversion has reduced the species' range and decreased breeding populations (Stebbins 1985). Currently, the California tiger salamander occurs in the Central Valley and Sierra Nevada foothills, from Yolo County or Colusa County south to Tulare County, and in the coastal valleys and foothills, from Sonoma County south to Santa Barbara County (Zeiner et al. 1988). Isolated populations are found at the Gray's Lodge Wildlife Area in Butte County and at Grass Lake in Siskiyou County (Zeiner et al. 1988). Most populations occur at elevations below 1,500 feet, but tiger salamanders have been recorded at elevations up to 4,500 feet. Although populations have declined, the species continues to breed at a large number of locations within its current range (59 FR 18353–18354, April 18, 1994).

Occurrences within the ECCC HCP Inventory Area

Because a comprehensive survey for the California tiger salamander has not been conducted in the HCP inventory area, neither the current population size nor the locations of all occurrences are known.

Biology

Habitat

California tiger salamanders require 2 major habitat components: aquatic breeding sites and terrestrial estivation or refuge sites. California tiger salamanders inhabit valley and foothill grasslands and the grassy understory of open woodlands, usually within 1 mile of water (Jennings and Hayes 1994). The California tiger salamander is terrestrial as an adult and spends most of its time underground in subterranean refugia. Underground retreats usually consist of ground-squirrel burrows and occasionally human-made structures. Adults emerge from underground to breed, but only for brief periods during the year. Tiger salamanders breed and lay their eggs primarily in vernal pools and other ephemeral ponds that fill in winter and often dry out by summer (Loredo et al. 1996); they sometimes use permanent human-made ponds (e.g., stock ponds), reservoirs, and small lakes that do not support predatory fish or bullfrogs (see “Ecological Relationships” discussion below) (Stebbins 1972, 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 up to 1 mile from their breeding sites (Jennings and Hayes 1994).

The California tiger salamander is particularly sensitive to the duration of ponding in aquatic breeding sites. Because tiger salamanders have a long developmental period, the longest lasting seasonal ponds or vernal pools are the most suitable type of breeding habitat for this species; these pools are also typically the largest in size (Jennings and Hayes 1994). Because at least 10 weeks are required to complete metamorphosis (see “Demography” below) (Feaver 1971), aquatic sites that are considered suitable for breeding should at least pond or retain water for a minimum of 10 weeks. 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 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). The presence and abundance of tiger salamanders in many areas are limited by the number of small-mammal burrows available; salamanders are typically absent from areas that appear suitable other than their lack of burrows. Loredo et al. (1996) emphasized the importance of California ground squirrel burrows as refugia for California tiger salamanders, and suggested that a commensal relationship existed between the California tiger salamander and California ground squirrel in which tiger salamanders benefit from the burrowing activities of squirrels. In a study conducted near Concord, California, Loredo et al. (1996) found that California ground squirrel burrows were used almost exclusively as refuge sites by California tiger salamanders. Also, tiger salamanders apparently do not avoid burrows occupied by ground squirrels (Loredo et al. 1996).

The proximity of refuge sites to aquatic breeding sites also affects the suitability of salamander habitat. 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 mile (1.6 km) from breeding sites (Austin and Shaffer 1992, Mullen *in* U.S. Fish and Wildlife Service 2000). 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 118 feet (35.9 m) and 85 feet (26.0 m), respectively. Therefore, although salamanders may migrate up to 1 mile, migration distances are likely to be less in areas supporting refugia closer to breeding sites. Also, habitat complexes that include upland refugia relatively close to breeding sites are considered more suitable because predation risk and physiological stress in California tiger salamanders probably increases with migration distance.

Breeding Habitat Requirements

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). Tiger salamanders are rarely observed except during this period (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 winter and often dry by summer (Loredo et al. 1996). Spawning usually occurs within a few days after migration, and adults probably leave the breeding sites at night soon after spawning (Barry and Shaffer 1994 citing Storer 1925). 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).

California tiger salamanders breed in vernal pools and other temporary rainwater ponds. This species will also use permanent human-made ponds, without predatory fish, for reproduction. Females lay eggs on submerged vegetation in shallow water. In ponds without vegetation, females will lay eggs on objects on the pond bottom (Jennings and Hayes 1994). After breeding, adults leave the breeding ponds and return to small mammal burrows.

After approximately 2 weeks, the salamander eggs begin to hatch into larvae. Once larvae reach a minimum body size they metamorphose to the terrestrial juvenile salamander. Larvae in small ponds develop faster, while larvae inhabiting ponds that retain water for longer will be larger at time of metamorphosis. In general, salamanders require 10 weeks living in ponded water for complete metamorphosis. If a pond dries prior to metamorphosis, the larvae will desiccate and die (U.S. Fish and Wildlife Service 2000).

The California tiger salamander breeds primarily in vernal pools and swales—unique ecosystems that fill with winter rains and dry completely by summer—and then spends most of its lifecycle estivating underground in adjacent valley oak woodland or grassland habitat, primarily in abandoned rodent burrows. Research has shown that dispersing juveniles can roam up to 1 mile from their breeding ponds and that a minimum of 480 acres of uplands habitat is needed surrounding a breeding pond in order for the species to survive over the long term. Reserves of multiple breeding ponds surrounded by 1000 acres or more of habitat are recommended to ensure the persistence of the species.

Foraging Requirements

Aquatic larvae feed on algae, small crustaceans, and small mosquito larvae for about 6 weeks after hatching (U.S. Fish and Wildlife Service 2000). Larger larvae feed on zooplankton, amphipods, mollusks, and smaller tadpoles of pacific treefrogs, red-legged frogs, western toads and spadefoot toads (Zeiner et al. 1988, U.S. Fish and Wildlife Service 2000). During estivation, California tiger salamanders eat very little (Shaffer et al. 1993 *in* U.S. Fish and Wildlife Service 2000). 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 2000). Adults eat earthworms, snails, insects, fish, and small mammals (Stebbins 1972).

Demography

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). Individuals have been known to live for more than 10 years (Trenham et al. 2000 *in* U.S. Fish and Wildlife Service 2000).

Dispersal

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

Ecological Relationships

California tiger salamander larvae and embryos 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 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 detected only in unvegetated ponds. This suggests that vegetation structure in aquatic breeding sites may be important for survival. Because of their secretive behavior and limited periods above ground, adult California tiger salamanders have few predators (U.S. Fish and Wildlife Service 2000).

Threats

California tiger salamander populations have experienced dramatic declines throughout the historical range of the species, particularly in the Central Valley. California tiger salamander populations have declined as a result of 2 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 California tiger salamander's range have removed or fragmented vernal pool complexes, eliminated refuge sites adjacent to breeding areas, and reduced habitat suitability for the species over much of the Central Valley (Barry and Shaffer 1994, Jennings and Hayes 1994). Grading activities have probably also eliminated large numbers of salamanders directly (Barry and Shaffer 1994).

The introduction of bullfrogs, Louisiana red swamp crayfish, and non-native fishes (mosquitofish, bass, and sunfish) into aquatic habitats has also contributed to declines in tiger salamander populations (Jennings and Hayes 1994; 59 FR 18353–18354, April 18, 1994, U.S. Fish and Wildlife Service 2000). These non-native 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 2000).

Vehicular related mortality is an important threat to California tiger salamander populations (Barry and Shaffer 1994, Jennings and Hayes 1994). California tiger salamanders will readily attempt to cross roads during migration, and roads that sustain heavy vehicle traffic or barriers that impede seasonal migrations may have impacted tiger salamander populations in some areas (Shaffer and Fisher 1991, Shaffer and Stanley 1992, Barry and Shaffer 1994). Therefore, establishing artificial structures that could impede movements or maintaining 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).

Conservation and Management

The California tiger salamander is a Federal Candidate Species and California Species of Special Concern within the ECCC HCP/NCCP inventory area (U.S. Fish and Wildlife Service 1994). On January 19, 2000, the Santa Barbara County population of the California tiger salamander was listed as endangered species on an emergency listing (U.S. Fish and Wildlife Service 2000). On June 11, 2001, the Sonoma County population of California tiger salamander was petitioned for listing as endangered on an emergency basis (Center for Biological Diversity and Citizens for a Sustainable Cotati 2001). These actions strongly indicate the imperiled nature of this species and the potential that important habitat loss within the ECCC HCP/NCCP inventory area could lead to listing of the Contra Costa population of tiger salamanders during the projected 50-year HCP permit period. Currently, neither the Candidate listing nor the Species of Concern designation provides formal protection to this species.

Existing conservation measures for this species include preservation of occupied habitat, mitigative replacement of lost habitat, and prevention of contamination of aquatic habitat used by the species. Research has shown that dispersing juveniles can roam up to 1 mile from their breeding ponds and that a minimum of 480 acres of uplands habitat is needed surrounding a breeding pond in order for the species to survive over the long term. Reserves of multiple breeding ponds surrounded by 1000 acres or more of habitat are recommended to ensure the persistence of the species (Center for Biological Diversity 2002).

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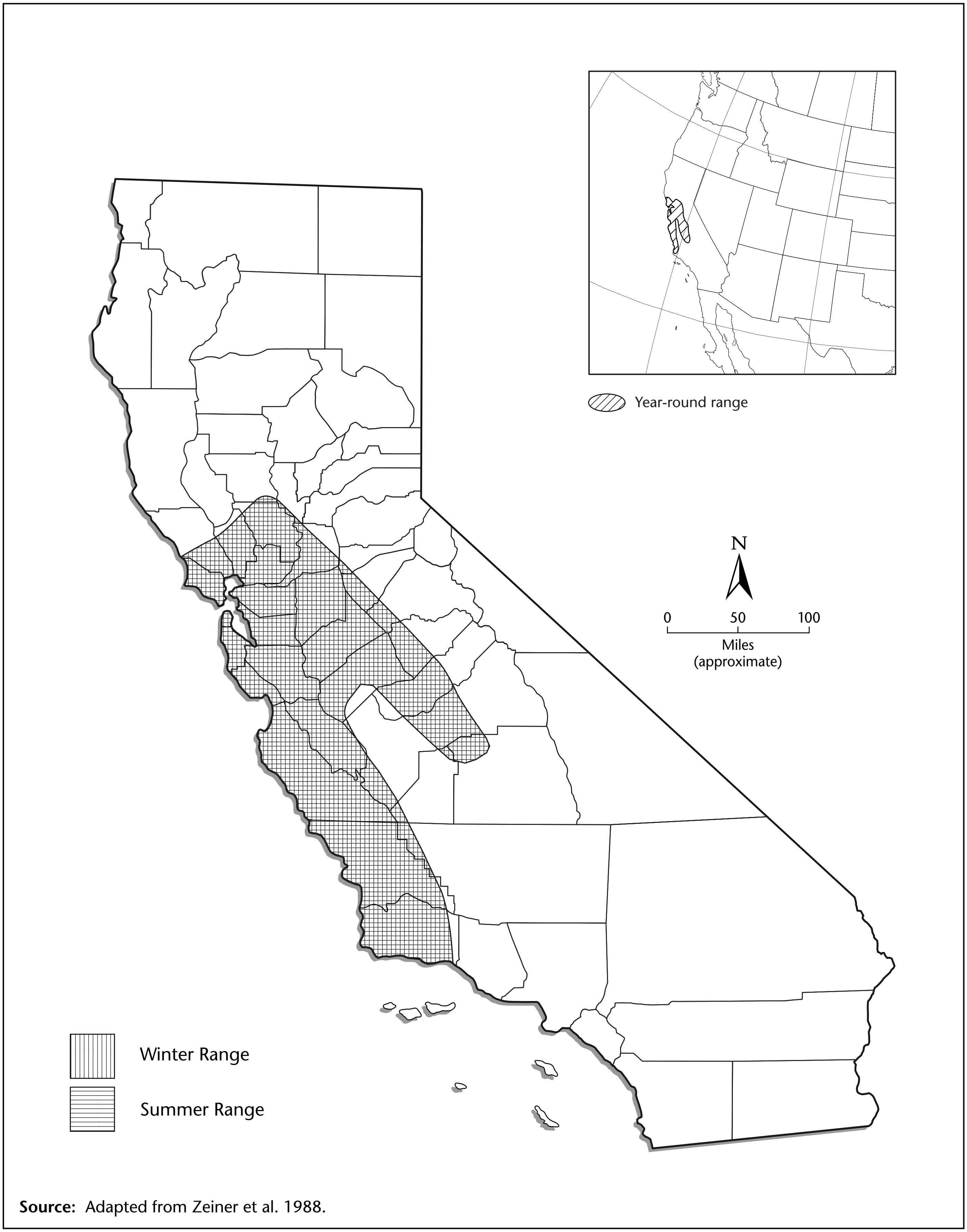

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Source: Adapted from Zeiner et al. 1988.

Jones & Stokes

California Tiger Salamander *(Ambystoma californiense)* Distribution

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

Status

State: Meets requirements as a "rare, threatened or endangered species" under CEQA

Federal: Threatened

Population Trend

Global: State endemic; declining

State: Declining

Within Inventory Area: Apparently stable in some areas

Data Characterization

The location database for the California red-legged frog (*Rana aurora draytonii*) within its known range in California includes 419 data records dated from 1919 to 2001. Of these records, 344 were documented within the past 10 years; of these, 203 are of high precision and may be accurately located within the inventory area. Approximately 81 of these high-precision records are located within or near the inventory area. These records occur within non-native grassland, riparian forest, riparian woodland, riparian scrub, freshwater marsh, and wetland.

A moderate amount of literature is available regarding the California red-legged frog because of its threatened status and the recent trend in global decline in amphibians. Most of the literature pertains to habitat requirements, population trends, ecological relationships, threats, and conservation efforts. A draft recovery plan for the California red-legged frog has been published by the U.S. Fish and Wildlife Service (2000).

Range

The historical range of the California red-legged frog extended along the coast from the vicinity of Point Reyes National Seashore, Marin County, California and inland from Redding, Shasta County southward to northwestern Baja California, Mexico (Jennings and Hayes 1985, Hayes and Krempels 1986). The current distribution of this species includes only isolated localities in the Sierra Nevada, northern Coast and Northern Traverse Ranges. It is still common in the San Francisco Bay area and along the central coast. It is now believed to be extirpated from the southern Traverse and Peninsular ranges (U.S. Fish and Wildlife Service 2000).

Occurrences within the ECC HCP/NCCP Inventory Area

Contra Costa and Alameda Counties contain the majority of known California red-legged frog occurrences in the San Francisco Bay Area (U.S. Fish and

Wildlife Service 2000). However, this species seems to have been nearly eliminated from the western lowland portions of these counties, particularly near urbanization. Eighty-one occurrences of California red-legged frogs have been documented within the inventory area (California Natural Diversity Database 2001). Numerous ponds and creeks in Simas Valley support California red-legged frogs (Dunne 1995). Sizeable breeding populations are also found at Sand Creek (Black Diamond Mines Regional Park) and Round Valley (Round Valley Regional Preserve) (S. Bobzien in litt. 1900 cited in U.S. Fish and Wildlife Service 2000). Some of the highest densities of California red-legged frog occur in many of the stock ponds within the Los Vaqueros watershed.

Biology

Habitat

Within their range, California red-legged frogs occur from sea level to about 5,000 feet above sea level (U.S. Fish and Wildlife Service 2000). Almost all of the documented occurrences of this species, however, are located below 3,500 feet. Breeding sites include a variety of aquatic habitats—larvae, tadpoles and metamorphs use streams, deep pools, backwaters within streams and creeks, ponds, marshes, sag ponds, dune ponds, and lagoons. Breeding adults are commonly found in deep (more than 2 feet), still or slow-moving water with dense, shrubby riparian or emergent vegetation (Hayes and Jennings 1988). Adult frogs have also been observed in shallow sections of streams that are not shrouded by riparian vegetation. Generally, streams with high flows and cold temperatures in spring are unsuitable for eggs and tadpoles. Stock ponds are frequently used by this species if they are managed to provide suitable hydroperiod, pond structure, vegetative cover, and control of nonnative predators.

During dry periods, California red-legged frogs are seldom found far from water. However, during wet weather, individuals may make overland excursions through upland habitats over distances up to 2 miles. These dispersal movements are generally straight-line, point-to-point migrations rather than following specific habitat corridors. Dispersal distances are believed to depend on the availability of suitable habitat and prevailing environmental conditions. Very little is known about how California red-legged frogs use upland habitats during these periods.

During summer, California red-legged frogs often disperse from their breeding habitat to forage and seek summer habitat if water is not available (U.S. Fish and Wildlife Service 2000). This habitat may include shelter under boulders, rocks, logs, industrial debris, agricultural drains, watering troughs, abandoned sheds, or hay-ricks. They will also use small mammal burrows, incised streamed channels, or areas with moist leaf litter (Jennings and Hayes 1994; U.S. Fish and Wildlife

Service 1996, 2000). This summer movement behavior, however, has not been observed in all California red-legged frog populations studied.

Foraging Requirements

California red-legged frogs consume a wide variety of prey. Adult frogs typically feed on aquatic and terrestrial insects, crustaceans and snails (Stebbins 1985, Hayes and Tennant 1985), as well as worms, fish, tadpoles, smaller frogs (e.g. *Hyla regilla*), and occasionally mice (*Peromyscus californicus*) (U.S. Fish and Wildlife Service 2000). Aquatic larvae are mostly herbivorous algae grazers (Jennings et al. in litt. 1992). Feeding generally occurs along the shoreline of ponds or other watercourses and on the water surface. Juveniles appear to forage during both daytime and nighttime, whereas subadults and adults tend to feed more exclusively at night (Hayes and Tennant 1985).

Reproduction

California red-legged frogs breed from November through April (Storer 1925, U.S. Fish and Wildlife Service 2000). Males usually appear at the breeding sites 2 to 4 weeks before females. Females are attracted to calling males. Females lay egg masses containing about 2,000 to 5,000 eggs, which hatch in 6 to 14 days, depending on water temperatures (U.S. Fish and Wildlife Service 2000). Larvae metamorphose in 3.5 to 7 months, typically between July and September (Storer 1925, Wright and Wright 1949, U.S. Fish and Wildlife Service 2000). Sexual maturity is usually attained by males at 2 years of age and females at 3 years of age.

Demography

Adult California red-legged frogs can live 8 to 10 years (Jennings et al. 1992), but the average life span is probably much lower (Scott pers. comm. in U.S. Fish and Wildlife Service 2000). Most mortality occurs during the tadpole stage (Licht 1974). No long-term studies have been conducted on the population dynamics of red-legged frogs.

Ecological Relationships

California red-legged frogs are primary, secondary, and tertiary consumers in the aquatic/terrestrial food web of their habitat. As described above, they prey on a wide variety of invertebrates and vertebrates, as well as algae as larvae. Numerous predators prey on these frogs, including racoons *(Procyon lotor*), great blue herons (*Ardea herodias*), American bitterns (*Botaurus lentiginosus*), black-crowned night herons (*Nycticorax nycticorax*), red-shouldered hawks (*Buteo* *lineatus*), opossums (*Didpephis virginiana*), striped skunks (*Mehpitis mephitis*), spotted skunks (*Spilogale pituorius*), and garter snakes (*Thamnophis* spp.) (Fitch 1940, Fox 1952, Jennings and Hayes 1990, Rathbun and Murphy 1996). In some areas, introduced aquatic vertebrates and invertebrates also prey on one or more of the life stages of California red-legged frogs. These predators include bullfrogs (*Rana catesteiana*), African clawed frogs (*Xenopus laevis*), red swamp crayfish (*Procambarus clarkii*), signal crayfish (*Pacifastacus leniusculus*), bass (*Micropterus* spp.), catfish (*Ictalurus* spp.), sunfish (*Lepomis* spp.), and mosquitofish (*Gambusia affinis*) (Hayes and Jennings 1986).

Threats and Reasons for Decline

The viability of existing California red-legged frog populations is threatened by numerous human activities that often act synergistically and cumulatively with natural disturbances (i.e. droughts or floods) (U.S. Fish and Wildlife Service 2000). These activities include those that result in the degradation, fragmentation, and loss of habitat through agriculture, urbanization, mining, overgrazing, recreation, timber harvesting, nonnative plants, impoundments, water diversions, degraded water quality, and introduced predators.

Over 90% of the historic wetlands in the Central Valley have been lost due to conversion for agriculture or urban development (U.S. Fish and Wildlife Service 1978, Dahl 1990). This has resulted in a significant loss of frog habitat throughout the species' range (U.S. Fish and Wildlife Service 2000). Habitat along many stream courses has also been isolated and fragmented, resulting in reduced connectivity between populations and lowered dispersal opportunities. These isolated populations are now more vulnerable to extinction through stochastic environmental events (i.e. drought, floods) and human-caused impacts (i.e., grazing disturbance, contaminant spills) (Soulé 1998). In a comprehensive evaluation of prevailing hypotheses on the causes of declines in the California red-legged frog populations, Davidson et al. (2001) determined that there is a strong statistical correlation between locations where frog numbers had declined and upwind agricultural land use. They concluded that wind-borne agrochemicals may be an important factor in these declines.

Increasing urbanization in the Central Valley is also resulting in the continuing loss and fragmentation of California red-legged frog habitat and creates barriers to dispersal by frogs to neighboring populations. Isolated populations are subject to increased predation from nonnative predators, changes in hydroperiod due to variable wastewater outflows, and increased potential for toxic runoff from developments. All of these conditions can reduce the viability of affected frog populations. Similarly, agricultural expansion in the Central Valley has resulted in habitat loss and fragmentation, the introduction of fertilizers, fungicides, pesticides, and herbicides into riparian ecosystesm and water diversions and impoundments that can reduce historic flows necessary to support adequate aquatic habitat for frogs and other species (U.S. Fish and Wildlife Service 2000). Poorly managed recreation, mining, timber harvest, and infrastructure maintenance activities, such as road construction and repair, trail development and facilities development, can also have significant detrimental effects on remaining California red-legged habitat through disturbance, contamination, and introduction of nonnative species that prey on or compete with the frogs.

Conservation and Management

The California red-legged frog was federally listed as threatened in 1996. Since then, numerous conservation efforts have been undertaken by various federal, state, and local and private organizations to minimize impacts and establish preserves and protective policies to ensure the viability of this species (U.S. Fish and Wildlife Service 2000). A draft recovery plan for the California red-legged frog was completed in January 2000 that calls for the preservation of all known populations and their habitat, the establishment of a viable metapopulation, development of effective land use policies and guidelines, continued research on the ecological requirements of California red-legged frogs necessary for conservation, continued monitoring, and the establishment of an outreach program.

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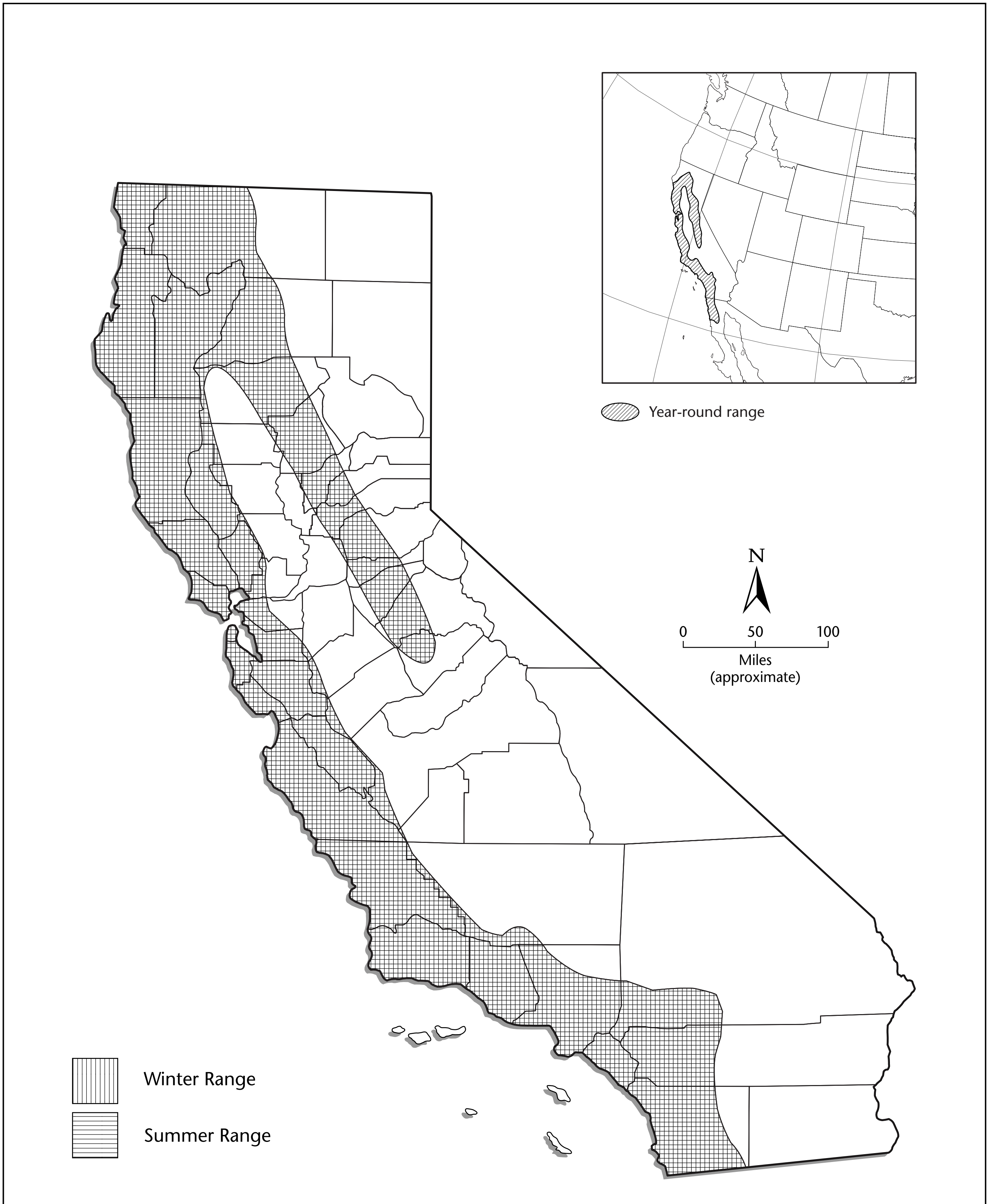

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Source: Adapted from Zeiner et al. 1988.

Jones & Stokes

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

Foothill Yellow-Legged Frog (*Rana boylii*)

Status

State: Species of Concern
Federal: Species of Concern

Population Trend

Global: Declining
State: Declining
Within Inventory Area: Unknown

Data Characterization

The location database for the foothill yellow-legged frog (*Rana boylii)* within its known range in California includes 288 occurrence records dated from 1958 to 2001. None was documented for the inventory area, but Jennings and Hayes (1994) show 11 occurrence records of foothill yellow-legged frog in Contra Costa County. Eight of these populations are believed to be extinct. The 3 remaining records are concentrated in the Mount Diablo region.

A moderate amount of literature is available for the foothill yellow-legged frog because of its local availability for study and the recent trend in global decline in amphibians. Most of the literature pertains to habitat requirements, population trends, ecological relationships, threats, and conservation efforts.

Range

The species is historically known from the Santiam River system in Oregon to the San Gabriel River system in California (Storer 1923, 1925; Fitch 1938; Marr 1943; Zweifel 1955). Its known elevation range extends from near sea level to approximately 2,040 meters above sea level (Stebbins 1985).

Occurrences within the ECCC HCP/NCCP Inventory Area

Foothill yellow-legged frogs occur in numerous perennial streams throughout the inventory area. As described above, there 11 documented occurrence records of foothill yellow-legged frog in Contra Costa County—8 believed to be extinct and 3 concentrated in the Mount Diablo region.

Biology

Habitat

Foothill yellow-legged frogs require shallow, flowing water in small to moderate-sized streams with at least some cobble-sized substrate (Hayes and Jennings 1988, Jennings 1988). This habitat is believed to favor oviposition (Storer 1925, Fitch 1936, Zweifel 1955) and refuge habitat for larvae and postmetamorphs (Hayes and Jennings 1988, Jennings 1988). This species has been found in streams without cobble (Fitch 1938, Zweifel 1955), but it is not clear whether these habitats are regularly used (Hayes and Jennings 1988, Jennings and Hayes 1994). Foothill yellow-legged frogs are usually absent from habitats where introduced aquatic predators, such as various fishes and bullfrogs, are present (Hayes and Jennings 1986, 1988; Kupferberg 1994). The species deposits its egg masses on the downstream side of cobbles and boulders over which a relatively thin, gentle flow of water exists (Storer 1925, Fitch 1936, Zweifel 1955). The timing of oviposition typically follows the period of high flow discharge from winter rainfall and snowmelt (Jennings and Hayes 1994). The embryos have a critical thermal maximum temperature or 26ºC (Zweifel 1955).

Foraging Requirements

Adult foothill yellow-legged frogs feed primarily on both aquatic and terrestrial insects (Fitch 1936); tadpoles preferentially graze on algae (Jennings and Hayes 1994). Postmetamorphs eat aquatic and terrestrial insects (Storer 1925, Fitch 1936).

Reproduction

Foothill yellow-legged frogs in California generally breed between March and early June (Storer 1925, Grinnell et al. 1930, Wright and Wright 1949, Jennings and Hayes 1994). Masses of eggs are deposited on the downstream side of cobbles and boulders. After oviposition, a minimum of approximately 15 weeks is required to reach metamorphosis, which typically occurs between July and September (Storer 1925, Jennings 1988). Larvae attain adult size in 2 years (Storer 1925).

Demography

Masses of 300 to 1,200 eggs are deposited during oviposition by each breeding female. Juvenile and adult survivorship is unknown. Adult longevity is unknown.

Ecological Relationships

Garter snakes are considered one of the most prominent predators of foothill yellow-legged frog tadpoles (Fitch 1941, Zweifel 1955, Lind 1990, Jennings and Hayes 1994). Salamanders, including the rough-skinned newt (*Taricha tarosa*), are believed to prey on the species' eggs.

Threats

Habitat loss and degradation, introduction of exotic predators, and toxic chemicals (including pesticides) pose continued and increasing threats to the long-term viability amphibians throughout California (Jennings and Hayes 1994). In addition, poorly timed water releases from upstream reservoirs can scour egg masses of this species from their oviposition substrates (Jennings and Hayes 1994), and decreased flows can force adult frogs to move into permanent pools, where they may be more susceptible to predation (Hayes and Jennings 1988)

Conservation and Management

The principal conservation measures necessary for maintaining viable populations of this species include habitat preservation, restoration, and management to retain ecological conditions necessary for survival and population growth. However, information on the range of ecological conditions that can be tolerated by this species is limited. Studies on the habitat requirements of the foothill yellow-legged frog larvae and early postmetamorphic states are urgently needed (Jennings and Hayes 1994). An understanding of the variation in flow and shear conditions that egg masses and larvae will tolerate is needed, as well as a more precise understanding of the critical thermal maxima of the embryonic stages (Jennings and Hayes 1994). In managed streams, Jennings and Hayes (1994) recommend avoiding water releases that create excess flow and shear conditions when egg masses and the more-fragile younger larval stages are present.

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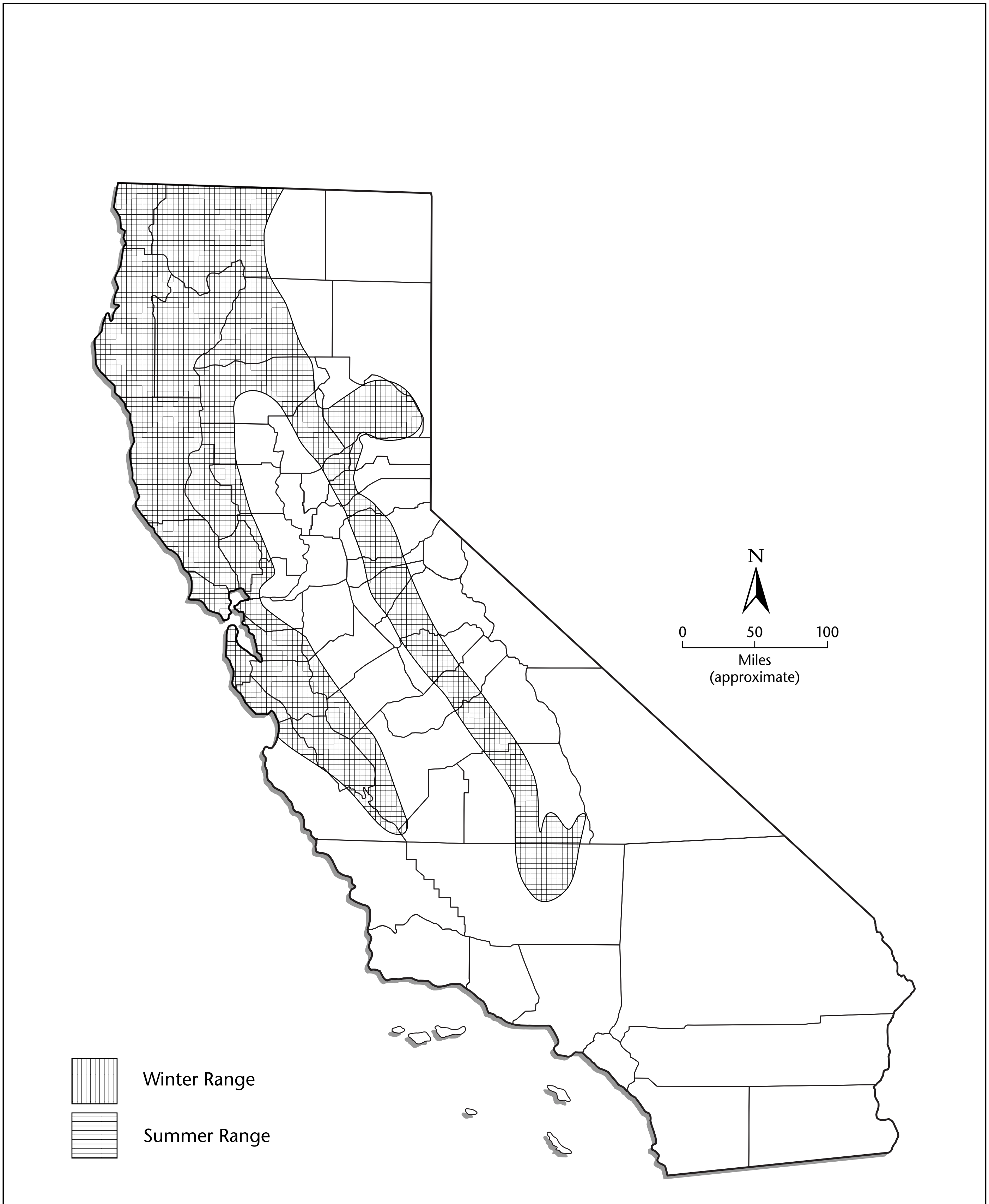

Source: Adapted from Zeiner et al. 1984.

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Jones & Stokes

Foothill Yellow-Legged Frog *(Rana boylii)* Distribution

Vernal Pool Fairy Shrimp (*Brachinecta lynchi*)

Status

State: Meets the requirements as a "rare, threatened, or endangered species" under CEQA

Federal: Threatened

Population Trend

Global: Declining due to habitat loss and fragmentation (Eriksen and Belk 1999)

State: As above

Within Inventory Area: Unknown

Data Characterization

The location database for the vernal pool fairy shrimp (*Brachinecta lynchi)* within the inventory area includes 6 records from 1993, 1997, and 1999 from within the ECCC HCP/NCCP inventory area. The majority of locations are vernal pools within non-native grassland. Other natural and artificial habitats have a high probability of being occupied by additional populations of the vernal pool fairy shrimp throughout the grassland habitats within the ECCC HCP/NCCP inventory area.

Beyond the original description (Eng et al. 1990), a scanning electron micrograph of the cyst (resting egg) (Hill and Shepard 1997), and some generalized natural history data (Helm 1997), no peer-reviewed technical literature has been published concerning the vernal pool fairy shrimp. Eriksen and Belk (1999) presented a brief discussion of the vernal pool fairy shrimp and provided a distribution map.

Range

The vernal pool fairy shrimp is found from Jackson County near Medford, Oregon, throughout the Central Valley, and west to the central Coast Ranges. Isolated southern populations occur on the Santa Rosa Plateau and near Rancho California in Riverside County (Eng et al.1990, Eriksen and Belk 1999, Jones & Stokes file information). In 1996, the U.S. Fish and Wildlife Service reported that there were 32 known populations of the vernal pool fairy shrimp, ranging from the Stillwater Plain in Shasta County through most of the length of the Central Valley to Paisley in Tulare County, and along the central Coast Range from northern Solano County to Pinnacles National Monument in San Benito County. Disjunct populations were also reported to occur in San Luis Obispo County, Santa Barbara County, and Riverside County.

Vernal pool fairy shrimp have been observed in the western portions (Central Valley region) of Tehama, Butte, Yuba, Placer, Stanislaus, Madera, Fresno, and Tulare Counties (Eriksen and Belk 1999). This species has also been observed in the eastern portions of Alameda, Yolo, and Glenn Counties (Eriksen and Belk 1999). It has been observed in Sacramento, Colusa, and Merced Counties as well.

Occurrences within the ECCC HCP/NCCP Inventory Area

Six records for this species exist in the ECCC HCP/NCCP inventory area. Vernal pool fairy shrimp may also be found elsewhere throughout the inventory area in appropriate habitats. The paucity of data points within the open space areas is due to a lack of survey effort.

Existing vernal pool fairy shrimp records include numerous occupied pools on the Cowell Ranch on the northeast side of Mount Diablo, artificial pools in a railroad access road near Pittsburgh, and pools in the Byron Hot Springs area.

Biology

Habitat

Typical habitat for special-status fairy shrimp in California include vernal pools, seasonally ponded areas within vernal swales, rock outcrop ephemeral pools, playas, and alkali flats (Eng et al. 1990). Other kinds of depressions that hold water of a similar volume, depth, and area, and for a similar duration and seasonality as vernal pools and ponded areas within swales also may be potential habitat. These other depressions, however, are typically artificial habitats and partially or completely unvegetated. Examples are railroad toe-drains, roadside ditches, abandoned agricultural drains, ruts left by heavy construction vehicles, and depressions in firebreaks (Eng et al. 1990).

Vernal pool fairy shrimp have also been found in water pooled in sandstone outcrops and in alkaline vernal pools. Vernal pools that support these fairy shrimp are often grass or mud bottomed, with clear to tea-colored water, and are often found in basalt-flow depression pools on unplowed grasslands (U.S. Fish and Wildlife Service 1994). Vernal pools are subject to seasonal variations, and vernal pool fairy shrimp are dependent on the ecological characteristics of such variations. These characteristics include duration of inundation and presence or absence of water at specific times of the year (U.S. Fish and Wildlife Service 1994). The vernal pool fairy shrimp is capable of living in Central Valley vernal pools of relatively short duration (pond 6 to 7 weeks in winter and 3 weeks in spring) (Eriksen and Belk 1999). Other factors contributing to the suitability of pools for vernal pool fairy shrimp include alkalinity, total dissolved solids (TDS), and pH (U.S. Fish and Wildlife Service 1994; Eriksen and Belk 1999). This fairy shrimp occurs in pools with alkalinity ranging from 22 to 274 ppm (parts per million), 48 to 481 ppm TDS, and a pH range from 6.3 to 8.5 (Eriksen and Belk 1999). U.S. Fish and Wildlife Service (1994) described the water in pools occupied by vernal pool fairy shrimp as having low conductivity and chloride, though specific numbers were not given. Vernal pool fairy shrimp have been found in pools ranging from 0.1 acre to 0.05 acre (Eriksen and Belk 1999). However, Platenkamp (1998) found that at Beale Air Force Base in Yuba County vernal pool fairy shrimp occurred more frequently in small, deep pools. Specific descriptions of the size and depth of occupied vernal pools were not reported in this paper

Feeding

Vernal pool fairy shrimp are omnivorous filter-feeders. Fairy shrimp indiscriminately filter particles from the surrounding water, including bacteria, unicellular algae, and micrometazoa (Eriksen and Belk 1999). The precise size of items these fairy shrimp are capable of filtering is currently unknown. However, fairy shrimp will attempt to consume whatever material they can fit into their feeding groove and do not discriminate based upon taste, as do some other crustacean groups (Eriksen and Belk 1999). Vernal pool fairy shrimp will also rasp periphyton from sticks, stems, and slender leaves (Rogers in prep.).

Ecology

Vernal pool fairy shrimp are a component of the planktonic crustacea within seasonal temporary pools and can occur in densities as high as 200 per liter of water. Planktonic crustacea are important in the food web, as they represent a high-fat, high-protein resource for migratory waterfowl. Mallard, Green-winged Teal, Bufflehead, Greater Yellowlegs, and Killdeer all forage actively in Central Valley vernal pools on the invertebrate and amphibian fauna during the winter months.

Predator consumption of fairy shrimp cysts (resting eggs) aids in distributing populations of fairy shrimp. Predators expel viable cysts in their excrement, often at locations other than where they were consumed (e.g. Wissinger et al. 1999). If conditions are suitable, these transported cysts may hatch at the new location and potentially establish a new population. Cysts can also be transported in mud carried on the feet of animals, including livestock, that may wade through the habitat (Rogers in prep.).

Beyond inundation of the habitat, the specific cues for hatching are unknown (Eriksen and Belk 1999), although temperature is believed to play a large role. Typically, midvalley fairy shrimp mature in about 16 days when water temperatures reach at least 20 degrees Celsius (Eriksen and Belk 1999).

Vernal pool fairy shrimp commonly co-occur with the California linderiella *(Linderiella occidentalis*) (Eriksen and Belk 1999). This species has also been reported co-occurring with the midvalley pool fairy shrimp (*Branchinecta mesovallensis*) on 3 occasions, in which the midvalley fairy shrimp was probably washed into the vernal pool fairy shrimp habitat by abnormally high rainfall (Eriksen and Belk 1999).

Threats

Vernal pool fairy shrimp are threatened by the same activities as other vernal pool invertebrates. These threats include the conversion of vernal pool habitat to agricultural lands and urban development, and stochastic extinction because of the small and isolated nature of remaining populations (U.S. Fish and Wildlife Service 1994). The limited and disjunct distribution of vernal pools, coupled with the even more limited distribution of the vernal pool fairy shrimp, means that any reduction in vernal pool habitat quantity could adversely affect this species.

Habitat fragmentation can isolate and reduce population size, resulting in a process of progressive population extinction. Small or isolated populations are more susceptible to extinction from random environmental disturbance. Recolonization opportunities are also diminished when physical barriers, such as development or lack of vernal pool habitat, isolate populations from one another or inhibit transport of cysts. Isolated populations are potentially more susceptible to inbreeding depression, which can result in local extinction or reduced fitness (Gilpin and Soule 1986, Goodman 1987a, 1987b). However, this has never been demonstrated for branchiopod crustaceans.

Activities that alter the suitability of habitat may impact the special-status crustaceans dependent on these habitats. These activities include damaging the impermeable clay and /or hardpan layers of the habitat bottom, filling in the habitat, and altering (e.g. through contaminants) or destroying the watershed that conveys overland flow into the habitat. Additionally, introduction of non-native plants, destruction or degradation of the surrounding upland habitat, introduction of fish (such as *Gambusia* spp.) into special-status shrimp habitats, and activities that would discourage or prevent waterfowl and waders from feeding at occupied habitats and thereby restrict gene flow between populations would also significantly affect midvalley fairy shrimp populations.

Conservation and Management

The conservation of vernal pool fairy shrimp is directly tied to the conservation of suitable vernal pool habitat. However, because comprehensive surveys for the vernal pool fairy shrimp in the ECCC HCP/NCCP inventory area have not been conducted and because known occurrences throughout the species range are based mostly on incidental observations (e.g., CNDDB), the population size and locations of this species in the ECCC HCP/NCCP inventory area are not known. Also, suitable habitat for the vernal pool fairy shrimp in the ECCC HCP/NCCP inventory area was identified based on a general classification of land cover types. Field evaluation of the habitat classification has not been conducted, and the extent to which vernal pools in the inventory area meet the habitat requirements of vernal pool fairy shrimp is unknown. Also, the importance of artificial habitats that may support vernal pool fairy shrimp in the ECCC HCP/NCCP inventory area has not been evaluated.

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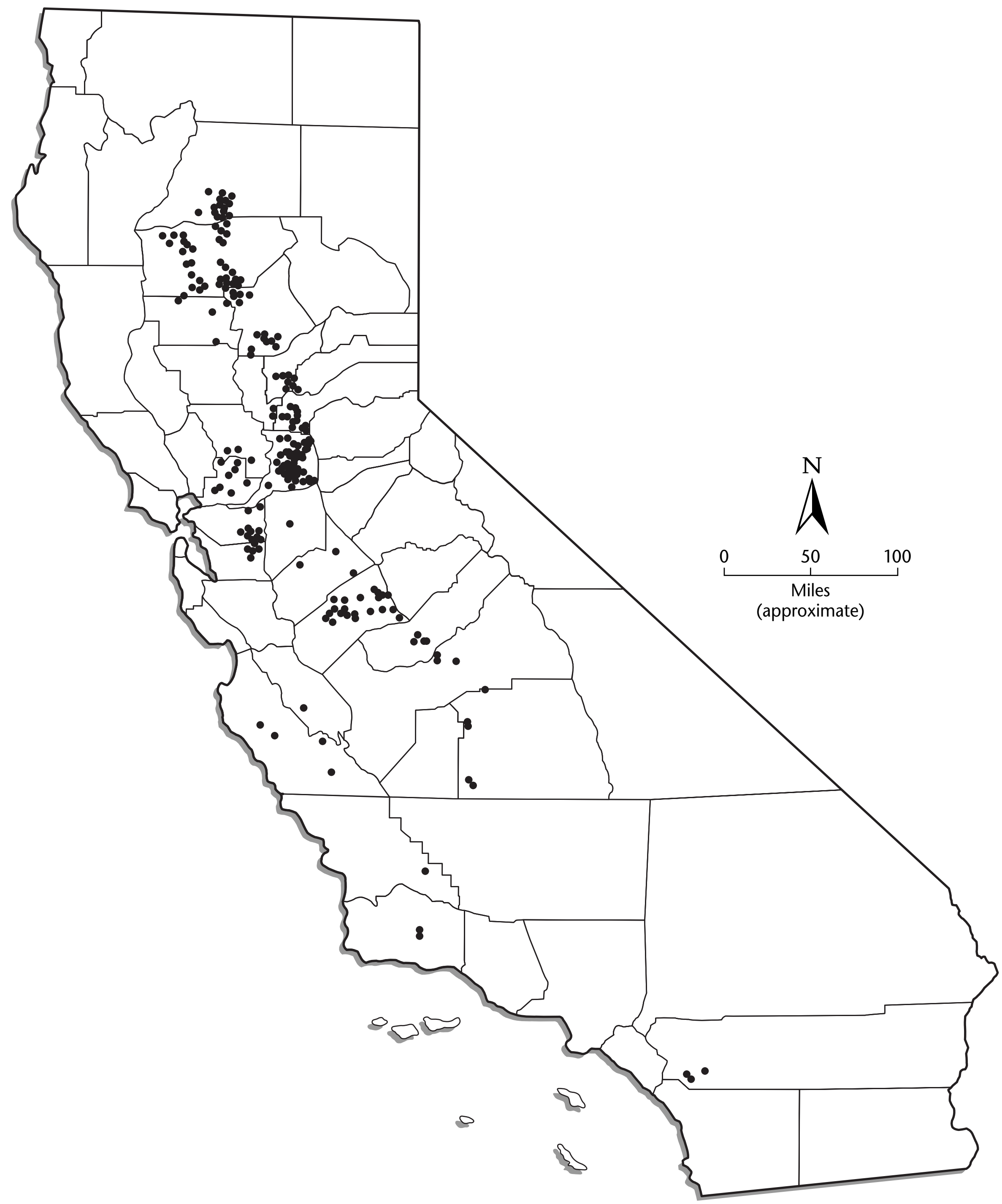

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Source: Erickson and Belk 1999.

Jones & Stokes

Vernal Pool Fairy Shrimp *(Branchinecta lynchi)*
Known Occurrences

Longhorn Fairy Shrimp (*Branchinecta longiantenna*)

Status

State: Meets the requirements as a "rare, threatened, or endangered species" under CEQA
Federal: Endangered

Population Trend

Global: Declining due to habitat loss and fragmentation (Eriksen and Belk 1999)
State: As above
Within Inventory Area: Unknown

Data Characterization

The location database for the longhorn fairy shrimp (*Branchinecta longiantenna)* within the inventory area includes 2 records from 1982 and 1990 near the Los Vaqueros Reservoir (Eng et. al. 1990, California Natural Diversity Database 2001). These 2 locations are shallow sandstone rock outcrop vernal pools within non-native grassland. Other natural and artificial habitats have a high probability of being occupied by additional populations of the longhorn fairy shrimp throughout the grassland habitats within the ECCC HCP/NCCP inventory area.

In addition to the original description (Eng et. al. 1990), Eriksen and Belk (1999) presented a brief discussion of the longhorn fairy shrimp and provided a distribution map. Hill and Shepard (1997) produced a scanning electron micrograph of the cyst (resting egg), and Helm (1997) provided some generalized natural history data. No other peer-reviewed technical literature has been published concerning the longhorn fairy shrimp.

Range

Only 8 populations of the longhorn fairy shrimp are known (U.S. Fish and Wildlife Service 1996). The distribution of the longhorn fairy shrimp is limited to rock outcrop pools in the central Coast Ranges of Contra Costa and Alameda Counties, alkaline pools in San Luis Obispo County, and grassy-bottomed pools in Madera County (Eng et al.1990, Eriksen and Belk 1999, Jones & Stokes file information).

Occurrences within the ECCC HCP/NCCP Inventory Area

Longhorn fairy shrimp potentially may be found throughout the inventory area in appropriate habitats. Two records for this species exist in the ECCC HCP/NCCP inventory area: the Souza Ranch (type locality), and a rock outcrop at Los Vaqueros. The paucity of data points within the inventory area is likely due to a lack of survey effort.

Biology

Habitat

Typical habitat for special-status fairy shrimp in California include vernal pools, seasonally ponded areas within vernal swales, ephemeral pools in rock outcrops, playas, and alkali flats (Eng et al. 1990). Other kinds of depressions that hold water of a similar volume, depth, and area, and for a similar duration and seasonality as vernal pools and ponded areas within swales also may be potential habitat. These other depressions--typically artificial habitats and partially or completely unvegetated--may be suitable for this species. Examples of artificial habitats that may be suitable for this species are railroad toe-drains, roadside ditches, abandoned agricultural drains, ruts left by heavy construction vehicles, and depressions in firebreaks (Eng et al. 1990).

Longhorn fairy shrimp in Contra Costa and Alameda Counties are primarily reported from water pooled in sandstone depressions. Vernal pools in other parts of California that support these fairy shrimp are either alkaline pools or grass bottomed, with clear to tea-colored water (U.S. Fish and Wildlife Service 1994). The seasonal pool habitat is subject to seasonal variations, and longhorn fairy shrimp are dependent on the ecological characteristics of such variations. These characteristics include duration of inundation and presence or absence of water at specific times of the year (U.S. Fish and Wildlife Service 1994). The longhorn fairy shrimp is capable of living in vernal pools of relatively short duration (pond 6 to 7 weeks in winter and 3 weeks in spring) (Eriksen and Belk 1999).

Feeding

Longhorn fairy shrimp are omnivorous filter-feeders. Fairy shrimp indiscriminately filter particles from the surrounding water, including bacteria, unicellular algae, and micrometazoa (Eriksen and Belk 1999). The precise size of items these fairy shrimp are capable of filtering is currently unknown. However, fairy shrimp will attempt to consume whatever material they can fit into their feeding groove and do not discriminate based upon taste, as do some other crustacean groups (Eriksen and Belk 1999).

Ecology

Longhorn fairy shrimp are a component of the planktonic crustacea within seasonal temporary pools and can occur in densities as high as 200 per liter of water. Planktonic crustacea are important in the food web, as they represent a high-fat, high-protein resource for migratory waterfowl. Mallard, Green-winged Teal, Greater Yellowlegs, and Killdeer all forage actively in Central Valley vernal pools on the invertebrate and amphibian fauna during the winter months.

Predator consumption of fairy shrimp cysts (resting eggs) aids in distributing populations of fairy shrimp. Predators expel viable cysts in their excrement, often at locations other than where they were consumed (e.g., Wissinger et al. 1999). If conditions are suitable, these transported cysts may hatch at the new location and potentially establish a new population. Cysts can also be transported in mud carried on the feet of animals, including livestock, that may wade through their habitat (Rogers, unpublished data).

Beyond inundation of the habitat, the specific cues for hatching are unknown (Eriksen and Belk 1999), although temperature is believed to play a large role. Typically, midvalley fairy shrimp mature in about 16 days when water temperatures reach at least 20 degrees Celsius (Eriksen and Belk 1999). Longhorn fairy shrimp have been reported to co-occur with the vernal pool fairy shrimp (*Branchinecta lynchi*).

Threats

Longhorn fairy shrimp are threatened by the same activities as other vernal pool invertebrates. These threats include the conversion of vernal pool habitat to agricultural lands and urban development, and extinction because of the small and isolated nature of remaining populations (U.S. Fish and Wildlife Service 1994). The limited and disjunct distribution of vernal pools, coupled with the even more limited distribution of the longhorn fairy shrimp, means that any reduction in vernal pool habitat could adversely affect this species.

Habitat fragmentation can isolate and reduce population size, resulting in a process of progressive population extinction. Small or isolated populations are more susceptible to extinction from random environmental disturbance. Recolonization opportunities are also diminished when physical barriers, such as development or lack of vernal pool habitat, isolate populations from one another or inhibit transport of cysts. Isolated populations are potentially more susceptible to inbreeding depression, which can result in local extinction or reduced fitness (Gilpin and Soule 1986, Goodman 1987a, 1987b). However, this has never been demonstrated for branchiopod crustaceans.

Activities that alter the suitability of habitat may impact the special-status crustaceans dependent on these habitats. These activities include damaging the impermeable clay and /or hardpan layers of the habitat bottom, filling in the habitat, altering (e.g. through contaminants) or destroying the watershed that conveys overland flow into the habitat. Additionally, introduction of non-native plants, destruction or degradation of the surrounding upland habitat, introduction of fish (such as *Gambusia* spp.) into special-status shrimp habitats, and activities that would discourage or prevent waterfowl and waders from feeding at occupied habitats and thereby restrict gene-flow between populations would also significantly affect longhorn fairy shrimp populations.

Conservation and Management

Because comprehensive surveys for the longhorn fairy shrimp in the ECCC HCP/NCCP inventory area have not been conducted and known occurrences throughout the species range are based mostly on incidental observations (e.g., CNDDB), the population size and locations of this species in the ECCC HCP/NCCP inventory area are not known. Also, suitable habitat for the longhorn fairy shrimp in the ECCC HCP/NCCP inventory area was identified based on a general classification of land-cover types. Field evaluation of the habitat classification has not been conducted, and the extent to which vernal pools in the inventory area meet the habitat requirements of longhorn fairy shrimp is unknown. Also, the importance of artificial habitats that may support longhorn fairy shrimp in the ECCC HCP/NCCP inventory area has not been evaluated.

However, the dependency of this species on vernal pool habitats provides some useful information on the types of impacts that can occur to longhorn fairy shrimp from covered activities. Based on its restricted distribution, the current USFWS conservation requirement for this species is that no take (individuals or habitat) will be allowed.

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Source: Erickson and Belk 1999.

Jones & Stokes

Longhorn Fairy Shrimp *(Branchinecta longiantenna)*
Known Occurrences

Midvalley Fairy Shrimp (*Branchinecta mesovallensis*)

Status

State: Meets the requirements as a “rare, threatened or endangered species” under CEQA

Federal: None; petitioned for endangered status

Population Trend

Global: Declining due to habitat loss and fragmentation (Center for Biological Diversity 2001, Eriksen and Belk 1999, Belk & Fugate 2000)

State: As above

Within Inventory Area: Unknown

Data Characterization

The location database for the midvalley fairy shrimp (*Branchinecta mesovallensis)* within the study area includes a single data record from 1997 near the Byron Airport and can be accurately located within the inventory area. The single location is a shallow vernal pool within nonnative grassland. Additional natural and artificial habitats have a high probability of being occupied by the midvalley fairy shrimp throughout the grassland habitats within the inventory area.

Except for the original description (Belk and Fugate 2000), a scanning electron micrograph of the cyst (resting egg) (Hill and Shepard 1997), and over-generalized natural history data (Helm 1997), no peer-reviewed technical literature has been published concerning the midvalley fairy shrimp. However, a U.S. Fish and Wildlife Service study is currently in progress, and the data from that study is available. In addition, Eriksen and Belk (1999) have presented a brief discussion of the midvalley fairy shrimp and provided a distribution map.

Range

Midvalley fairy shrimp is endemic to California Central Valley grassland vernal pools (Belk and Fugate 2000). Known occurrences include scattered populations from the Mather Field area of Sacramento south through Galt from Sacramento County; the Jepson Prairie, Travis Air Force Base, and Vacaville areas in Solano County; from Lodi north to the county border in San Joaquin County; the Byron Airport in Contra Costa County; the Virginia Smith Trust (Haystack Mountain) and Arena Plains National Wildlife Reserve in Merced County; 1 location in central Madera County; and 1 in northern Fresno County (Erickson and Belk 1999, Belk and Fugate 2000, Rogers in prep.).

Occurrences within the ECCC HCP/NCCP Inventory Area

Midvalley fairy shrimp could be found throughout the inventory area in appropriate habitats. A single record for this species exists near the Byron Airport. The paucity of data points within open space areas is due to a lack of survey effort. Because this species has a brief life cycle and inhabits shallow temporary pools and artificial habitats that may only pond between 4 and 14 days, it is very likely that this species would be missed during typical U.S. Fish and Wildlife Service protocol-level surveys (U.S. Fish and Wildlife Service 1996).

Biology

Habitat

Typical habitat for special-status fairy shrimp in California includes vernal pools, seasonally ponded areas within vernal swales, rock outcrop ephemeral pools, playas, and alkali flats (Eng et al. 1990). Other kinds of depressions that hold water of a similar volume, depth, and area, and for a similar duration and seasonality to vernal pools and ponded areas within swales may also be potential habitat. These other depressions, however, are typically artificial habitats and are partially or completely unvegetated. Examples include railroad toe-drains, roadside ditches, abandoned agricultural drains, ruts left by heavy construction vehicles, and depressions in fire breaks (Eng et al. 1990).

Midvalley fairy shrimp require seasonally ephemeral aquatic habitats that pool in winter and spring. This species most commonly occurs in small to medium grassy or clay-bottomed vernal pools, roadside ditches, and railroad toe-drains (Rogers in prep.). The midvalley fairy shrimp is adapted to habitats that are inundated for short periods and can complete its life cycle (cyst to adult with fertilized eggs) in as little as 4 days, especially under extreme circumstances, such as years with below-average rainfall (Rogers in prep.). The ability to rapidly complete its life cycle allows the midvalley fairy shrimp to use habitats that are extremely hydrologically unstable (i.e., fill and dry quickly).

Little is known about midvalley fairy shrimp habitat requirements. Typically, the midvalley fairy shrimp is found in small, shallow, “flashy” vernal pools that only pond for 4 days, but it also can also be found in artificial habitats, such as railroad toe-drains, that may be up to 20 centimeters deep and pond for 3 months (Rogers in prep.). Further study may reveal that the species occurs in a wider range of conditions and pool types.

Feeding

Midvalley fairy shrimp are omnivorous filter-feeders. Fairy shrimp indiscriminately filter particles from the surrounding water, including bacteria, unicellular algae, and micrometazoa (Eriksen and Belk 1999). The precise size of items the fairy shrimp are capable of filtering is currently unknown (Eriksen and Belk 1999), but fairy shrimp will attempt to consume whatever material they can fit into their feeding groove and do not discriminate based on taste like other crustacean groups (Eriksen and Belk 1999). Midvalley fairy shrimp will also rasp periphyton from sticks, stems and slender leaves (Rogers in prep.).

Ecology

Midvalley fairy shrimp are a component of the planktonic crustacea within seasonal temporary pools and can occur in densities as high as 200 per liter of water. Planktonic crustacea are important in the food web because they represent a high-fat, high-protein resource for migratory waterfowl. Mallard, green-winged teal, bufflehead, greater yellowlegs, and killdeer all forage actively in Central Valley vernal pools on the invertebrate and amphibian fauna during winter.

Predator consumption of fairy shrimp cysts (resting eggs) aids in distributing populations of fairy shrimp. Predators expel viable cysts in their excrement, often at locations other than where they were consumed (Wissinger et. al. 1999). If conditions are suitable, these transported cysts may hatch at the new location and potentially establish a new population. Cysts are also be transported in mud carried on the feet of animals, including livestock, that may wade through the habitat (Rogers in prep.).

Other than inundation of the habitat, the specific cues for hatching are unknown (Eriksen and Belk 1999), although temperature is believed to play a large role. Typically, midvalley fairy shrimp mature in about 16 days once water temperatures reach at least 20ºC (Eriksen and Belk 1999). However, midvalley fairy shrimp can hatch, mature, and produce viable cysts in 4 days under extreme circumstances (Rogers in prep.).

Midvalley fairy shrimp have been found co-occurring with the fairy shrimp Linderiella occidentalis in the Lodi and Galt areas (Rogers in prep.). This species has also been reported co-occurring with the vernal pool fairy shrimp (*Branchinecta lynchi*) on 3 occasions, where it was likely washed into the vernal pool fairy shrimp habitat by abnormally high rainfall (Eriksen and Belk 1999).

Threats

Midvalley fairy shrimp are threatened by the same activities as other vernal pool invertebrates. These threats include the conversion of vernal pool habitat to agricultural lands and urban development, and stochastic extinction because of the small and isolated nature of remaining populations (U.S. Fish and Wildlife Service 1994). Although only recently described, midvalley fairy shrimp has probably declined over its range as a result of agricultural, suburban, and industrial conversion of its habitat (Eriksen and Belk 1999, Belk and Fugate 2000). Because of the limited and disjunct distribution of vernal pools, coupled with the even more limited distribution of the midvalley fairy shrimp, any reduction in vernal pool habitat quantity could adversely affect this species.

Habitat fragmentation can isolate and reduce population size, resulting in a process of progressive population extinctions. Small or isolated populations are more susceptible to extinction from random environmental disturbance. Recolonization opportunities are also diminished when physical barriers, such as development or lack of vernal pool habitat, isolate populations from one another or inhibit transport of cysts. Isolated populations are potentially more susceptible to inbreeding depression, which can result in local extinction or reduced fitness (Gilpin and Soule 1986, Goodman 1987a, 1987b), although this has never been demonstrated for branchiopod crustaceans.

Activities that alter the suitability of habitat may impact the special-status crustaceans dependent on these habitats. These activities include damaging the impermeable clay and/or hardpan layers of the habitat bottom, filling in the habitat, and altering (e.g., through contaminants) or destroying the watershed that conveys overland flow into the habitat. In addition, introducing nonnative plants, destroying or degrading the surrounding upland habitat, introducing fish (e.g., *Gambusia* sp.) into special-status shrimp habitats, and activities that would discourage or prevent waterfowl and waders from feeding at occupied habitats (thereby restricting gene-flow between populations), would also significantly affect midvalley fairy shrimp populations.

Conservation and Management

Conservation of the midvalley fairy shrimp is directly tied to conservation of suitable vernal pool habitat. However, because comprehensive surveys for the midvalley fairy shrimp in the inventory area have not been conducted and because known occurrences throughout the species range are based mostly on incidental observations (e.g., the California Natural Diversity Database), the population size and locations of this species in the inventory area are not known. Also, suitable habitat for the midvalley fairy shrimp in the inventory area was identified based on a general classification of land cover types. Field evaluation of the habitat classification has not been conducted, and the extent to which vernal pools in the study area meet the habitat requirements of midvalley fairy shrimp is unknown. Further, the importance of artificial habitats that may support midvalley fairy shrimp in the inventory area has not been evaluated. However, the primary data gap concerning suitable habitat for the midvalley fairy shrimp is the lack of understanding of what defines suitable habitat.

The rapid life cycle of this species (as little as 4 days) can also result in a lack of detections even while conducting protocol surveys. The U.S. Fish and Wildlife Service (1996) protocol special-status shrimp survey guidelines require that surveys are conducted in 2-week intervals, from initial inundation of the habitat in winter to its subsequent drying in spring. Therefore, standard special-status shrimp surveys according to the required protocols may not detect populations of the midvalley fairy shrimp during years with reduced rainfall.

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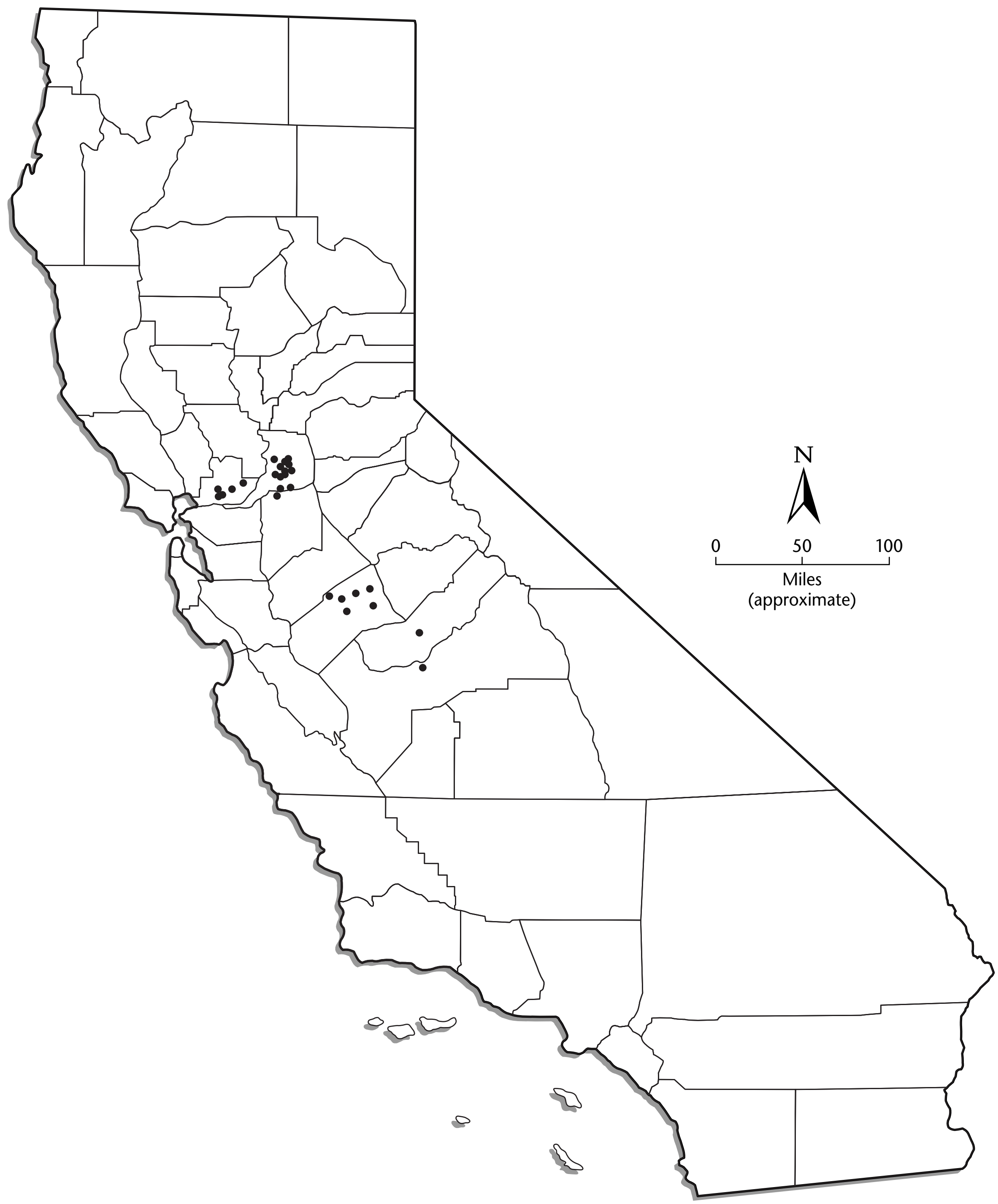

Source: Erickson and Belk 1999.

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Jones & Stokes

Midvalley Fairy Shrimp *(Brachinecta mesovallensis)* Distribution

Vernal Pool Tadpole Shrimp (*Lepidurus packardi*)

Status

State: Meets the requirements as a "rare, threatened or endangered species" under CEQA
Federal: Endangered

Population Trend

Global: Declining due to habitat loss and fragmentation (Eriksen and Belk1999).
State: Same as above
Within Inventory Area: Unknown

Data Characterization

The location database for the vernal pool tadpole shrimp (*Lepidurus packardi)* within the inventory area does not include any records from within the ECCC HCP/NCP inventory area. However, this species commonly co-occurs with the vernal pool fairy shrimp, which is known to occur within the inventory area, and is also recorded from areas adjacent to the inventory area. Other natural and artificial habitats throughout the grassland habitats within the inventory area have a high probability of being occupied by additional populations of the vernal pool tadpole shrimp. Systematic, distribution, and ecological data are presented in Rogers (2001).

Range

The vernal pool tadpole shrimp is a California Great Central Valley endemic species, with the majority of the populations occurring in the Sacramento Valley. This species has also been reported from the Sacramento River Delta to the east side of San Francisco Bay, and from a few scattered localities in the San Joaquin Valley from San Joaquin County to Madera County (Rogers 2001).

Occurrences within the ECCC HCP/NCCP Inventory Area

The vernal pool tadpole shrimp is not known to be present within the ECCC HCP/NCCP inventory area. However, due to the presence of suitable habitat and populations within close proximity to the inventory area, unrecorded populations may be present in vernal pool and swale habitat of the non-native annual grassland and in other depressions that seasonally collect rainwater. Since a comprehensive survey for the vernal pool tadpole shrimp has not been conducted in the inventory area, the current population distribution within the inventory area is unknown. As of January 2001, the California Natural Diversity Database listed 5 occurrences of vernal pool tadpole shrimp adjacent to the inventory area. The lack of data points within the open-space areas is probably due to a lack of survey effort.

Biology

Habitat

Vernal pool tadpole shrimp occur in a wide variety of seasonal habitats, including vernal pools, clay flats, alkaline pools, ephemeral stock tanks, roadside ditches, and road ruts (Rogers 2001, California Natural Diversity Database 2001). Habitats where vernal pool tadpole shrimps have been observed range in size from small, clear, well-vegetated vernal pools to highly turbid, alkali scald pools to large winter lakes (Rogers 2001). Tadpole shrimp cysts (resting eggs) must dry out before they will hatch.

Typically the vernal pool tadpole shrimp is found in habitats that are deeper than 12 centimeters, pond for 15 to 30 days, and do not suffer wide daily temperature fluctuations. The vernal pool tadpole shrimp has not been reported as utilizing strongly saline habitats. This species is found in seasonal wetlands and other winter/springtime temporarily ponded areas of sufficient size (depth and area) and seasonality that pond for a sufficient duration to maintain conducive water temperatures to allow the vernal pool tadpole shrimp to complete their life cycles (Rogers 2001).

Feeding

Tadpole shrimp are omnivores. Typically, they forage while digging through sediments at the bottom of their habitats, feeding on plants as well as metazoans. Tadpole shrimp are cannibalistic and have been observed consuming newly molted fellow tadpole shrimp (Rogers pers. comm.). In addition, vernal pool tadpole shrimp will consume fairy shrimp, including both vernal pool fairy shrimp and midvalley fairy shrimp. Though they do not actively seek out these species, they will consume them if the fairy shrimp are present at the bottom of the pool where the tadpole shrimp is foraging.

Ecology

Vernal pool tadpole shrimp are either hermaphroditic or parthenogenic (Rogers 2001). All animals produce cysts (resting eggs), which are typically shed as the animal moves about (Rogers in press). These cysts diapause (enter into a quiescent, dormant stage), remaining in the soil through the drying phase of the habitat, and then hatching as the subsequent rainy season inundates the habitat. Cysts may hatch at various times, anywhere from 1 hour to 3 weeks after the pools are inundated. The exact hatching stimuli are unknown. The vernal pool tadpole shrimp mature more slowly than fairy shrimp, and are longer lived. Typically, adults will survive until the vernal pool dries or until temperatures of 10 to 15 degrees Celsius are reached (Rogers pers. comm.). Vernal pool tadpole shrimp can begin shedding cysts in as little as 15 days.

Vernal pool tadpole shrimp are prey to amphibians and waterfowl. Predator consumption of tadpole shrimp cysts aids in distributing populations of tadpole shrimps. Predators expel the cysts in their excrement, often at a location other than where they were consumed (Rogers in prep.). If conditions are suitable, these transported cysts may hatch at the new location and potentially establish a new population. Cysts can also be transported in mud carried on the feet of animals that may wade through the habitat, such as tule elk, (*Cervus elaphus nelsoni*) feral pigs, and livestock (Rogers in prep.).

Vernal pool tadpole fairy shrimp have been found co-occurring with the fairy shrimp *Linderiella occidentalis*, *Branchinecta lynchi*, *Branchinecta coloradensis*, *Branchinecta lindahli*, and *Branchinecta conservatio* (Rogers in prep.).

Threats

Vernal pool tadpole shrimp are threatened by the same activities as other vernal pool invertebrates. These threats include the conversion of vernal pool habitat to agricultural lands and urban development, and stochastic extinction because of the small and isolated nature of remaining populations (U.S. Fish and Wildlife Service 1994). The limited and disjunct distribution of vernal pools, coupled with the even more limited distribution of the vernal pool tadpole shrimp, means that any reduction in vernal pool habitat quantity could adversely affect this species.

Habitat fragmentation can isolate and reduce population size, resulting in a process of progressive population extinctions. Small or isolated populations are more susceptible to extinction from random environmental disturbance. Recolonization opportunities are also diminished when physical barriers, such as development or lack of vernal pool habitat, isolate populations from one another or inhibit transport of cysts. Isolated populations are potentially more susceptible to inbreeding depression, which can result in local extinction or reduced fitness (Gilpin and Soule 1986, Goodman 1987a, 1987b). However this has never been demonstrated for branchiopod crustaceans.

Activities that alter the suitability of habitat may impact the special-status crustaceans dependent on these habitats. These activities include damaging the impermeable clay and/or hardpan layers of the habitat bottom, filling in the habitat, and altering (e.g. through contaminants) or destroying the watershed that conveys overland flow into the habitat. Additionally, introducing non-native plants, destroying or degrading the surrounding upland habitat, introducing fish (such as *Gambusia* spp.), into special-status shrimp habitats, and engaging in activities that discourage or prevent waterfowl and waders from feeding at occupied habitats and thereby restricting gene-flow between populations may also significantly affect midvalley fairy shrimp populations.

Conservation and Management

Conservation of the vernal pool tadpole shrimp is directly tied to conservation of suitable vernal pool habitat. However, because comprehensive surveys for the vernal pool tadpole shrimp in the ECCC HCP/NCCP inventory area have not been conducted, the population size and locations of this species in the inventory area are not known. Also, suitable habitat for the vernal pool tadpole shrimp in the inventory area was identified based on a general classification of land-cover types. Field evaluation of the habitat classification has not been conducted, and the extent to which vernal pools in the inventory area meet the habitat requirements of vernal pool tadpole shrimp is unknown. The importance of artificial habitats that may support vernal pool tadpole shrimp in the inventory area has not been evaluated.

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Source: Rogers, D.C. 2001.

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Jones & Stokes

Vernal Pool Tadpole Shrimp *(Lepidurus packardi)*
Known Occurrences

Mount Diablo Manzanita (*Arctostaphylos auriculata*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for Mount Diablo manzanita (*Arctostaphylos auriculata)* includes 19 data records dated from 1889 to 1995 (California Natural Diversity Database 2001). Only 1 occurrence was documented in the last 10 years, but all occurrences except 1 are believed to be extant (California Natural Diversity Database 2001). Fifteen of the occurrences are of high precision and may be accurately located within the inventory area.

Very little ecological information on Mount Diablo manzanita is available. The literature on the species pertains primarily to its taxonomy. The main sources of general information are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

Mount Diablo manzanita is endemic to Contra Costa County, where it occurs only on Mount Diablo and in the adjacent foothills. It is found between 700 and 1,860 feet above sea level.

Occurrences within the ECCC HCP/NCCP Inventory Area

Fourteen occurrences of Mount Diablo manzanita are within the inventory area. Eight of these occurrences are in Mount Diablo State Park or on East Bay Regional Park District lands.

Biology

Physical Description

Mount Diablo manzanita is an evergreen, perennial shrub, generally between 1 and 4.5 meters tall (Hickman 1993). Its blooming period is from January to March (California Native Plant Society 2001).

Habitat

Mount Diablo manzanita occurs primarily in chamise or manzanita chaparral. It can also be found as an understory shrub in coast live oak woodland (California Natural Diversity Database 2001).

Species Associated with Mount Diablo Manzanita	
Adenostoma fasciculata	chamise
Arctostaphylos glandulosa	Eastwood manzanita
Arctostaphylos manzanita	common manzanita
Artemisia californica	California sagebrush
Baccharis pilularis	coyote bush
Ceanothus cuneatus	wedge-leaf ceanothus
Ericameria linearifolia	narrowleaf goldenbush
Eriodictyon californica	yerba santa
Eriogonum nudum	naked-stem wild buckwheat
Galium porrigens	climbing bedstraw
Helianthella castanea	Diablo helianthella
Heteromeles arbutifolia	toyon
Mimulus aurantiacus	bush monkeyflower
Pickeringia montana	chaparral pea
Pinus attenuata	knob-cone pine
Pinus coulteri	Coulter pine
Pinus sabiniana	gray pine
Quercus agrifolia	coast live oak
Quercus chrysolepis	canyon live oak
Quercus durata	leather oak
Rhus trilobata	skunkbrush
Salvia mellifera	black sage
Zigadenus fremontii	Fremont's death-camas

Threats

Mount Diablo manzanita is restricted to a few occurrences in a limited area, but it does not appear to be endangered (California Native Plant Society 2001). Potential threats to Mount Diablo manzanita include direct loss of plants and disturbance that could alter the stand composition. Direct loss of plants could occur from clearing for firebreaks, trail maintenance, road maintenance, and facilities development (California Natural Diversity Database 2001). Activities such as grazing, off-road vehicle use, and dumping cause disturbances that could alter the interaction between chaparral and the adjacent plant communities or allow invasion by exotic species.

Conservation and Management

The long-term maintenance of Mount Diablo manzanita stands will likely depend on fire management practices in the area in which the stands occur. Periodic fires have had a major role in shaping the structure and composition of chaparral stands. Stands are affected by fire intensity and frequency, and by the response to fire by individual plant species. Mount Diablo manzanita does not resprout after fire (Jepson 1922); instead, stands regenerate by recruiting new individuals from seed. In older stands, much of the aboveground biomass consists of dead stems and litter from fallen leaves and twigs. Fire is necessary to allow the establishment of new plants from seeds by removing the overtopping vegetation; it may also stimulate seed germination. Prescription fire plans may need to be created that include conservation measures for Mount Diablo manzanita, such as let-burn areas, controlled burns, and fire intervals.

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Jones & Stokes

Mt. Diablo Manzanita *(Arctostaphylos auriculata)*
Habitat Distribution

Brittlescale (*Atriplex depressa*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The California Natural Diversity Database (2001) reports 57 occurrences of brittlescale (*Atriplex depressa*), but only 40 of the occurrences are brittlescale populations; 17 of the reported occurrences in central and southern San Joaquin Valley are based on misidentifications of lesser saltscale (*Atriplex minuscula*) populations (Preston pers. comm.). The records are dated from 1920 to 1996. Twenty-four of the occurrences were documented within the last 10 years. All of the occurrences are believed to be extant (California Natural Diversity Database 2001). Eight of the occurrences are within the inventory area. The occurrences were mapped with high precision and may be accurately located, including those within the inventory area.

Very little information is available on the ecology of brittlescale. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001) and in Jones & Stokes file records.

Range

Brittlescale occurs along the western side of the Great Valley from Glenn County to Merced County and in the small valleys of the inner Coast Ranges, including the Livermore Valley. It occurs in the broad flood basins of the valley floor and on alluvial fans associated with the major streams draining from the inner Coast Range foothills. It is generally found at low elevations but has been collected up to 1,055 feet above sea level.

Occurrences within the ECCC HCP/NCCP Inventory Area

Eight occurrences of brittlescale are present in the inventory area (California Natural Diversity Database 2001). Three occurrences are on Contra Costa Water

District lands at Los Vaqueros Reservoir; all others are on private lands south and west of Byron.

Biology

Physical Description

Brittlescale is a diminutive annual herb that generally grows prostrate and rarely exceeds 20 centimeters in height (Hickman 1993).

Habitat

Brittlescale occurs on alkali soils of the Pescadero and Solano series. Brittlescale typically occurs in barren areas within alkali grassland, alkali meadow, and alkali scrub. It is occasionally found on the margins of alkali vernal pools.

Species Associated with Brittlescale	
Atriplex cordulata	heartscale
Atriplex coronata	crownscale
Atriplex fruticulosa	ball saltscale
Atriplex joaquiniana	San Joaquin spearscale
Centromadia pungens	common spikeweed
Distichlis spicata	saltgrass
Frankenia salina	alkali heath
Hordeum depressum	low barley
Hordeum marinum ssp. *gussoneanum*	Mediterranean barley
Nitrophila occidentalis	western niterwort
Salicornia subterminalis	Parish's pickleweed
Spergularia macrotheca	large-flowered sand-spurry
Suaeda moquinii	bush seepweed

Threats

Brittlescale is known from only a limited number of occurrences and is endangered in a portion of its range (California Native Plant Society 2001). Population trends are unknown (California Natural Diversity Database 2001), but are likely stable or declining. The principal threat to brittlescale has been the historic conversion of much of the alkali grassland to agriculture. Present threats include flooding of alkali grassland to create waterfowl habitat, grazing, and urban development (California Natural Diversity Database 2001, California Native Plant Society 2001).

Conservation and Management

Areas with alkali soils are prepared for agriculture by treating the soils with gypsum or other substances that allow sodium salts to be leached from the soil by irrigation. This practice alters the soil chemistry, making restoration of former brittlescale habitat impractical.

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Personal Communications

Preston, R. E. Botanist, Jones & Stokes. May 2001—visit to the University of California and Jepson Herbaria to examine Atriplex specimens.

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Jones & Stokes

Brittlescale *(Atriplex depressa)*
Habitat Distribution

San Joaquin Spearscale (*Atriplex joaquiniana*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for San Joaquin spearscale (*Atriplex joaquiniana)* includes 50 data records dated from 1891 to 1996 (California Natural Diversity Database 2001). All of the occurrences are presumed to be extant, but 5 occurrences are historic and have not been surveyed to determine whether the populations are still present. Most of the occurrences are of high precision and may be accurately located, including those in the inventory area.

Very little information is available on the ecology of San Joaquin spearscale. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

San Joaquin spearscale occurs along the western side of the Great Valley from Glenn County to Merced County and in the small valleys of the inner Coast Ranges, including the Livermore Valley. It occurs in the broad flood basins of the valley floor and on alluvial fans associated with the major streams draining from the inner Coast Ranges foothills. It is generally found at low elevations, but has been collected up to 1,055 feet above sea level.

Occurrences within the ECCC HCP/NCCP Inventory Area

Twenty-one occurrences of San Joaquin spearscale are within the inventory area (California Natural Diversity Database 2001). The occurrences are on private lands in the eastern portion of the inventory area, including within Lone Tree Valley, Briones Valley, and the Brushy Creek watershed south of Byron.

Biology

Physical Description

San Joaquin spearscale is an annual herb between 1 and 3 feet tall (Hickman 1993). It blooms from April to October (California Native Plant Society 2001).

Habitat

San Joaquin spearscale typically occurs in alkali grassland and alkali meadow, or on the margins of alkali scrub. It occurs on clay soils, often in areas of high alkalinity.

Species Associated with San Joaquin Spearscale	
Allenrolfea occidentalis	iodine bush
Atriplex coronata	crownscale
Atriplex depressa	brittlescale
Centromadia pungens	common spikeweed
Cordylanthus palmatus	palmate bird's-beak
Distichlis spicata	saltgrass
Frankenia salina	alkali heath
Hordeum depressum	low barley
Hordeum marinum ssp. *gussoneanum*	Mediterranean barley
Lolium multiflorum	Italian ryegrass
Nitrophila occidentalis	western niterwort
Salicornia subterminalis	Parish's pickleweed
Spergularia macrotheca	large-flowered sand-spurry
Suaeda moquinii	bush seepweed

Threats

San Joaquin spearscale is known from only a limited number of occurrences and is endangered in a portion of its range (California Native Plant Society 2001). Population trends are unknown (California Natural Diversity Database 2001), but are likely stable or declining. The principal threat to San Joaquin spearscale has been the historic conversion of much of the alkali grassland to agriculture. Present threats include habitat conversion to urban use, overgrazing, and impacts associated with road and utility line construction and maintenance (California Natural Diversity Database 2001).

Conservation and Management

Areas with alkali soils are prepared for agriculture by treating the soils with gypsum or other substances that allow sodium salts to be leached from the soil by irrigation. This practice alters the soil chemistry, making restoration of former San Joaquin saltscale habitat impractical.

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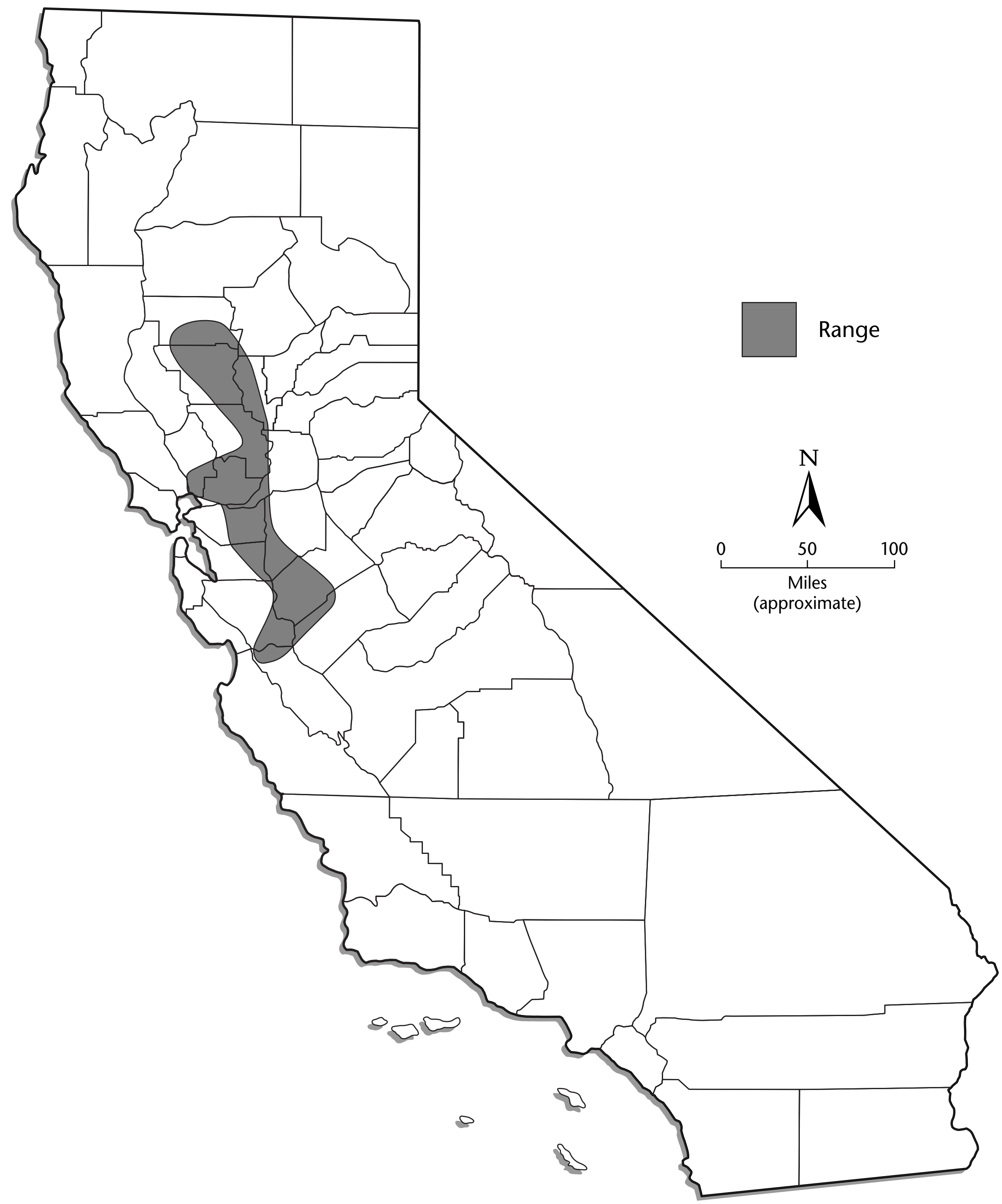

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Jones & Stokes

San Joaquin Spearscale *(Atriplex joaquiniana)*
Habitat Distribution

Big Tarplant (*Blepharizonia plumosa*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for big tarplant includes 36 data records dated from 1916 to 1998 (California Natural Diversity Database 2001). Twenty-nine of the occurrences were documented within the last 10 years. Seven of the occurrences have not been observed for over 60 years, but all the other occurrences are believed to be extant (California Natural Diversity Database 2001). Most of the occurrences are of high precision and may be accurately located, including those within the inventory area.

Very little ecological information is available for big tarplant. The published literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the Jepson Manual (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

Big tarplant is endemic to the Mount Diablo foothills and is found primarily in eastern Contra Costa, eastern Alameda, and western San Joaquin Counties (Hoover 1937).

Occurrences within the ECCC HCP/NCCP Inventory Area

In the inventory area, big tarplant is known from 4 occurrences on Cowell Ranch, west of Brentwood, and 7 occurrences on Roddy Ranch, south of Antioch (California Natural Diversity Database 2001). The historic occurrences in Antioch are likely to have been extirpated, although at least 1 population is present at Black Diamond Mines Regional Park (Preston pers. comm.). Big tarplant may also be present in the hills south of Pittsburg, where it was collected in 1937 and last seen in 1992 (Preston pers. comm.).

Biology

Physical Description

Big tarplant is an herbaceous annual that grows to between 1 and 3 feet tall. Seedlings appear in early spring, but the plants do not begin to bloom until mid-summer. The blooming period, during which the plants produce many heads with white flowers, generally occurs between August and October.

Two species of big tarplant are present in the inventory area: big tarplant and viscid big tarplant (*Blepharizonia laxa*). Viscid big tarplant is the more widely distributed species, ranging throughout most of the south Coast Ranges and reaching its northern limit in Contra Costa County. The two species, which often occur in adjacent populations, can be differentiated by their branching patterns, the amount and color of the simple and glandular hairs on the stems and leaves, the chemical compounds produced by the glands, and by genetic markers (Hickman 1993, Baldwin et al. 2000, Preston pers. comm.). The two species can hybridize, but the hybrids are infertile (Baldwin et al. 2000).

Habitat

Big tarplant occurs in annual grassland on clay to clay-loam soils, usually on slopes and often in burned areas, below 1,500 feet (California Natural Diversity Database 2001). In Contra Costa County, the occurrences are primarily on soils of the Altamont series.

Species Associated with Big Tarplant	
Avena species	wild oats
Bromus species	brome grasses
Epilobium brachycarpum	panicled willow-herb
Eriogonum angulosum	angle-stemmed wild buckwheat
Eriogonum gracile	slender woolly wild buckwheat
Grindelia camporum	Great Valley gumplant
Holocarpha obconica	San Joaquin tarplant
Holocarpha virgata	virgate tarplant
Lagophylla ramosissima	common hareleaf
Lolium multiflorum	Italian ryegrass
Nassella pulchra	purple needlegrass

Threats

Big tarplant occurs in only a few highly restricted populations and is endangered throughout its range (California Native Plant Society 2001). The primary threat to big tarplant has been habitat loss from conversion to urban development. Ground disturbance and erosion caused by cattle grazing and competition from invasive exotics such as yellow star-thistle (*Centaurea solstitialis*) may also pose a threat to populations (California Natural Diversity Database 2001).

Conservation and Management

Big tarplant may require management of nonnative annual grasses for long-term population viability. Prescribed burns may be an effective method for managing grasslands in which big tarplant occurs. Such burns should be conducted under conditions that favor low-intensity fire because high plant mortality appears to result from high-intensity fires.

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Jones & Stokes

Big Tarplant ***(Blepharizonia plumosa)***
Habitat Distribution

Mount Diablo Fairy-Lantern (*Calochortus pulchellus*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for Mount Diablo fairy-lantern (*Calochortus pulchellus)* includes 29 data records dated from 1940 to 1996 (California Natural Diversity Database 2001). Over half of the occurrences were documented in the previous 10 years, and all of the occurrences are believed to be extant. Most of the occurrences are of high precision and may be accurately located, including those within the inventory area.

Very little information is available on the ecology of Mount Diablo fairy-lantern. The literature on the species pertains primarily to its taxonomy. The main sources of general information on the species are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

Mount Diablo fairy-lantern is endemic to the Diablo Range in Contra Costa County, ranging in elevation between 650 and 2,600 feet (Hickman 1993). These occurrences are mostly located on lands managed by the California Department of Parks and Recreation, East Bay Recreation and Park District, and City of Walnut Creek, with several populations occurring on privately owned land or land of unknown ownership (California Natural Diversity Database 2001).

Occurrences within the ECC HCP/NCCP Inventory Area

Six occurrences of Mount Diablo fairy-lantern are within the inventory area. Five of the occurrences are either in Mount Diablo State Park or East Bay Regional Park District lands.

Biology

Physical Description

Mount Diablo fairy-lantern, a member of the lily family (Liliaceae), is a bulbiferous perennial herb that grows 10 to 30 centimeters tall (Hickman 1993). It blooms from April through June, producing bright yellow, pendant flowers.

Habitat

Mount Diablo fairy-lantern grows on grassy slopes and in openings in chaparral and oak woodland communities (California Natural Diversity Database 2001).

Species Associated with Mount Diablo Fairy-Lantern	
Arctostaphylos species	manzanita
Quercus species	oaks
Pinus sabiniana	foothill pine
Aesculus californica	California buckeye
Toxicodendron diversiloba	poinson-oak
Melica torreyana	Torrey melic
Festuca californica	California fescue
Dodecatheon species	shooting-stars
Phacelia species	phacelia

Threats

Mount Diablo fairy-lantern is known from only a limited number of occurrences and is endangered in a portion of its range (California Native Plant Society 2001). Population trends are unknown (California Natural Diversity Database 2001), but are likely stable. Threats to Mount Diablo fairy-lantern include grazing, road and trail maintenance, excessive erosion, and collecting (California Natural Diversity Database 2001).

Conservation and Management

There are no measures being taken in the inventory area to conserve or manage populations of Mount Diablo fairy-lantern.

Literature Cited

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Jones & Stokes

Mt. Diablo Fairy-Lantern *(Calochortus pulchellus)*
Habitat Distribution

Recurved Larkspur (*Delphinium recurvatum*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for recurved larkspur (*Delphinium recurvatum)* includes 63 data records dated from 1902 to 1998 (California Natural Diversity Database 2001). Thirteen of the occurrences are more than 50 years old, and only 20 of the occurrences were documented in the previous 10 years, but most of the occurrences are assumed to be extant. Twenty-seven of the occurrences are of high precision and may be accurately located, including 2 of 4 located within the inventory area.

Very little ecological information is available for recurved larkspur. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

Historically, recurved larkspur was widely distributed in California's Great Valley, ranging from Butte County to Kern County. Most of the known occurrences are in Kern, Tulare, and San Luis Obispo Counties. The species now appears to be very rare outside the southern San Joaquin Valley (California Natural Diversity Database 2001).

Occurrences within the ECCC HCP/NCCP Inventory Area

Four occurrences are reported from the inventory area, all on private land southeast of Byron.

Biology

Physical Description

Recurved larkspur is a perennial herb and a member of the buttercup family (Ranunculaceae). Recurved larkspur is distinguished from other larkspur species by its pale blue, recurved sepals (Hickman 1993). The flowering period for recurved larkspur is generally from March through May (California Native Plant Society 2001).

Habitat

Recurved larkspur occurs on sandy or clay alkaline soils, generally in annual grasslands or in association with saltbush scrub or valley sink scrub habitats, ranging in elevation from 100 to 2,000 feet above sea level (California Natural Diversity Database 2001).

Species Associated with Recurved Larkspur	
Atriplex polycarpa	allscale
Atriplex spinifera	spinescale
Bromus madritensis ssp. *rubens*	red brome
Centromadia pungens	common spikeweed
Distichlis spicata	saltgrass
Erodium cicutarium	red filaree
Frankenia salina	alkali heath
Isocoma acradenia var. *bracteosa*	alkali goldenbush
Lasthenia californica	California goldfields
Sporobolus airoides	alkali sacaton
Suaeda moquinii	bush seepweed

Threats

Recurved larkspur is known from only a limited number of occurrences and is endangered in a portion of its range (California Native Plant Society 2001). Population trends are unknown (California Natural Diversity Database 2001), but are likely stable or declining. The principal threat to recurved larkspur has been the historic conversion of much of the alkali habitat of the Great Valley to agriculture. At present, the primary threat to recurved larkspur is overgrazing. Other threats include road and utility line construction and competition from invasive exotics (California Natural Diversity Database 2001)

Conservation and Management

Areas with alkali soils are prepared for agriculture by treating the soils with gypsum or other substances that allow the sodium salts to be leached from the soil by irrigation. This practice alters the soil chemistry, making restoration of former recurved larkspur habitat impractical.

Literature Cited

California Native Plant Society. 2001. *Inventory of Rare and Endangered Plants of California* (sixth edition). Rare Plant Scientific Advisory Committee, David P. Tibor, Convening Editor. Sacramento, CA.

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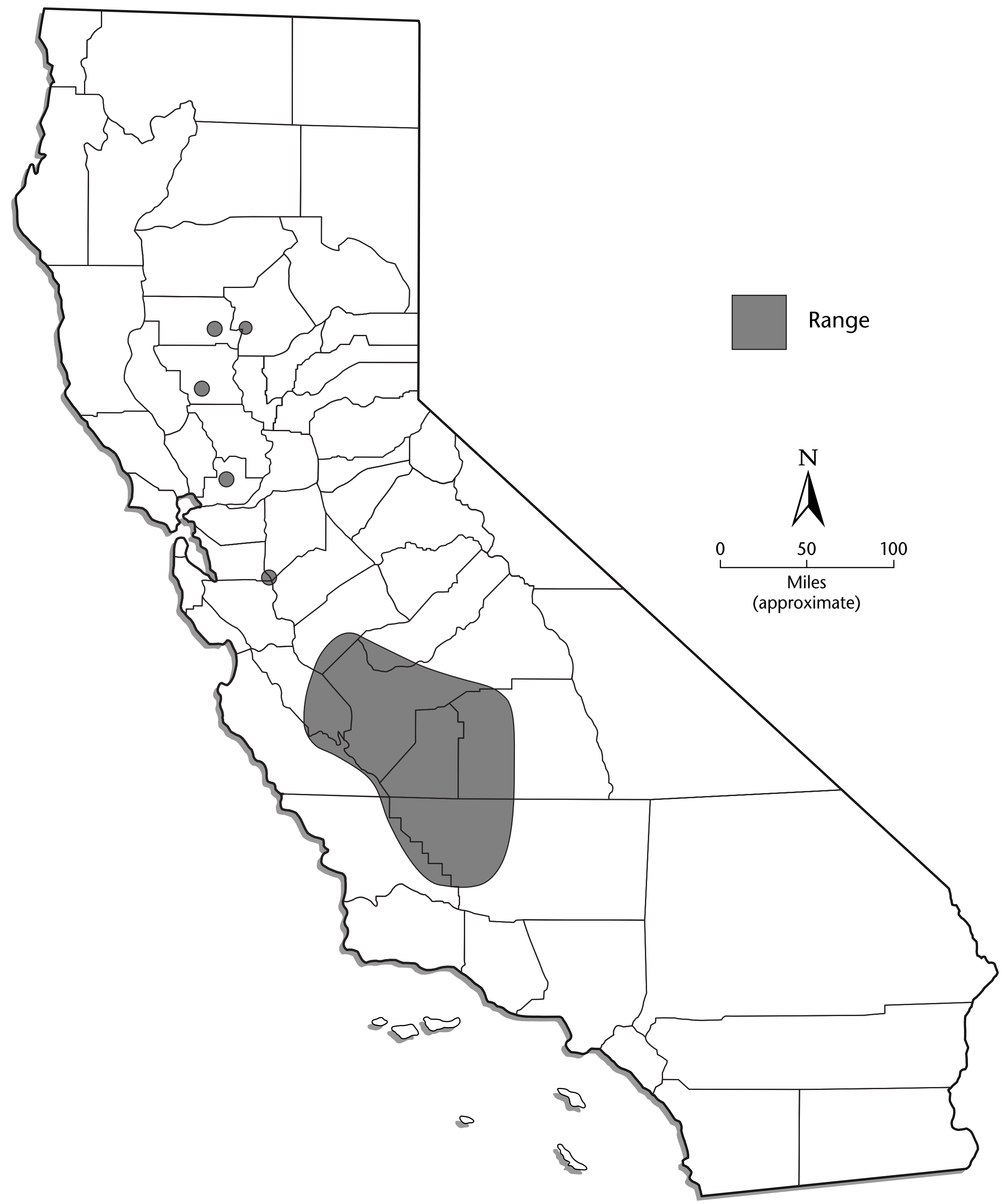

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Jones & Stokes

Recurved Larkspur *(Delphinium recurvatum)*
Habitat Distribution

Diamond-Petaled Poppy (*Eschscholzia rhombipetala*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Declining
State: Declining
Within Inventory Area: Possibly extirpated

Data Characterization

The location database for diamond-petaled poppy (*Eschscholzia rhombipetala)* includes 8 data records from 1889 to 1997 (California Natural Diversity Database 2001). Only 2 occurrences were documented in the previous 10 years. Prior to the observation of these 2 populations, the species was believed to be extinct (Skinner and Pavlik 1994). The occurrences are general and may not be accurately located.

Very little information is available for diamond-petaled poppy. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are Ernst (1964), the *Jepson Manual* (Hickman 1993), Clark (2000), and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are present in the California Natural Diversity Data Base (California Natural Diversity Database 2001).

Range

The species had been known to occur primarily in the eastern foothills of the Mount Hamilton and Diablo Ranges from Alameda County to Stanislaus County, with a disjunct occurrence in the Carrizo Plains, in San Luis Obispo County (Ernst 1964). The species was reported from San Luis Obispo County in 1988, but the identification of plants from this location is disputed (California Natural Diversity Database 2001). The species was collected in 1993 from the Carrizo Plains by David Keil and again in 1995 by Curtis Clark (Clark 2000). The species was rediscovered in Corral Hollow in Alameda County in 1997 (California Natural Diversity Database 2001).

Occurrences within the ECCC HCP/NCCP Study Area

Diamond-petaled poppy was collected near Byron in 1888 and near Antioch in 1889. It has not been observed in Contra Costa County since that time.

Biology

Physical Description

Diamond-petaled poppy is a diminutive annual herb. It grows less than 30 cm tall, and the flowers are typically less than 1 cm wide (Hickman 1993). Because of its small stature, the species may often be overlooked (Clark 2000). The blooming period is in March and April.

Habitat

Diamond-petaled poppy occurs in grasslands on heavy clay soils. It generally grows where cover of non-native annual grasses is low.

Species Associated with Diamond-Petaled Poppy	
Avena barbata	slender wild oat
Bromus madritensis ssp. *rubens*	red brome
Lasthenia californica	California goldfields
Microseris douglasii	Douglas' microseris
Poa secunda	one-sided bluegrass
Stylomecon heterophylla	wind poppy

Threats

The principal threat to diamond-petaled poppy may be competition from non-native annual grasses. The diminutive stature may be part of a set of adaptations to harsh habitat conditions (Clark and Jernstedt 1978). In years with favorable conditions, non-native annual grasses may outcompete diamond-petaled poppy by occupying most of the suitable habitat and shading out the poppy.

Conservation and Management

Any populations rediscovered in the inventory area should be preserved and protected.

Literature Cited

California Native Plant Society. 2001. *Inventory of Rare and Endangered Plants of California* (sixth edition). Rare Plant Scientific Advisory Committee, David P. Tibor, Convening Editor. California Native Plant Society, Sacramento, CA.

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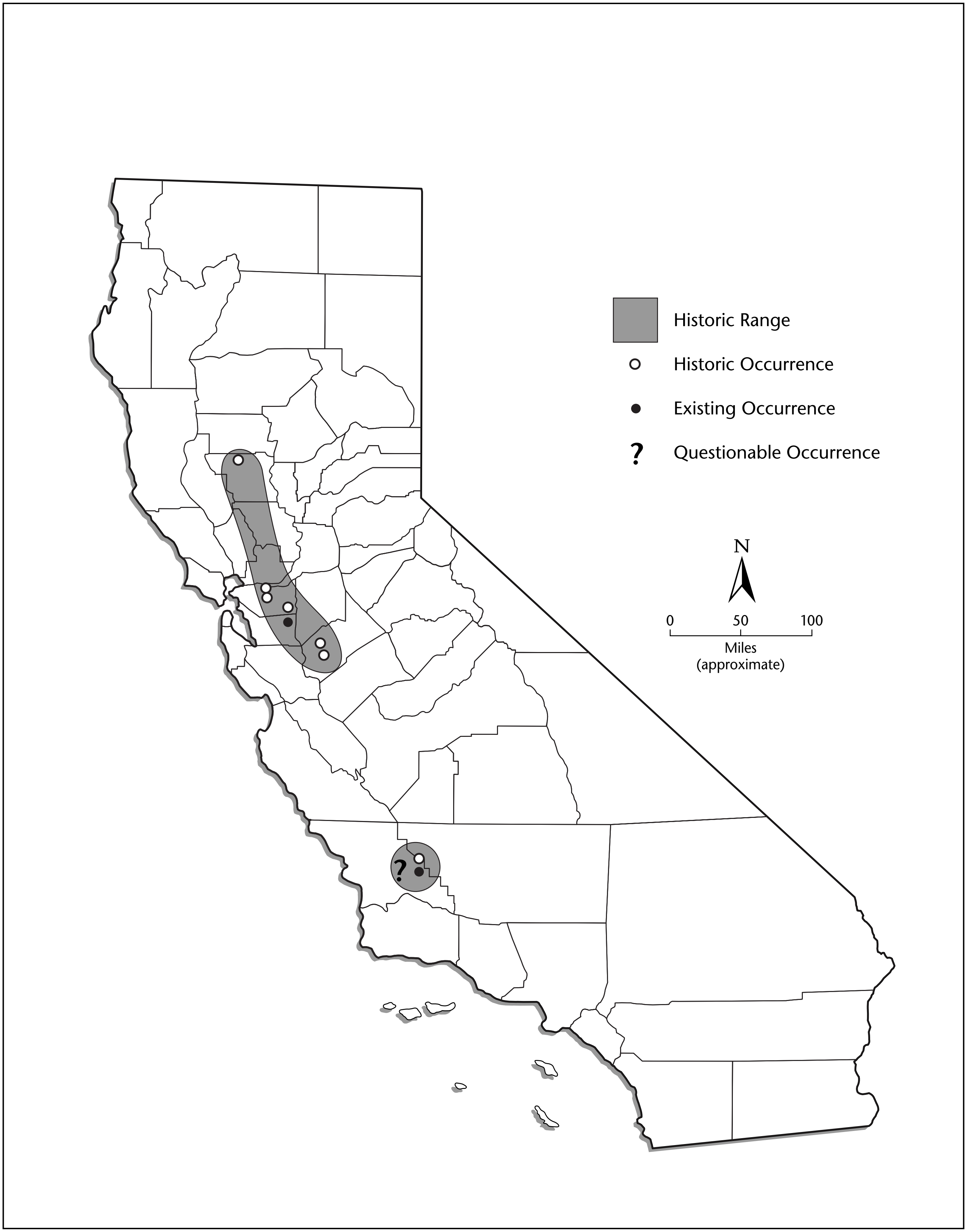

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Jones & Stokes

Diamond-Petaled Poppy (*Eschscholtzia rhombipetala*)
Habitat Distribution

Diablo Helianthella (*Helianthella castanea*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for Diablo helianthella (*Helianthella castanea*) includes 71 data records dated from 1920 to 1998 (California Natural Diversity Database 2001). Forty-two of the occurrences were documented in the last 10 years, and most of the occurrences were documented in the last 20 years. All of the occurrences are believed to be extant. Most of the occurrences are of high precision and may be accurately located, including those within the inventory area.

Very little ecological information is available for Diablo helianthella. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the Jepson Manual (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

Diablo helianthella is endemic to the San Francisco Bay Area, occurring in the Diablo Range, Berkeley Hills, and San Bruno Mountain (California Natural Diversity Database 2001).

Occurrences within the ECCC HCP/NCCP Inventory Area

Ten occurrences are reported from the inventory area: 8 in Mount Diablo State Park on East Bay Regional Park District lands and 2 on private land.

Biology

Physical Description

Diablo helianthella is a perennial herb of the sunflower family (Asteraceae) that grows 10 to 50 centimeters tall (Hickman 1993). It blooms from April through June (California Native Plant Society 2001).

Habitat

Diablo helianthella associated with thin, rocky, well-drained soils on east-facing slopes. It is found in grassy openings in woodlands, chaparral, and coastal scrub, often at the transition zone between woodland and chaparral (California Natural Diversity Database 2001).

Species Associated with Diablo Helianthella	
Adenostoma fasciculata	chamise
Artemisia californica	California sage
Avena species	wild oats
Baccharis pilularis	coyote brush
Bromus species	brome grasses
Heteromeles arbutifolia	toyon
Mimulus aurantiacus	bush monkeyflower
Nassella species	needlegrass
Quercus agrifolia	coast live oak
Quercus douglasii	blue oak
Salvia species	sage
Toxicodendron diversilobum	poison-oak
Umbellularia californica	California bay
Wyethia species	mule-ears

Threats

Diablo helianthella is known from only a limited number of occurrences and is endangered in a portion of its range (California Native Plant Society 2001). Population trends are unknown (California Natural Diversity Database 2001), but are likely stable. Many of the occurrences on park lands are subject to impacts from recreation and associated activities, such as trail construction and maintenance, road maintenance, brush-clearing, and off-trail travel (California

Natural Diversity Database 2001). Diablo helianthella grows in openings in chaparral and at chaparral margins; because chaparral species can invade these open areas in the absence of fire, fire suppression may lead to the loss of suitable habitat. Other threats include urban development, road and utility line construction, grazing, and competition from invasive exotics (California Natural Diversity Database 2001). Grazing and other ground-disturbing activities can also lead to erosion in habitat areas.

Conservation and Management

The long-term maintenance of Diablo helianthella may depend on fires that create openings in the woody overstory of scrub and woodland habitats in which the species occurs.

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Jones & Stokes

Diablo Helianthella *(Helianthella castanea)*
Habitat Distribution

Brewer's Dwarf Flax (*Hesperolinon breweri*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

The location database for Brewer's dwarf flax (*Hesperolinon breweri*) includes 25 data records dated from 1885 to 1997 (California Natural Diversity Database 2001). Only 3 occurrences were documented in the last 10 years, but all occurrences are believed to be extant (California Natural Diversity Database 2001). Fourteen of the occurrences are of high precision and may be accurately located, including 8 occurrences within the inventory area.

Very little ecological information is available for Brewer's dwarf flax. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (2001). Specific observations on habitat and plant associates, threats, and other factors are summarized in the California Natural Diversity Database (2001).

Range

Brewer's dwarf flax is endemic to California, where it is restricted to the Mount Diablo and adjacent foothills in the east San Francisco Bay Area and to the Vaca Mountains of the southern interior North Coast Ranges (Hickman 1993, California Natural Diversity Database 2001). It occurs below 2,900 feet above sea level.

Occurrences within the ECCC HCP/NCCP Inventory Area

Twelve occurrences of Brewer's dwarf flax are within the inventory area. Eleven of the occurrences are in Mount Diablo State Park, East Bay Regional Park District lands, or Contra Costa Water District lands west of Los Vaqueros Reservoir.

Biology

Physical Description

Brewer's dwarf flax, a member of the flax family (Linaceae), is an annual herb that grows 5 to 20 centimeters tall (Hickman 1993). It blooms from May through July (California Native Plant Society 2001).

Habitat

The species grows on rocky soils on serpentine, sandstone, or volcanic substrates. It is associated with grassland, oak woodland, and chaparral communities. It typically appears in areas with low vegetative cover, such as the transition zone between grassland and chaparral or open areas in chaparral.

Species Associated with Brewer's Dwarf Flax	
Adenostoma fascicularis	chamise
Arctostaphylos species	manzanita
Avena species	wild oat
Calochortus species	fairy-lantern
Ceanothus cuneatus	buckbrush
Heteromeles arbutifolia	toyon
Nassella species	needlegrass
Navarretia pubescens	downy navarretia
Perideridia kelloggii	Kellogg's yampah
Pinus sabiniana	foothill pine
Quercus species	oak
Streptanthus species	jewelflower

Threats

Brewer's dwarf flax is known from only a limited number of occurrences and is reported to be endangered in a portion of its range (California Native Plant Society 2001). Population trends are unknown (California Natural Diversity Database 2001), but are likely stable. Brewer's dwarf flax generally occurs on public lands with few identifiable threats. Populations adjacent to trails may be subject to foot traffic or trail maintenance (California Natural Diversity Database 2001).

Conservation and Management

There are no known conservation or management activities occurring in the inventory area to address Brewer's dwarf flax.

Literature Cited

California Native Plant Society. 2001. *Inventory of Rare and Endangered Plants of California* (sixth edition). Rare Plant Scientific Advisory Committee, David P. Tibor, Convening Editor. Sacramento, CA.

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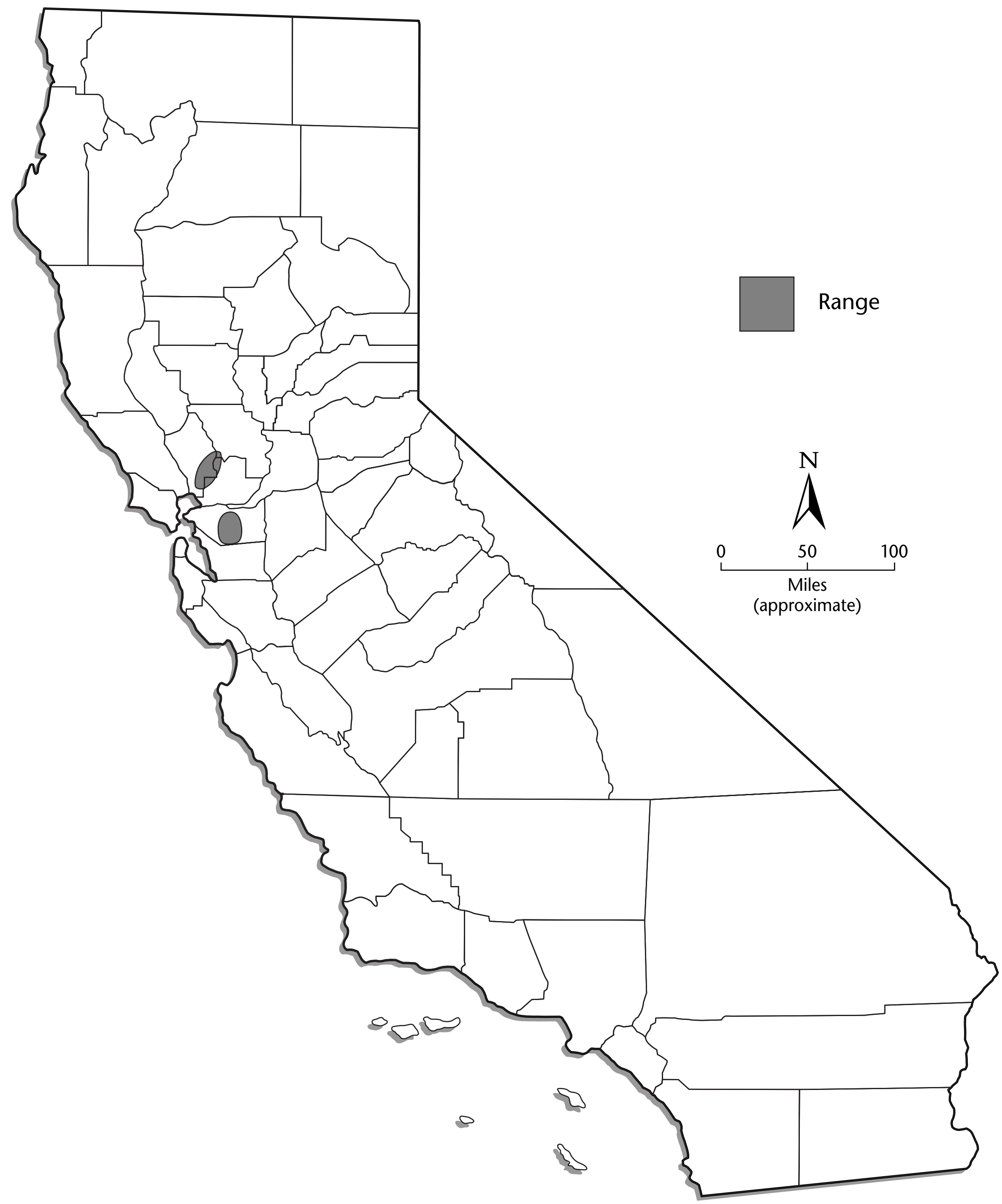

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Jones & Stokes

Brewer's Dwarf Flax *(Hesperolinon breweri)*
Habitat Distribution

Showy Madia (*Madia radiata*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Possibly extirpated

Data Characterization

The location database for showy madia (*Madia radiata)* includes 32 data records from 1930 to 1995 (California Natural Diversity Database 2001). Only 5 occurrences were documented in the previous 10 years. Except for the most recent observations, the occurrences are general and may not be accurately located.

Very little information is available for showy madia. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the *Jepson Manual* (Hickman 1993) and the California Native Plant Society (California Native Plant Society 2001). Specific observations on habitat and plant associates, threats, and other factors are present in the California Natural Diversity Data Base (California Natural Diversity Database 2001).

Range

Showy madia is known from scattered populations in the interior foothills of the South Coast Ranges found between 80 and 3,700 feet elevation (Hickman 1993; California Natural Diversity Database 2001).

Occurrences within the ECCC HCP/NCCP Inventory Area

Showy madia has been collected historically near Antioch and between Antioch and Lone Tree Valley (California Natural Diversity Database 2001). The last observation of this species in Contra Costa County was in 1941 (California Natural Diversity Database 2001).

Biology

Physical Description

Showy madia is an annual herb that blooms from March to May (California Native Plant Society 2001).

Habitat

Showy madia grows in grasslands and oak woodlands on heavy clay soils (California Natural Diversity Database 2001).

Species Associated with Showy Madia	
Ancistrocarphus filagineus	woolly fishhooks
Astragalus didymocarpus	two-seeded milkvetch
Eremalche parryi	Parry's mallow
Guillenia flavescens	yellow-flowered guillenia
Layia heterotricha	pale-yellow layia
Lupinus microcarpus	chick lupine
Monolopia major	cupped monolopia
Phacelia ciliata	Great Valley phacelia
Salvia columbariae	chia

Threats

General threats reported for showy madia include grazing, road maintenance, off-road vehicle traffic, and competition from non-native invasive plant (California Native Plant Society 2001; California Natural Diversity Database 2001). Occurrences in the vicinity of Antioch may have been extirpated by urban development, although 2 of the occurrences are on rural lands that have not yet been developed (California Natural Diversity Database 2001).

Conservation and Management

No populations of showy madia are currently known in the ECCC HCP/NCCP inventory area, although suitable habitat is likely to be present. Any populations rediscovered in the inventory area should be preserved and protected. Areas of suitable habitat could be preserved and protected that may harbor undetected occurrences of showy madia or that could be used for potential reintroduction.

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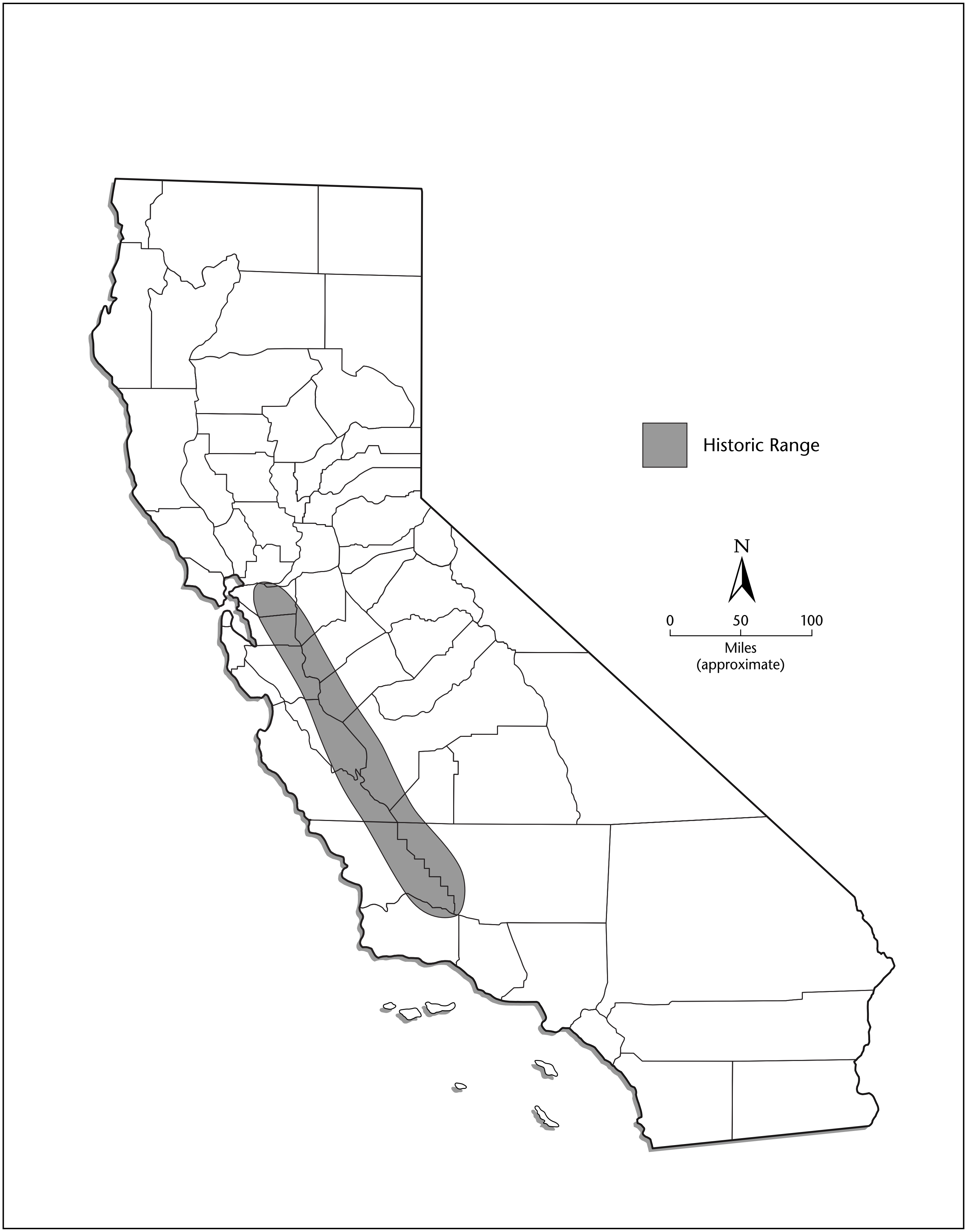

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Jones & Stokes

Showy Madia *(Madia radiata)*
Habitat Distribution

Adobe Navarretia (*Navarretia nigelliformis* ssp. *nigelliformis*)

Status

Federal: None
State: None
CNPS: List 1B

Population Trend

Global: Unknown
State: Unknown
Within Inventory Area: Unknown

Data Characterization

Location databases include 12 data records for adobe navarretia from 1898 to 2000 (CalFlora 2000; California State University Chico 2002). Only 1 of these occurrences was documented in the previous 10 years. All of the occurrences are of general precision and may not be accurately located.

Very little information is available for adobe navarretia. The literature on the species pertains primarily to its taxonomy. The main sources of general information on this species are the Jepson Manual (Hickman 1993) and the location data records.

Range

Adobe navarretia is reported to occur in the Sierra Nevada foothills, the Central Valley, and the inner South Coast Ranges, between 325 and 3,300 feet elevation (Hickman 1993).

Occurrences within the ECCC HCP/NCCP Inventory Area

Adobe navarretia has been collected in the vicinity of Antioch (CalFlora 2000) and has been reported from the Los Vaqueros area (Ertter 1997).

Biology

Physical Description

Adobe navarretia is an annual herb that blooms in April and May (Munz 1959). The small flowers are yellow with brown spots below the petal lobes (Hickman 1993).

Habitat

Adobe navarretia occurs in heavy clay soils of vernal pools and other low, seasonally moist areas in grasslands (Hickman 1993).

Species Associated with Adobe Navarretia	
Achyrachaena mollis	blow-wives
Bromus hordeaceus	soft chess
Deschampsia danthonioides	Annual hairgrass
Epilobium pygmaeum	smooth spike-primrose
Eryngium sp.	coyote-thistle
Gastridium ventricosum	nitgrass
Hordeum marinum ssp. *gussoneanum*	Mediterranean barley
Juncus bufonius	toad rush
Plagiobothrys acanthocarpus	adobe popcorn-flower
Vulpia bromoides	foxtail fescue

Threats

Specific threats to adobe navarretia are not known, although general threats to the species would be similar to those for other vernal pool species, including habitat conversion.

Special Biological Considerations

Adobe navarretia appears to be restricted to areas with a vernally moist, summer-dry hydrologic regime.

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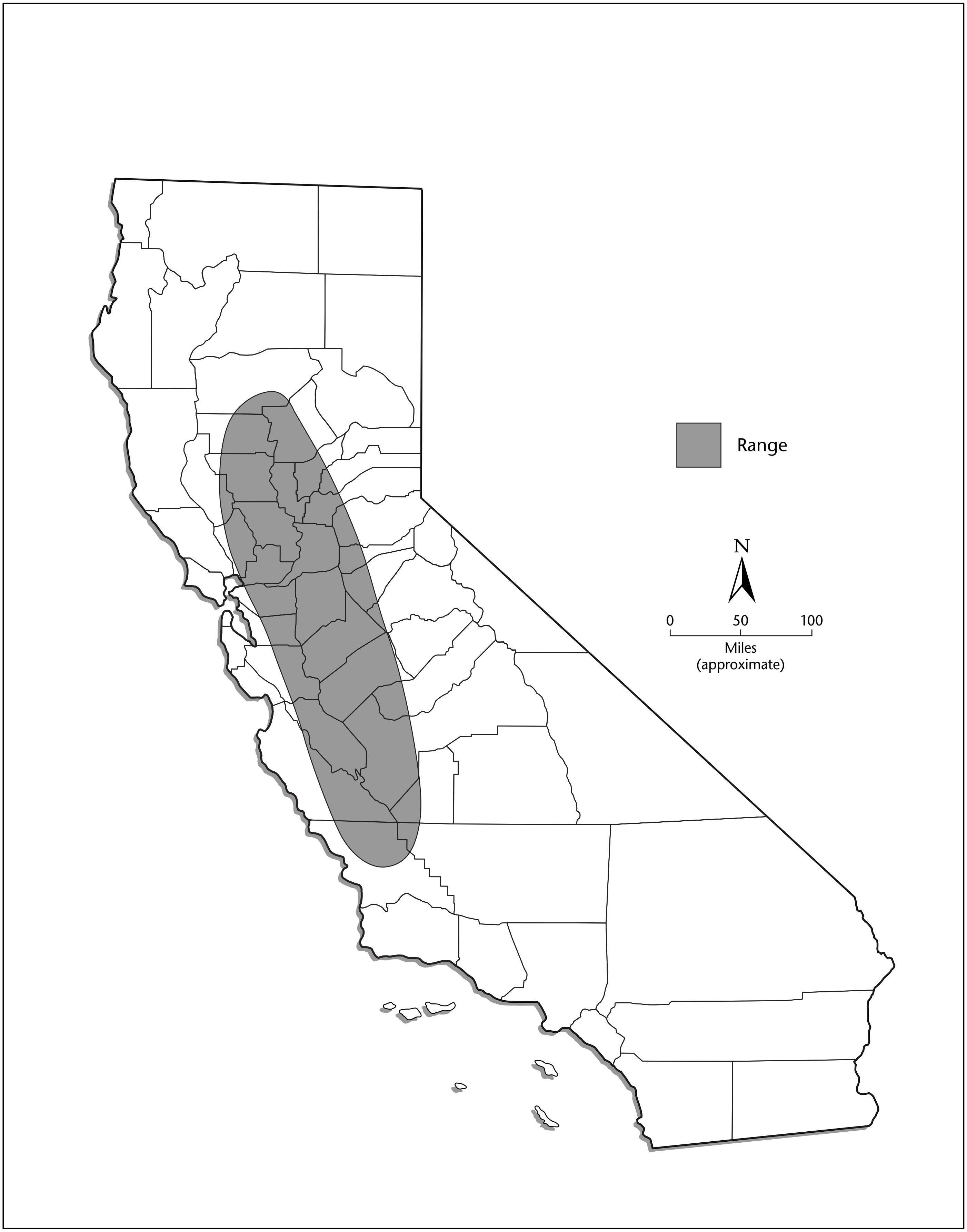

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Jones & Stokes

Prostrate Navarretia *(Navarretia nigelliformis* ssp. *nigelliformis)*
Habitat Distribution