Appendix D

Species Accounts

Appendix D. Species Accounts

February 2016

The species accounts provide a summary of the biology of the Covered Species addressed in the Placer County Conservation Program, Western Placer County HCP/NCCP:

Birds
1. Swainson's hawk
2. California black rail
3. Western burrowing owl
4. Tricolored blackbird

Reptiles
5. Giant garter snake
6. Western pond turtle

Amphibians
7. Foothill yellow-legged frog
8. California red-legged frog

Fish
9. Steelhead
10. Chinook salmon

Invertebrates
11. Valley elderberry longhorn beetle
12. Vernal pool fairy shrimp
13. Vernal pool tadpole shrimp
14. Conservancy fairy shrimp

Each species account contains a description, an envirogram, and a map of the species occurrence and modeled habitat in Western Placer County.

The <u>description</u> typically presents information on:

Regulatory Status
Distribution
Population status and trends Natural
history
Threats
Context for a regional conservation strategy Modeled
species distribution in the Plan Area References

The <u>envirograms</u> use a flowchart to show the most important ecological factors that affect a population or group of populations of a particular species. See description below.

The <u>maps</u> show known occurrence records and modeled potential habitat. The occurrence data and the methodology for habitat modeling is described in Chapter 3, Section 3.3.2 *Covered Species*. Most Covered Species are associated with one or more land cover types. Land cover associations and spatial habitat features (e.g., elevation, proximity to other land cover types) were used to develop habitat distribution models. The models are an approximation: not all of a land cover type will include the specific habitat requirements for a species, and conversely, habitat may be present in small-scale landscape features that were not mapped.

Envirograms

Envirograms were created for each species from the information contained in the species accounts. The envirograms are included as a component of the Covered Species accounts at the recommendation of the Report of the Science Advisors for the Placer County Natural Communities Conservation Plan and Habitat Conservation Plan, *Planning Principles, Uncertainties, and Management Recommendations*, January 8, 2004. The envirograms were prepared under the direction of Peter Brussard, PhD, University of Nevada, Reno, chairman of the science advisors. The following description is adapted from that report:

An envirogram is a tool that sharpens our understanding of the most important ecological factors that affect a population or group of populations of a particular species. The concept was developed originally by Andrewartha and Birch (1984), and envirograms were first applied to conservation planning by James et al. (1997) who used them to identify factors limiting the abundance of endangered Red-cockaded Woodpeckers in the southeastern United States. The version described below is modified somewhat from these previous applications.

An envirogram consists of a “centrum,” components of the environment that directly affect a species’ chances to survive and reproduce, and several “webs,” distal factors that act in sequence to affect the proximate components of the centrum. The centrum consists of four major categories, resources, reproduction, hazards, and dispersal. Each of these can be subdivided as necessary. For example, resources could be subdivided into foraging habitats, breeding habitats, and food; reproduction could be divided into finding mates, nesting, and fledging. Hazards can be divided into predators (an animal that consumes the subject species in whole or part) and “malentities” (organisms or events that can adversely influence the subject species in other ways such as a cow stepping on a dispersing western spadefoot or the premature drying of a vernal pool). Dispersal also can be subdivided since it can occur at different times in a species’ life cycle and it can be either local (such as moving from one habitat type another) or long-distance.

The web identifies the underlying ecological processes or human actions that influence each centrum component. The idea is that distal factors in the web flow in to activate proximate components of the centrum. Each of these flows is called a pathway. Pathways in the web are constructed from right to left, with Web-1 factors directly affecting centrum components, Web-2 factors affecting Web-1 factors, and so on. It is usually unnecessary to have more than three webs to track a centrum component along a pathway to its ultimate underlying influence.

A web factor can have both positive and negative aspects. For example, precipitation is critical to vernal pools. Too little rain results in pools that dry up before their dependent species can complete their life cycles, but greater than average rainfall can result in flooding and dispersal of individuals among pools—an event necessary for gene flow and to replenish dwindling populations.

The centrum components of the envirograms should be accurate reflections of the information in the species profiles, and the web pathways should be logical linkages of indirect environmental components to the proximate drivers of population processes in the centrum. Envirograms are not intended to be stand-alone documents but should be used in conjunction with species profiles and maps showing the distribution of populations and suitable habitat. They are considered to be “works in progress” and always can be modified by new and better information.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Swainson's Hawk
(*Buteo swainsoni*)

Status

Federal: Bird Species of Conservation Concern; Federal Migratory Bird Treaty Act

State: Threatened

Critical Habitat: Not Applicable (N/A)

Recovery Plan: N/A

Distribution

North America

Swainson's hawk inhabits grasslands, sage-steppe plains, and agricultural regions of western North America during the breeding season and winters in grassland and agricultural regions from Central Mexico to southern South America (Woodbridge et al. 1995a; Bechard et al. 2010). The North American breeding range extends north from California to British Columbia east of the Sierra Nevada and Cascade Ranges, east to Saskatchewan, and south to northern Mexico (Bechard et al. 2010). Small numbers also breed in interior valleys of British Columbia (Campbell et al. 1990 as cited in Bechard et al. 2010). Several disjunct populations occur throughout the breeding range; these include populations in Alaska, western Missouri, and the Sacramento and San Joaquin Valleys, as well as the valleys of the Sierra Nevada, in California (Bechard et al. 2010).

Swainson's hawk is a long distance migrant. The majority of the population winters in South America, primarily on the Argentine pampas. It appears, however, that the California population is distinctive in that it winters in Mexico, Central America, and Columbia, although a few have been discovered spending a portion of the winter in the Sacramento–San Joaquin Delta in the last decade (Bechard et al. 2010). They are also regular, but uncommon, in Florida in the winter (Bechard et al. 2010).

California

In California, the nesting distribution includes Great Basin sage-steppe communities and associated agricultural valleys in extreme northeastern California, isolated valleys in the Sierra Nevada in Mono and Inyo Counties, the Sacramento and San Joaquin valleys, and at least one known isolated breeding site in the Mojave Desert. The majority of Swainson's hawks in California nest in Sacramento, San Joaquin, and Yolo counties with Solano, Merced, Stanislaus, Sutter, Glenn, and Colusa counties all important to the central range of the bird (Bradbury 2009). Also important are the Swainson's hawks found in Owens Velley and Klamath Basin, though these are considered part of the Great Basin population since they nest east of the Sierra crest (Bradbury 2009). The historic breeding distribution also included much of southern California, particularly the inland valleys, where the species was once considered common (Sharp 1902; Bent 1937; Bloom 1980). Breeding populations in California have been extirpated or are nearly extirpated from coastal southern California, the Mojave Desert, and the central Coast Ranges (Bloom 1980).

Placer County Plan Area

Historical

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

Current

Swainson's hawk nests and forages primarily in the valley portion of the Plan Area, below 400 feet elevation. Swainson's hawk has a very patchy distribution in western Placer County. There are seventeen relatively recent records (one in 1996, the rest between 2001 and 2014) of nesting in the Plan Area (CNDDB 2015; Moeszinger 2014).

Population Status & Trends

North America

Partners in Flight estimated 460,000 Swainson's hawks in North American (Rich et al. 2004). As many as 845,000 migrants have been counted over Panama City, Panama, during migration. Population declines have been noted in several portions of the species' range, and the current range-wide population is likely reduced from historic times (Bechard et al. 2010).

California

Early accounts described Swainson's hawk as one of the most common nesting raptors in California, occurring throughout much of lowland California (Sharp 1902). Bloom (1980) estimated as many as 17,136 pairs of Swainson's hawks historically nested in California. Knowledge that an estimated 91% decline in the breeding population had occurred (Bloom 1980) led the California Fish and Wildlife[1] to designate the Swainson's hawk as threatened in 1983 (Estep 1989). Since the mid-1800s, native habitats have undergone a gradual conversion to agricultural use. Today, few native grasslands remain in the state, and only remnants of the formerly extensive riparian forests and oak woodlands still exist (Katibah 1983). This habitat loss has caused a substantial reduction in the breeding range and the size of the breeding population in California (Bloom 1980; Bechard et al. 2010). Swainson's hawk is also sensitive to habitat fragmentation (Estep and Teresa 1992).

The state supported 1,770 – 2,393 breeding pairs in 2005 and 2006, with about 95% of the state's population breeding in the Central Valley (Anderson et al. 2007). Breeding populations in California have

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[1] As of January 1, 2013, the California Department of Fish and Game (CDFG) was renamed the California Department of Fish and Wildlife. When this document cites reports prepared by the Department prior to 2013, the reference includes the prior department name of CDFG. Both CDFW and CDFG refer to the same agency.

been extirpated or are nearly extirpated from coastal southern California, the Mojave Desert, and the central Coast Ranges (Bloom 1980).

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

Placer County Plan Area

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

Natural History

The habitat requirements, ecological relationships, life history, and threats to Swainson's hawk described below are summarized in diagram form in Envirogram 1 Swainson's Hawk.

Habitat Requirements

Swainson's hawk is typically present in California from early March, when individuals arrive on breeding grounds, through mid-October, when birds have departed for wintering grounds in Central and South America. In California, Swainson's hawk habitat generally consists of large, flat, open, undeveloped landscapes that include suitable grassland or agricultural foraging habitat and sparsely distributed trees for nesting (Bechard et al. 2010).

Swainson's hawk usually nests in large, native trees such as valley oaks (*Quercus lobata*), cottonwoods (*Populus fremontia*), and willows (*Salix* spp.), although nonnative trees such as eucalyptus (*Eucalyptus* spp.) are also used (Bechard et al. 2010). Nests occur in riparian woodlands, roadside trees, trees along field borders, isolated trees, small groves, trees in windbreaks, and on the edges of remnant oak woodlands (Bechard et al. 2010). Nesting areas are within easy flying distance to alfalfa or hay fields. In some Central Valley locales, urban nest sites have also been recorded (England et al. 1995) and a small number of nests have been reported on human-built structures, such as power poles or transmission towers (James 1992). Stringers of remnant riparian forest along drainages contain the majority (87%) of known nests in the Central Valley (England et al. 1995; Schlorff and Bloom 1984). In the Sacramento Valley and Sacramento River Delta in California, most nests were recorded on the flat valley floor (e.g., Yolo, Sacramento, San Joaquin, and Solano counties), with fewer nests located along the margins of the valley (Gifford et al. 2012). Searches for nests above 500 feet in elevation located only a single Swainson's hawk nest (Gifford et al. 2012). Nests are constructed using materials from the nest tree or nearby trees, are up to 24 inches in diameter, and are usually constructed as high as possible in the tree, providing optimal protection and visibility from the nest (Bechard et al. 2010). Nests appear more flimsy or ragged than that of other buteos (Bechard et al. 2010). Some nests are used for more than one year by the same pair or refurbished nests of other Swainson's hawks or avian species (e.g., American crow [*Corvus brachyrhynchos*], common raven [*Corvus corax*], black-billed magpie [*Pica pica*]); however, the majority of nests are likely freshly built (Fitzner 1978; Bechard et al. 2010).

Populations in the Great Basin often use juniper trees (*Juniperus* sp.) for nesting (Bechard et al. 2010), and at least three known nest sites in the Mojave Desert are in Joshua trees (*Yucca brevifolia*) (CNDDB 2015).

Nesting pairs in California have high fidelity to nesting territories and nesting trees (Fitzner 1980; Bechard et al. 2010). Many nest sites in the Sacramento Valley have been occupied annually since 1979 (Estep in prep.), and banding studies conducted since 1986 confirm a high degree of nest and mate fidelity (Estep in prep.).

Swainson's hawk requires wide-open landscapes for foraging. Historically, the species used grass-dominated and desert habitats throughout most of lowland California. Over the past century, conversion of much of the historic range to agricultural use has shifted the nesting distribution into open agricultural areas that mimic grassland habitats or otherwise provide suitable foraging habitat. Agricultural uses that provide suitable foraging habitat include a mixture of alfalfa and other hay crops, grain, row crops, and lightly grazed pasture with low-lying vegetation that support adequate rodent prey populations (Estep 1989; Bechard et al. 2010).

Telemetry studies have demonstrated that individual Swainson's hawks may require in excess of 15,000 acres of foraging habitat or range up to 18 miles from their nest in search of prey (Estep 1989). Other estimates indicate that under optimal conditions, individual nesting pairs require a minimum of approximately 741 acres of suitable foraging habitat; however, foraging ranges are geographically and temporally variable and are dependent largely on cover type and phenology and their relationship to prey availability (Fitzner 1978; Bechard 1982; Estep 1989; Babcock 1995). Agricultural landscapes that consist of a variety of seasonal crops with different planting, growth and harvest regimes, along with a patchwork of perennial cover types (e.g., alfalfa, irrigated pasture, annual grasslands) provide a relatively constant source of suitable foraging habitat for Swainson's hawks throughout the season (Estep 2009). Research in the Central Valley funded by the California Department of Fish and Wildlife identified the following preferred foraging habitats (Estep 1989).

1. Alfalfa: provides a relatively low abundance of prey at a steady rate of accessibility throughout the breeding season (March to September).
2. Fallow fields: provide a high abundance of accessible prey if such fields are not dominated by dense stands of thistle and other weedy vegetation.
3. Beet and tomato fields: provide the largest prey populations, but dense cover reduces accessibility of prey to foraging Swainson's hawk, except during harvesting operations when Swainson's hawk has been observed foraging almost exclusively in these fields (late-July to early-September).
4. Dry-land pasture: may provide primary foraging habitat for some individuals.
5. Irrigated pasture: provides suitable foraging habitat, especially during flooding.

Habitats unsuitable for foraging include any crop where prey are not available due to the high density of vegetation, or have low abundance of prey (i.e., flooded rice fields, mature corn, orchards, and cotton fields).

Reproduction

Most birds apparently do not breed until they are at least 3 years of age (J.K. Schmutz pers. comm. as cited in Bechard et al. 2010). In the Central Valley, Swainson's hawk arrives on the breeding grounds from early March to early April, significantly earlier than most other populations (Bechard et al. 2010). Pair bonding begins immediately and involves courtship displays, reestablishment of territorial boundaries, and nest construction or repair (Bechard et al. 2010). One to four eggs are usually laid in early to mid-April, and incubation continues for 34–35 days until mid-May when young begin to hatch. The brooding period typically continues through early to mid-July when young begin to fledge (Bechard et al. 2010). Nestlings fledge on average at 43 days (range 38–46 days) (Olendorff 1973; Fitzner 1978; Bechard et al. 2010). Studies conducted in the Sacramento Valley indicate that one or two (occasionally three) young typically fledge from successful nests, with an average of 1.6 young per successful nest (England et al. 1995; Estep in prep.). Reproductive success in California was found to be inversely correlated with distance to suitable foraging habitat (Woodbridge 1991; England et al. 1995). After fledging, young remain near the nest and are dependent on the adults for approximately 4 weeks, after which they permanently leave the breeding territory (Anderson et al. in prep.). By mid-August, breeding territories are no longer defended, and Swainson's hawks begin to form premigratory communal groups.

Dispersal Patterns

Woodbridge et al. (1995b) noted an average dispersal distance of 5.5 miles between natal sites and subsequent breeding sites in northeastern California. However, during the study period, one bird bred approximately 23 miles from its' natal site. In the Sacramento Valley, two birds banded as nestlings and subsequently resighted as breeding adults nested within 2.2 miles of their natal site (Estep 1989). Much greater dispersal distances from natal sites have been observed in other parts of the range, most notably distances up to 193 miles in Saskatchewan (Houston and Schmutz 1995). Briggs et al. (2012) found that natal dispersal in their study ranged from 0.6 to 17 miles for males and 0.1 to 28 miles for females. Therefore, female Swainson's hawks were found to disperse significantly farther than males (Briggs et al. 2012). Natal dispersal was negatively correlated with primary productivity and positively correlated with population density around the nest site (Briggs et al. 2012).

A high degree of nest site fidelity has been noted in Swainson's hawk in California. Individuals often use the same nest, the same tree, or a nearby tree in subsequent years (Fitzner 1980; Bechard et al. 1980). In the Sacramento Valley, mean inter-territory adult movement was approximately 328 feet (Estep in prep.). Less nest site fidelity was noted in northeastern California, where mean inter-territory movements between 1984 and 1994 were 1.4 miles (Woodbridge et al. 1995b).

Home range size of breeding adults varies greatly (Bechard et al. 2010). Larger home ranges are found in areas with crop types unsuitable for foraging, such as mature grains and row crops, orchards, and vineyards (Bechard 1982; Estep 1989). The smallest home ranges were reported at nest sites near alfalfa, fallow fields, and dry pastures (Bechard 1982; Estep 1989; Woodbridge 1991). A telemetry study to determine foraging requirements has shown that Swainson's hawks may forage up to 19 miles from the nest site and may use in excess of 15,000 acres habitat for foraging (Estep 1989). Home range size fluctuates throughout the breeding season as the foraging landscape changes (Estep 1989; Estep 2009).

Longevity

Very limited data are available on Swainson's hawk survivorship. In northeastern California, the mean age for hawks banded as nestlings in 1980–1992 and observed in 1993–1994 was 8.2 years (n = 36)

(Woodbridge et al. 1995b). In the Sacramento Valley, the mean age for hawks banded as nestlings in 1980 and observed in 1988–1995 was 8.8 years (n = 5); the oldest was 13 years (Estep in prep.). The oldest male in the banding records to date was a male banded by Peter Bloom in California and retrapped by Brian Woodbridge 24 years later; the longest-lived female was at least 21 years old, having been banded at the age of 2 (Bechard et al. 2010). Distance to agriculture and amount on agriculture in a territory are good predictors of apparent survival for Swainson's hawks (Briggs et al. 2011). Individuals that nested farther from agriculture had decreased nest success, suggesting that the further individuals had to travel for prey the greater the energetic costs incurred. Amount of agriculture in a territory was positively correlated with annual apparent survival (Briggs et al. 2011). Increased agriculture (particularly alfalfa) likely provides increased foraging opportunity and capture success, allowing individuals to spend less time foraging and more time engaged in activities that enhance survival (e.g., caloric intake, resting, ectoparsite removal) (Briggs et al. 2011).

Sources of Mortality

There is no information on predation of adult Swainson's hawks; however, adults have been reported to be killed on highways, shot, or killed in collisions with vehicles (Bechard et al. 2010). Nestlings are susceptible to predation by great horned owl (*Bubo virginianus*), American crow, and various mammalian predators (Dunkle 1977; Woodbridge 1991; Estep in prep.). Large die-offs of adult birds have been documented in Argentina on the wintering grounds following large-scale applications of insecticides (Woodbridge et al. 1995a).

Behavior

There are no data available on the size or characteristics of breeding territories; however, it has been noted that Swainson's hawk aggressively defends the area immediately surrounding nest sites (Rothfels and Lein 1983; Janes 1984; Fitzner 1978). Outside this relatively small area they appear more tolerant, and often forage communally with conspecifics and other buteos (Bechard et al. 2010; Estep 1989). Once young have fledged, adults begin to form communal foraging and premigratory groups and exhibit little territorial behavior.

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

During the breeding season, Swainson's hawk feeds primarily on small rodents, including voles (*Microtus* sp.), deer mice (*Peromyscus* sp.), house mice (*Mus musculus*), and pocket gophers (*Thomomys* sp.). Other, less frequent food items include reptiles, birds, and insects. Swainson's hawk typically forages in large fields that support low vegetative cover (to provide access to the ground) and provide the highest densities of prey (Bechard 1982; Estep 1989). In agricultural regions, these habitats include fields of hay and grain crops; certain row crops, such as tomatoes and sugar beets; and lightly grazed pasturelands. Fields lacking adequate prey populations (e.g., flooded rice fields) or those that are inaccessible to foraging birds (e.g., vineyards and orchards) are rarely used (Estep 1989; Babcock 1995).

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

During migration, Swainson's hawks may congregate in large groups (up to 100 or more birds) (Bechard et al. 2010). During this time, Swainson's hawks feed in grasslands and harvested fields, especially where grasshoppers (*Dichroplus* spp.) are numerous. They often perch on fence posts, telephone poles, and power poles (Bechard et al. 2010). Swainson's hawks exclusively eat insects, such as grasshoppers, dragonflies (*Aeshna bonariensi*), and moths (*Lepidoptera* sp.) in winter (Woodbridge et al. 1995b). Non-breeding Swainson's hawks typically hunt communally and will run or walk to catch prey (Bechard et al. 2010).

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

Movement and Migratory Patterns

In California, Swainson's hawk begins fall migration from late August to late-September (Bloom 1980; Estep 1989; Bechard et al. 2010; Kochert et al. 2011). Satellite radiotelemetry studies from 1995 to 2001 have identified migratory routes, timing, and wintering grounds (Woodbridge et al. 1995a). According to these and other telemetry studies, all but the Central Valley population migrates along the eastern edge of Mexico through Central and South America and winters in the pampas region of Argentina. Unlike other populations of Swainson's hawk, the Central Valley population winters primarily in Central Mexico and, to a lesser extent, throughout portions of Central and South America (Bradbury2009). Swainson's hawks' northward migration largely follows the southward route (Bechard et al. 2010). Swainson's hawks begin migrating north from mid-February through March (Kochert et al. 2011). Southbound migrations last 42 to 98 days and northbound migrations last 51 to 82 days (Kochert et al. 2011). In California, breeding adults arrive at the nesting territory from approximately early March to early April. Courtship and nest construction begin immediately upon arrival.

Ecological Relationships

Swainson's hawk is territorial during the breeding season; however, away from the nest sites adults are more tolerant of conspecifics and other raptors. During the prenesting period, adults are highly aggressive around the nest as they reestablish their territorial boundaries. During communal foraging events and from postfledging through migration and wintering periods, adults are gregarious and tolerate conspecifics as well as other raptor species (Fitzner 1978; Estep 1989; Bechard et al. 2010). Because Swainson's hawk generally arrives at the breeding grounds later than other sympatric buteos, individuals are often engaged in congeneric battles over control of nest sites.

Threats

The loss of agricultural lands and native grasslands to various residential and commercial developments is a serious threat to Swainson's hawks throughout California (Estep 2008; Bradbury 2009). Additional threats are habitat loss caused by riverbank protection projects; conversion from agricultural crops that provide abundant foraging opportunities to crops such as vineyards and orchards, which provide fewer foraging opportunities; shooting; pesticide poisoning of prey animals and hawks on wintering grounds; collision with stationary objects; competition from other raptors; and human disturbance at nest sites (California Department of Fish and Game 2000; Bechard et al. 2010).

Even though Swainson's hawks prey on agricultural pests, they were historically considered a varmint by many ranchers and farmers, until at least the late 1930s (Bechard et al. 2010). As a result, Swainson's hawks were often shot. Banding recoveries suggest that mortality resulting from shooting on breeding grounds has declined and may not be significant (Houston and Schmutz 1995).

Acute toxicity from poisoning by organophosphate insecticides (e.g., monocrotophos and dimethoate), used to control grasshopper outbreaks in alfalfa and sunflower fields, caused the death of nearly 6,000 Swainson's Hawks in Argentina in 1995 and 1996. Overall, an estimated 20,000 Swainson's hawks were killed in Argentina by pesticide applications. Deaths resulted immediately after hawks were sprayed directly by pesticide applicators while they foraged in fields or within several days after they ate poisoned grasshoppers (Woodbridge et al. 1995a). Since Central Valley Swainson's hawks are not known to migrate to the affected areas in Argentina, it is thought that the poisoning events did not affect this population (Bradbury 2009).

Houston and Schmutz (1995) discovered Swainson's hawks throughout the breeding, wintering, and migratory range died from collisions with stationary/moving objects or structures, such as cars, trains, powerlines, and fences.

Context for a Regional Conservation Strategy

There is little information on the distribution and density of Swainson's hawks in western Placer County. The species does nest in limited numbers in the Plan Area; since 1996, there have been seventeen active nests recorded. The species requires large, open landscapes that include suitable grassland or agricultural foraging habitat and sparsely distributed trees for nesting. In the region, Swainson's hawk is found primarily to the west of Placer County in the Sacramento and Central Valleys, which represents the greatest distribution of the species in California. Records of the bird are absent from counties directly north and south of Placer County, such as Nevada and El Dorado counties. The species has also been recorded to a lesser extent in the north in Siskiyou, Modoc and Lassen counties, and to the south in Mono and Inyo counties. As records of Swainson's hawk are abundant in counties west of Placer County, the population within the Plan Area is not of particular significance statewide. However, as the species is limited in its nesting distribution in western Placer County, and as nest fidelity is common, protection of individual nesting sites is of high priority in the conservation of Swainson's hawk. Also of conservation priority within the Plan Area is acquiring suitable foraging habitat.

Modeled Species Distribution in the Plan Area

Model Assumptions

<u>Nesting Habitat</u>

Swainson's hawk nesting habitat includes riverine/riparian, valley oak woodland, and eucalyptus land-cover types in the Valley floor below 200 feet elevation. The nesting habitat model does not capture single or small patches of trees, which is potentially suitable nesting habitat when it occurs amongst suitable foraging habitat.

Foraging Habitat

Swainson's hawk foraging habitat is defined by vernal pool complex, annual grassland, pasture, alfalfa, irrigated pasture and row crop land-cover types. Foraging habitat is also restricted to the Valley floor (< 200 feet elevation).

Rationale

In the Central Valley, Swainson's hawks generally nest in open terrain in large, native trees such as valley oaks, cottonwoods, and willows, although nonnative trees such as eucalyptus are also used. Nests occur in riparian woodlands and the edges of remnant oak woodlands. Swainson's hawks also nest in isolated, large, sparsely distributed trees along field borders in other open land-cover types such as annual grassland, along roadsides and agricultural fields. Potential nest sites that occur in isolated stands or individual trees in open terrain (e.g., grasslands, agricultural lands) are not captured in the habitat model as their spatial extent is considerably smaller than the corresponding land-cover mapping unit. Consequently, this model may not encompass every nesting site; however, the extent of nesting habitat not captured by the model is relatively small compared to the extent of nesting habitat that is captured by the model.

Foraging generally occurs within 10 miles of active nest sites; however, as nest site locations will vary throughout the 50 year term of the permit, foraging habitat was modeled to include suitable land-cover types throughout the entire area in the Valley floor (< 200 feet elevation) encompassed by these land-cover types.

Model Results

Species Map 1. *Swainson's Hawk Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for Swainson's hawk within the Plan area. Modeled nesting habitat is primarily restricted to valley foothill riparian along the Bear River, Coon Creek, Markham Ravine, Auburn Ravine, Pleasant Grove Creek, and Dry Creek. Scattered, open-canopy woodlands (i.e., valley oak woodland, oak woodland savanna, rural residential, and eucalyptus groves) comprise the remaining modeled nesting habitat. Many other sites throughout the Valley landscape may also provide suitable nesting habitat in the form of small woodlands and isolated trees. These areas, however, could not be identified in this model because these small-scale features were not mapped. In some cases, precise locations of nests did not occur on modeled primary habitat because the nest trees were likely isolated and not mapped as part of a primary habitat land-cover type.

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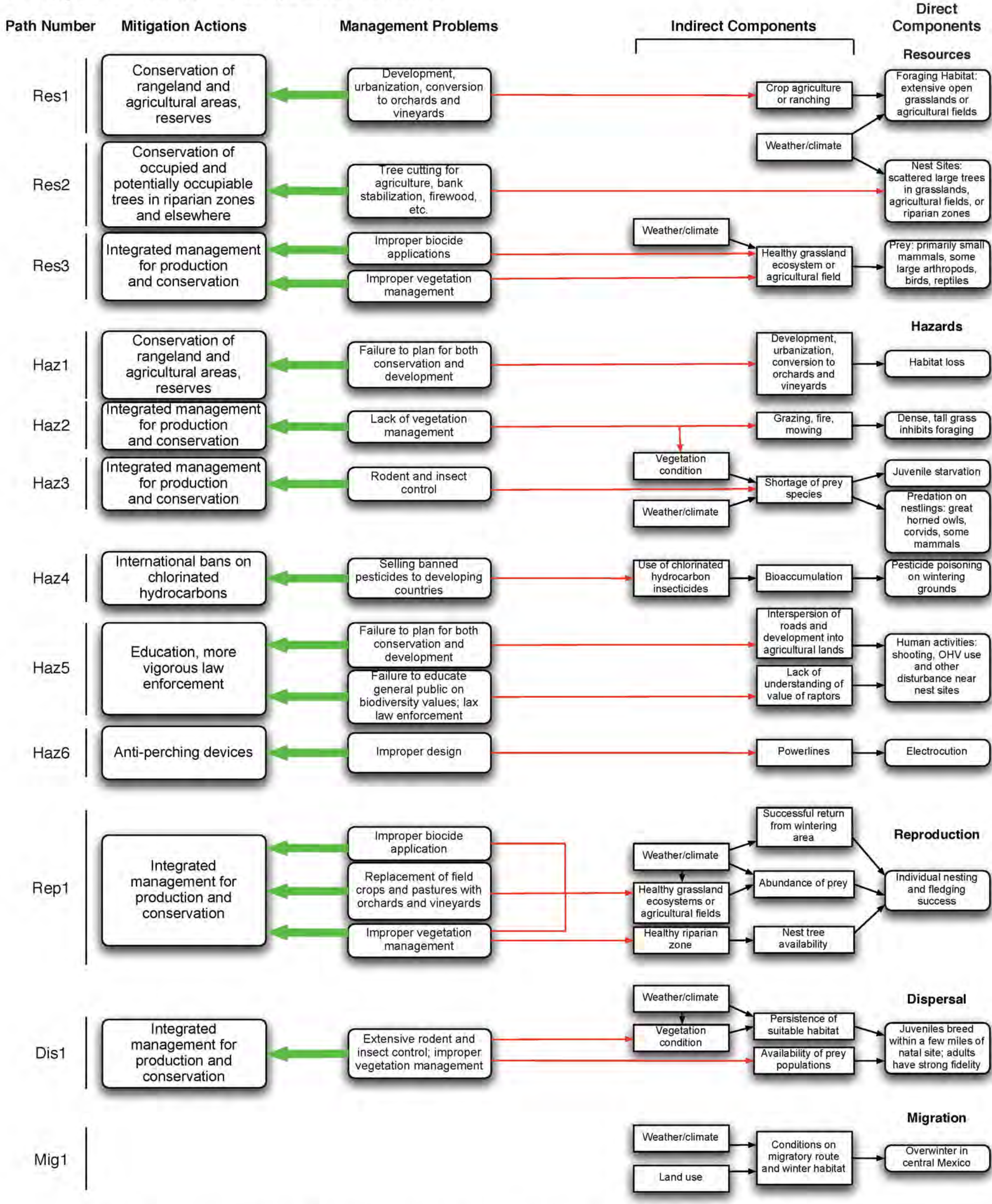
Envirogram 1 Swainson's hawk, *Buteo swainsoni*



Envirogram 1 Swainson's Hawk. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal; Mig = Migration.

Envirogram Narrative

Swainson's Hawk (*Buteo swainsoni*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary. For example, resources are subdivided into foraging habitat, nest sites, and prey.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Swainson's hawks rely upon extensive grasslands or agricultural fields for foraging habitat. In Placer County, this habitat is provided largely by cropland, annual grassland, and pasture. Various types of development and the conversion of fields and pastures to orchards and vineyards has diminished habitat for the hawk. Policies favorable to agricultural conservation help mitigate this loss.

Res2: Swainson's hawks nest in scattered large trees, mostly on the edges of riparian zones in Placer County. Cutting of trees for firewood, lumber, or other purposes diminishes the number of nest sites available; occupied and potential nest trees must be protected. Weather conditions and climate trends link both to foraging habitat and nest sites.

Res3: Swainson's hawks prey primarily on small mammals, although other small vertebrates and large insects are also taken. Favorable weather and healthy grasslands or agricultural fields are necessary for sufficient prey items to be available and improper vegetation management or excessive biocide application diminishes prey items. Management plans that integrate both agriculture and conservation must be developed.

Hazards

Haz1: Loss of rangeland and agricultural areas to residential and commercial development is probably the major reason that Swainson's hawks are in decline on the breeding grounds. Poor planning in the past can be mitigated to some extent by reserves and conservation of agricultural lands.

Haz2: In the absence of burning or grazing, grass growth makes fields unsuitable for foraging by Swainson's hawk, so vegetation management, such as livestock grazing at the proper time and intensity, is necessary.

Haz3: Adult Swainson's hawks are preyed upon rarely, but starvation and predators may kill nestlings and juveniles. Nest predators include great horned owls, crows and other corvids, and some mammals.

Healthy rodent populations provide alternate prey for these species and thereby lessen predation pressure on Swainson's hawk nestlings as well as help prevent starvation in young hawks. Good vegetation condition and favorable weather, along with management designed for both agricultural production and conservation, can maintain a healthy prey base.

Haz4: Swainson's hawks may be poisoned by or accumulate pesticides on their wintering ground (see dispersal and migration section), although this has not been definitively shown for the California population that winters in central Mexico. While this hazard cannot be controlled by the Placer County Conservation Plan, citizens should be encouraged to educate their legislators about the continuing perils of chlorinated hydrocarbon pesticides to the county's wildlife.

Haz5: Protecting Swainson's hawks from casual shooting and disturbance (e.g., OHV use, hiking) at their nest sites is also important. These situations generally result from people being unaware of the value of raptors and a failure to enforce laws that protect raptors and usually occur when residential areas are in close proximity to nest sites. Better education and more vigorous law enforcement can help alleviate this hazard.

Haz6: Swainson's hawks are subject to electrocution on power lines, which can be alleviated by raptor anti-perching devices. There also are devices that shield the dangerous locations on lines (transformers, etc.).

Reproduction

Rep1: The nesting success of individual pairs depends upon successful return from the wintering area, prey availability, and the availability of nest trees. Weather conditions affect successful migration, the state of the ecosystem on which the prey base depends, and the prey base directly. A sufficient prey base requires large expanses of foraging habitat, depending upon the type of vegetation and prey abundance. The replacement of forage and row crops by orchards and vineyards is very detrimental to Swainson's hawk foraging. Prey availability also is influenced by rodent and insect control activities and by the destruction of habitats that support prey species. Nest tree availability depends on conditions in and adjacent to riparian zones. All these problems can be addressed through management plans that integrate agricultural production and biodiversity conservation at a landscape scale.

Dispersal

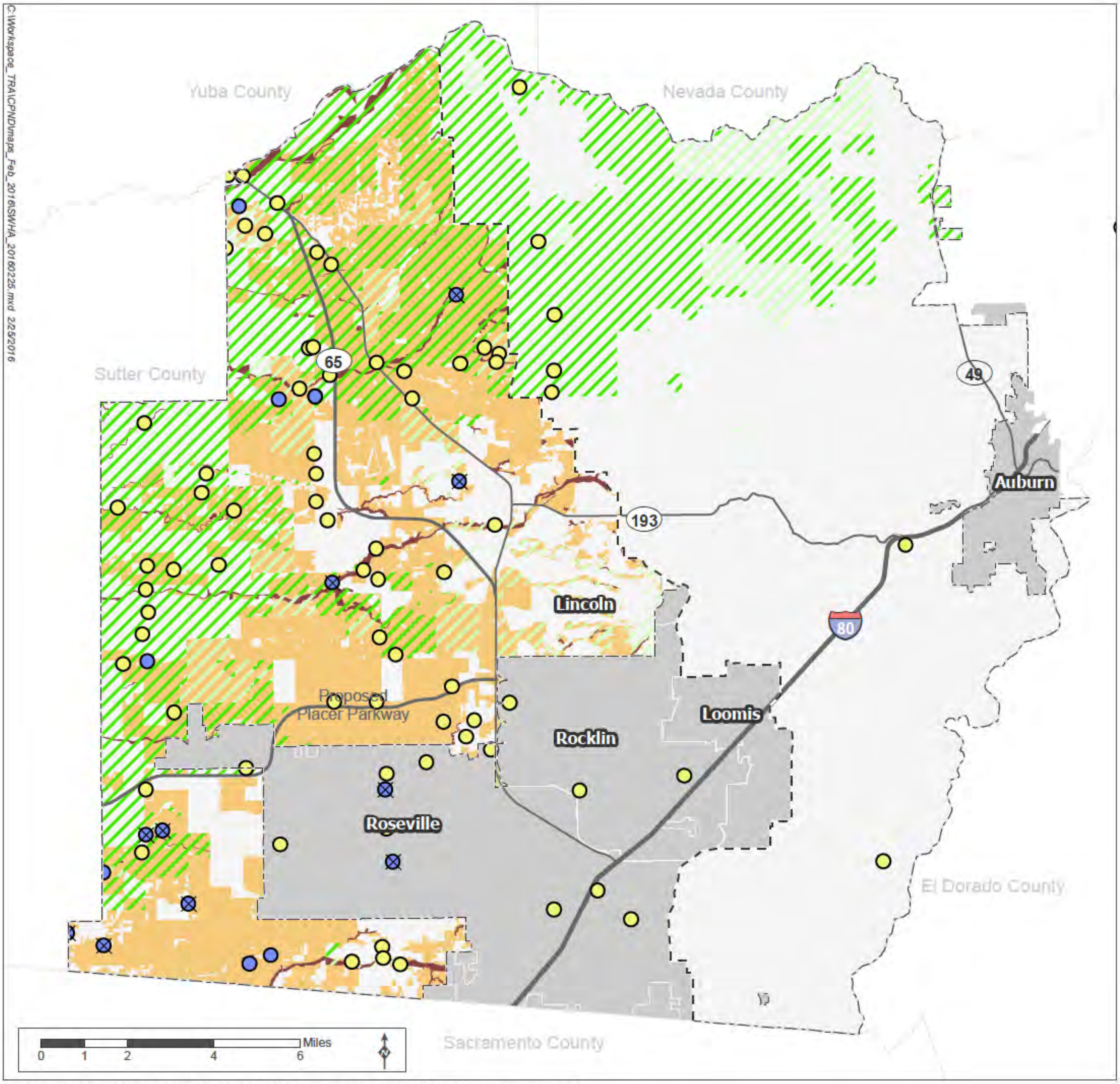
Dis1: Adult Swainson's hawks are highly philopatric, returning to their old nest sites every year, and young Swainson's hawks usually next in close proximity to their natal sites provided that suitable habitat and a prey base are still present. While this depends to some extent on weather conditions, it is also influenced by improper pesticide and vegetation management practices. Integrating management for both agriculture and conservation can help mitigate these problems.

Migration

Mig1: The California population of Swainson's hawk overwinters in central Mexico (although a few individuals may remain in California). The weather and habitat conditions along their migratory route and in their wintering habitats are critical to their survival. Chlorinated hydrocarbon pesticides applied to crops in the wintering areas may be a problem (see Haz5).

Summary

As predators at the top of the grassland food web, Swainson's hawks are highly sensitive to ecosystem conditions and require large expanses of foraging habitat. These factors suggest that the best strategy for Swainson's hawk conservation is to manage riparian zones to provide adequate nesting trees and secure easements on large acreages of agricultural lands so they will be managed for both biodiversity conservation and sustainable agriculture. The Swainson's hawks' requirements also are such that they are very compatible with large vernal pool-grassland ecosystem reserves provided that nest trees are available and disturbance is minimized especially during breeding.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2015; Placer Land Trust, 2010; eBird, 2015; Patrick Moeszinger, 2014

Occurrences
- Nest Site - Precise
- Nest Site - General
- eBird and Other Sources

Modeled Habitat
- Nes ing Habitat
- Foraging Habitat
- Non-habitat

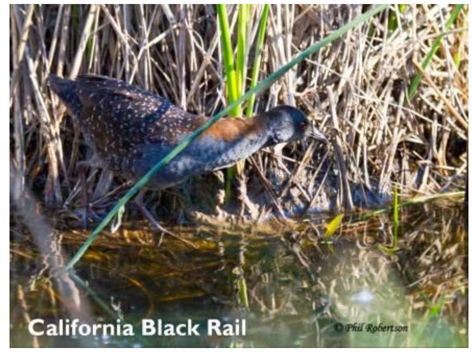
- Existing Protected Area
- Reserve Aquisition Area
- Non-participating City

- Major Road
- Valley/Foothill Divide
- Area A Boundary

MIG | TRA

Species Map 1.
Swainson's Hawk Modeled Habitat Distribution and Occurrence
Placer County Conservation Program – Western Placer County HCP/NCCP

California Black Rail
(*Laterallus jamaicensis coturniculus*)



© Phil Robertson

Status

Federal: Bird Species of Conservation Concern; Migratory Bird Treaty Act

State: Threatened; Fully Protected

Critical Habitat: Not Applicable (N/A)

Recovery Plan: **N/A**

Distribution

North America

The California black rail is one of two subspecies of black rail that inhabit North America including the Eastern black rail (*Laterallus jamaicensis jamaicensis*) and the California black rail. The Eastern black rail breeds primarily along the eastern seaboard from Connecticut south to southern Florida and along parts of the Gulf Coast. There are scattered small populations in the Midwest to the southern Great Plains, and interior North Carolina to northern Georgia (Eddleman et al. 1994). The California black rail breeds in the foothills of the Sierra Nevada, coastal California, northwestern Baja California, the lower Imperial Valley, and the lower Colorado River in Arizona and California (Aigner et al. 1995 ; Eddleman et al. 1994; Richmond et al. 2008).

California

California black rail populations were previously thought to be restricted to the San Francisco Bay Area, Bolinas Lagoon, Tomales Bay, Morro Bay, Suisun Bay, the Delta region to White Slough in San Joaquin County, the Salton Sea area, and the Lower Colorado River Valley (Grinnell and Miller 1944; Manolis 1978; Garrett and Dunn 1981; Evens et al. 1991; Eddleman et al. 1994). In 1994, however, populations were discovered in the western Sierra Nevada foothills of Yuba County (Aigner et al. 1995), and subsequent surveys revealed previously unknown populations in the foothills of Butte, Nevada, Placer, and San Joaquin Counties (Richmond et al. 2008). Genetic analysis suggests that California black rail was historically present in the foothills, and that the foothills population had gone undiscovered until recently, rather than this recent discovery reflecting a recent colonization of the foothills (Girard et al. 2010). As of 2014, California black rail has been found in over 200 wetlands in the foothills of Butte, Nevada, Yuba, Placer, and San Joaquin counties, almost all below 1,155 feet (Tecklin 1999; Richmond et al. 2008; Dudek 2014).

Placer County Plan Area

Historical

Detailed information concerning California black rails in the Sierra foothills is limited. How long they have occupied the area and the extent of their distribution is unknown (Tecklin 2006). There are no historical records of California black rail in the Plan Area.

Current

California black rails have been found in various locations in Nevada County, just north of the Placer County border (Tecklin 1999; Richmond et al. 2008; Dudek 2014). There are apparently earlier Christmas Bird Count records of California black rail presence in Placer County (Tecklin 2006; Dudek 2014), as well as recent verified records of occurrence in Clover Valley near Rocklin, California (Tecklin 2006). California black rail have been detected at numerous locations in Placer County since 2002. Formal and informal exploration and field activities of the Black Rail Project of the University of California, Berkeley (https://nature.berkeley.edu/beislab/rail/html/index.html) since its inception in 2002 has discovered black rail occurrences at several sites on private, developed properties and on lands administered by the Placer Land Trust. These more recent findings have not been published or widely reported. Based on these documented and probable records, it is highly likely that other parts of Placer County could represent an extension of the now well established populations in Butte, Nevada, and Yuba Counties. Confirmed California black rail detections in Placer County are shown in Species Map 2. *California Black Rail Modeled Habitat Distribution and Occurrence* and detailed in Table 1.

Table 1. Confirmed California Black Rail Detections in Placer County

Location	Date	Observer
Doty Ravine Preserve	10/2012	Tecklin and Hall
Swainson's Grassland Preserve	12/2011	Tecklin and Hall
Redwing on Yankee Slough	08/2011	Tecklin and Hall
Sun City Lincoln Hills Preserve (Ingram Slough)	08/2011	Tecklin and Hall
Sun City Lincoln Hills Preserve (12 Bridges Road, Hillside Springs)	08/2011	Tecklin and Hall
Little Ben and Big Ben Intersection	06/2009	Placer County Big Year Detection
Clover Valley	09/2006	Tecklin
Bickford Parking Area	06/2006	Widdowson
Spears Ranch (now Hidden Falls)	05/2005	Garrison
Near Camp Far West Reservoir	04/2003	Sterling

Dudek 2014; CNDDB 2015

Population Status & Trends

North America

Black rail populations have declined throughout the species' range primarily due to habitat destruction (Eddleman et al. 1994).

California

The current distribution of California black rail breeding range has contracted with the loss of wetland habitat (CDFG 1987[1]). California black rail populations have been extirpated from Ventura to San Diego counties (Garrett and Dunn 1981). The bulk of the population (>80%) of California black rail is confined to the northern reaches of the San Francisco Bay estuary, especially the tidal marshland of the San Pablo Bay and associated rivers (Evens et al. 1991). The loss of 95% of marsh habitat in the San Francisco Bay area likely had a substantial effect on California black rail populations. The remaining California black rail populations are small and isolated (Evens et al. 1991). Populations along the Lower Colorado River declined about 30% from 1973 to 1989 (Evens et al. 1991). The Sierra Nevada foothill population was estimated at 125-184 during 1997 and 1998 (Tecklin 1999). Richmond et al. (2008) estimated 734-1,466 individuals in over 200 marshes/freshwater wetlands based on intensive surveys in 1994-2006 of sites in the Sierra Nevada foothills. Occupied wetlands were found in five (Yuba, Nevada, Butte, Placer, and San Joaquin) of the 14 counties surveyed (Butte, Colusa, El Dorado, Glenn, Lake, Nevada, Placer, Sacramento, San Joaquin, Solano, Sutter, Tehama, Yolo, and Yuba).

California black rail in the Sierra Nevada foothills likely exists as a metapopulation (Richmond et al. 2008); a population of populations connected by dispersal across areas that do not provide habitat. Within a metapopulation, local populations at individual marshes can go extinct, whereas unoccupied marsh sites are colonized. This pattern of extinction and colonization was commonly observed in the Sierra Nevada foothills by Richmond et al. (2008) throughout their study.

Placer County Plan Area

Although California black rail populations were discovered in Yuba and Nevada counties in 1994, populations were only recently discovered in Placer County. Therefore there is no information on population trends in the Plan Area (J. Sterling, pers. comm.).

Natural History

The habitat requirements, ecological relationships, life history, and threats to California black rail described below are summarized in diagram form in Envirogram 2 Black Rail.

Habitat Requirements

California black rails inhabit saltwater, brackish water, and freshwater marshes (Grinnell and Miller 1944, Manolis 1978). California black rails found away from coastal estuaries and salt marshes, such as in the Sierra Nevada foothills, are found in perennial wetlands with standing or flowing water dominated by dense vegetation, including rush (*Juncus effusus and J. balticus*) and cattail (*Typha latifolia and T. domingensis*) and often with other associated plants such as bulrush (*Scirpus acutus),* spikerush (*Eleocharis macrostachya*) and dallis grass (*Paspalum dilatatum*)*,* fireweed (*Epilobium ciliatum*)*, and* cutgrass (*Leersia oryzoides*) (Aigner et al. 1995; Tecklin 1999; Richmond et al. 2008; Richmond et al. 2010). California black rails are most often found in wetlands with perennial standing water or flowing water (permanently or semipermanently flooded), although they are occasionally found in drier

[1] As of January 1, 2013, the California Department of Fish and Game (CDFG) was renamed the California Department of Fish and Wildlife. When this document cites reports prepared by the Department prior to 2013, the reference includes the prior department name of CDFG. Both CDFW and CDFG refer to the same agency

wetlands with seasonally flooded, intermittently exposed or saturated water regimes (Richmond et al. 2010). The source of water for the majority of the wetlands inhabited by California black rail in the Sierra Nevada foothills is from intentional and unintentional inputs of irrigation water, with 68% of wetlands primarily fed by irrigation, 22% by springs, 6% by streams, and 4% by rainfall (Richmond et al. 2010). These wetlands are in open grasslands, grazed pastures or oak savannas (Tecklin 1999). California black rails rarely use livestock water ponds (i.e., stock ponds) with narrow fringes of emergent vegetation and mostly deep water (Richmond et al. 2010). California black rails typically occur in the shallowest zones of wetland edges where water depths are less than 1.2 inches. They construct well concealed nests in dense vegetation over moist soil or very shallow water (Eddleman et al. 1994). Plant composition is not as important for California black rail habitat as the appropriate vegetation cover (i.e., high stem density and canopy coverage) (Richmond et al. 2010). California black rail occupancy declines when overgrazing substantially reduces wetland vegetation cover (Richmond et al. 2010). Wetlands in the Sierra Nevada foothills greater than 1 acre are more likely to support populations that persist over time, though California black rail was found in wetlands as small as 0.2 acres (Tecklin 1999; Richmond et al. 2010). Also, California black rail was not found during surveys of roadside ditches that had dense patches of cattails and bulrush (Tecklin 1999).

Reproduction

California black rail lays 3-8 eggs, incubates them for 17–20 days, and probably broods the semi-precocial chicks for several days after hatching (Eddleman et al. 1994). There is little information on parental care after hatching and no information is available on reproductive success and survivorship.

Dispersal Patterns

Relatively little is known about the dispersal patterns of black rail. A radiotelemetry study in Arizona tracked three black rails that were found to move an average of 0.89 miles between breeding seasons (Flores and Eddleman 1991). An analysis of occupancy patterns and metapopulation dynamics using incidence function methods estimated median dispersal ability of approximately 5 miles in the foothills (Risk et al. 2011). Recent genetic research suggests two-way movement of individuals between San Francisco Bay and Sierra Nevada foothill populations, and that more individuals tend to move from the foothills to the San Francisco Bay area than vice versa. This result is surprising considering that black rails are generally thought to be poor fliers (Girard et al. 2010).

Given the metapopulation structure of California black rail in the foothills, young birds likely disperse to seek new sites for colonization if densities in an occupied marsh exceed the habitat’s carrying capacity or if an occupied marsh is degraded. Richmond et al. (2008) observed several cases of rapid colonization within one year of marsh creation. This hypothesis is further supported by records of juveniles from other populations appearing in atypical habitats, records of black rails striking TV towers and buildings, and low recapture rates of banded juveniles compared to those of adults (Eddleman et al. 1994). The likelihood of occupancy of a wetland; however, decreases with distance from an occupied wetland (Richmond et al. 2012).

Longevity

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

Sources of Mortality

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

Behavior

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

Movement and Migratory Patterns

California black rail is mostly resident, although there is some local movement from San Pablo Bay south to the southern San Francisco Bay (Evens et al. 1991). Based on continual presence throughout the year, the Sierra Nevada foothill population is thought to be non-migratory (Richmond et al. 2008).

Ecological Relationships

Black rail occupies marshes with Virginia rail and sora rail (Tecklin 1999) but there is no information on interspecific interactions (Eddleman et al. 1994).

Threats

The primary population threats are destruction, desiccation, flooding, grazing and other forms of degradation of marsh/wetland habitats; development-related increases in predation pressures from domestic cats, herons, egrets, and other predators; and pollution carried by runoff into occupied marshes (Eddleman et al. 1994). At inland sites, agricultural practices, livestock grazing, and urbanization may threaten California black rail. Grazing occurs at 60% of the known wetlands occupied by California black rail in the Sierra Nevada foothills and is the most common threat to those wetlands (Tecklin 1999). California black rail occupancy declines when overgrazing substantially reduces wetland vegetation cover (Richmond et al. 2010; Richmond et al. 2012). Future irrigation practices will play an important role in the quantity and quality of wetland habitat in the Sierra Nevada foothills, as irrigation water is the primary source of water for the Sierra Nevada foothill wetlands inhabited by California black rail (Richmond et al. 2008).

Context for a Regional Conservation Strategy

In California, records of the California black rail are concentrated around the San Francisco Bay area, with a few scattered occurrences in southern California as well. The discovery in 1994 of populations of California black rail in Butte, Yuba and Nevada counties extended the known distribution of this species into the western Sierra Nevada foothills. Although not published or widely reported, formal and informal surveys of Placer County since 2002 through the California Black Rail Study Project associated with the University of California, Berkeley has led to the discovery of California black rail occurrences on several private, developed properties within the Plan Area. Because of the elusive nature of this bird, other yet undetected populations may also be present. The Placer County population is the most southern of the known foothill populations. Indeed, in his report to the California Natural Diversity Database (reported July 18, 2006), Tecklin speculates that “recent verified [California] black rails at northern Placer County sites and unverified detections at nearby locations indicate this is an important southern extension of the patchy inland distribution of the subspecies.”

Due to the species’ rarity within the region and state, protection of existing populations or potential habitat is emphasized. Perennial wetland systems, particularly those dominated by bulrushes and cattails, are of highest conservation or acquisition priority for the maintenance or potential increase of California black rail in the Plan Area. As juveniles may disperse from the wetland/marsh in which they were hatched, the maintenance of large, perennial wetland systems could contribute to population success. A series of wetland systems should be preserved, including sites unoccupied by California black rail, to allow for potential colonization from populations in neighboring counties that host California black rail populations. Metapopulations remain stable when the rate of extinction of populations is balanced by colonization of unoccupied sites. Therefore, it is necessary to protect suitable, but unoccupied, habitat to help maintain population stability and allow for the growth of the metapopulation.

Modeled Species Distribution in the Plan Area

Model Assumptions

Year-round Habitat

California black rail modeled habitat is defined as fresh emergent wetlands greater than 0.2 acres in the Plan Area. The scale of the land-cover data and mapping may be too coarse to specifically identify suitable year-round black rail habitat, but the estimated fresh emergent marsh component of mapped marsh complex land cover type is a reasonable measure of modeled habitat.

Rationale

California black rails are year-round residents in the Sierra Nevada foothills and Central Valley. They remain in fresh emergent wetlands year round, except to disperse to other fresh emergent wetlands. Little is known about dispersal habitat, as black rails are rarely found outside wetland habitat. They occupy perennial wetlands dominated by *Juncus effusus* and *J. balticus* and cattails (*Typha latifolia* and *T. domingensis*). These wetlands are in open grasslands, grazed pastures or oak savannas (Tecklin 1999). Wetlands greater than 0.4 hectares are more likely to support populations that persist over time, though California black rails have been found in wetlands as small as 0.2 acres (Tecklin 1999).

Model Results

Species Map 2. *California Black Rail Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for California black rail in the Plan Area. The majority of modeled potential habitat is where large blocks of fresh emergent marsh wetlands are mapped in the northeastern portion of the Plan Area. Because of the mapping methodology, this does not include all areas of actual fresh emergent marsh that may serve as habitat.

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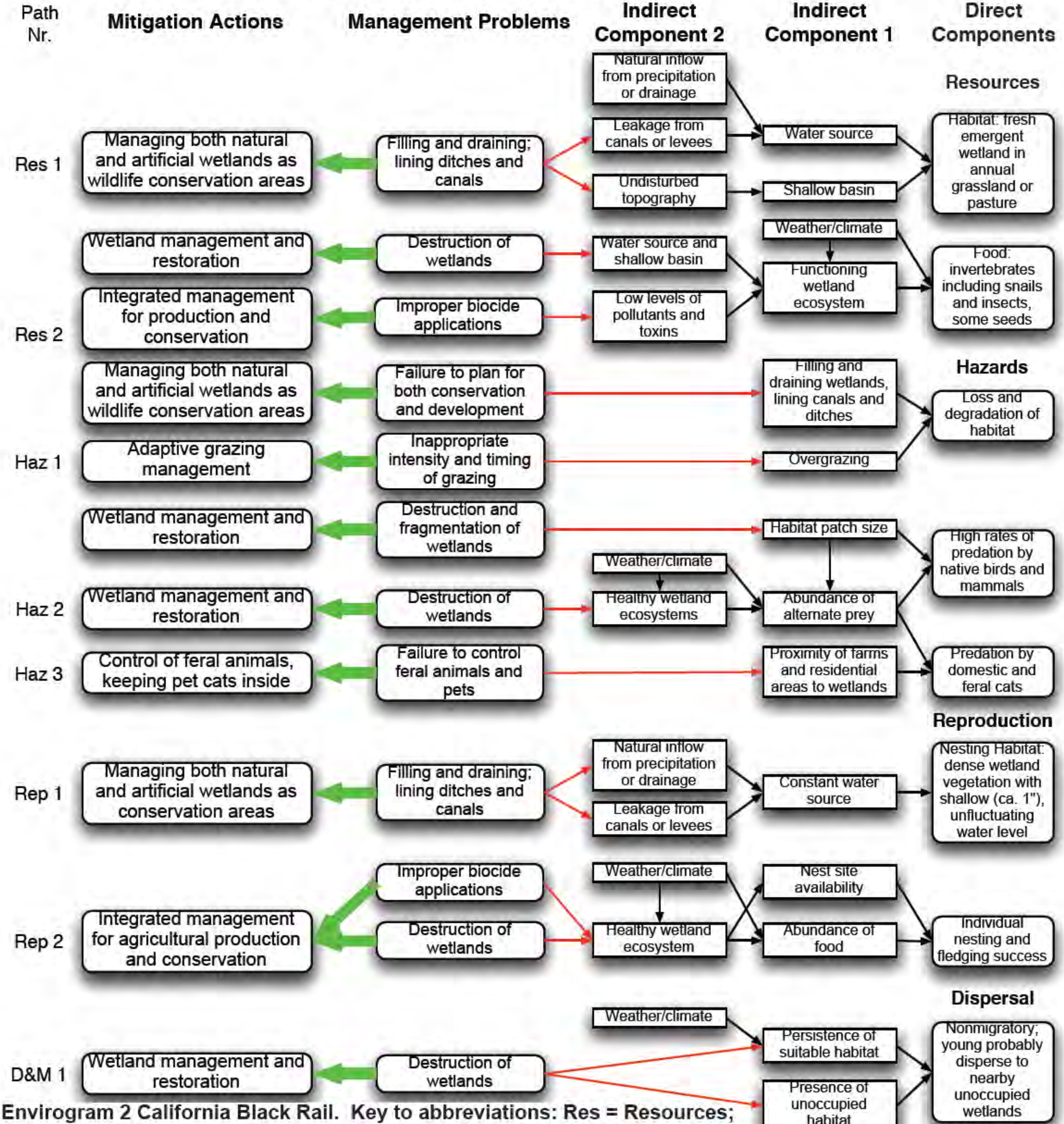
California Black Rail, *Laterallus jamaicensis coturniculus*



Envirogram 2 California Black Rail. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; D & M = Dispersal and Migration.

Envirogram Narrative

California Black Rail (*Laterallus jamaicensis coturniculus*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary. For example, resources are subdivided into food and habitat.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: California black rail requires fresh emergent wetlands in annual grassland or pasture with connectivity among wetlands. These wetlands depend on topography characterized by shallow basins and a water source from either natural inflow from precipitation or drainage or leakage from canals or levees. Filling and draining natural wetlands and lining ditches and canals results in the loss of this habitat, so all natural and many artificial wetlands should be managed as conservation areas.

Res2: California black rail feeds on invertebrates, including snails and a variety of insects. Some seeds are also eaten. The abundance of these prey items depend on a healthy wetland ecosystem, which in turn depends on a dependable water supply and low levels of toxins and other pollutants. Proper management or restoration of wetlands and ensuring that biocides and other agricultural chemicals do not end up in them are the keys to ecosystem health.

Hazards

Haz1: As with most species, the biggest threat to California black rail in Placer County is habitat loss—in this case the loss and degradation of wetlands from draining and filling, lining canals and ditches, and overgrazing. Natural and artificial wetlands should be managed as conservation areas, and grazing within these wetlands should be carefully monitored with regard to timing and intensity to ensure that livestock are not a source of degradation.

Haz2: California black rail is hunted by a variety of other birds such as egrets, northern harriers, gulls, and owls. Various mammals also prey on eggs and nestlings. Predation is related to patch size (small patches maximize edges and predator access) and a shortage of alternate prey. These are inter-related and affected by weather as well. Wetland restoration and management are the keys to reducing predation pressure.

Haz3: Predation by feral and domestic cats may also be a problem for California black rail, although less so in large wetlands. These predators are generally most abundant near homes and farms, particularly when alternate prey is not available. Controlling feral cats and keeping pet cats inside reduces this hazard.

Reproduction

Rep1: The California black rail nests in dense wetland vegetation where the water is consistently about one inch deep. These conditions require a reliable water source such as inflow from natural drainage or leakage from canals or levees. Filling and draining natural wetlands and lining ditches and canals results in the loss of nesting sites, further emphasizing that all natural and many artificial wetlands should be managed as conservation areas.

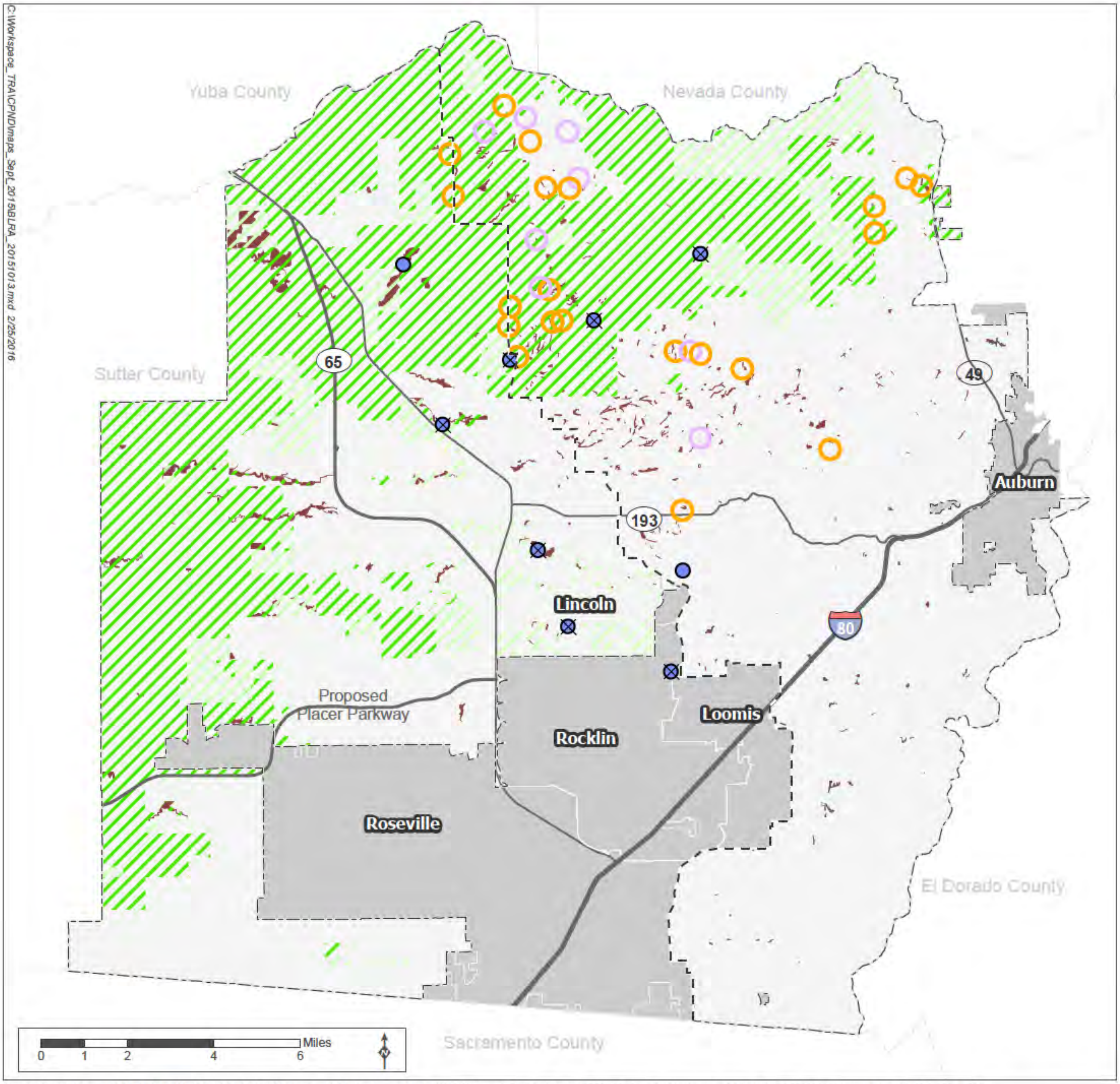
Rep2: Individual nesting and fledging success in California black rail depends on nest site availability and abundance of food. The former is related to the presence of wetlands, and the latter depends on productive foraging habitats associated with healthy wetland ecosystems, favorable weather, and proper application of biocides and other agricultural chemicals in nearby farms and ranches. These factors in turn are related to integrating wildlife needs and agricultural production.

Dispersal and Migration

D&M1: California black rail tends to be non-migratory and remain in Placer County throughout the year, provided that suitable habitat persists. Young birds may disperse and colonize new sites if densities in occupied wetlands become too high; this requires other suitable, unoccupied habitat to be available. This further emphasizes that all natural and many artificial wetlands should be managed as conservation areas and that connectivity among these patches should be maintained.

Summary

California black rail is rare in Placer County and depends on large patches of fresh emergent wetlands for foraging and breeding. This vegetation type has been destroyed or degraded in much of the western part of the County. Restoration and proper management of natural wetlands and wetlands resulting from leaky irrigation structures can have a positive effect on this species, provided that the wetlands are large enough and that connectivity among them is maintained. Large wetlands reduce the impacts of human disturbance and feral and native predators.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2015; Dudek, 2014; Tecklin and Hall, 2011; Widdowson, 2006; Sterling, 2003; Restoration Resources, 2011; eBird, 2015; Placer County Big Year Detection, 2009;

Occurrences
- Precise Location
- General Location

- Highly Suitable Habitat
- Moderately Suitable Habitat

Modeled Habitat
- Year-round Habitat
- Non-habitat

- Existing Protected Area
- Reserve Aquisition Area
- Non-participating City
- Major Road
- Valley/Foothill Divide
- Area A Boundary

Suitability of potential black rail habitat from Dudek (2014), based on reconnaissance roadside surveys of potentially suitable wetlands. This was not intended to be an exhaustive survey of potentially suitable wetlands in the Plan Area.
Reconnaissance surveys were focused in an area in the central and north central region of the Plan Area where California black rail have been previously detected. Further surveys are needed to assess occupancy.

MIG TRA

Species Map 2.
California Black Rail Modeled Habitat Distribution and Occurrence
Placer County Conservation Program – Western Placer County HCP/NCCP

Western Burrowing Owl

(*Athene cunicularia*)

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Status

Federal: Bird of Conservation of Concern; Federal Migratory Bird Treaty Act

State: Petitioned for listing under the California Endangered Species Act, but it was determined listing was not warranted (California Department of Fish and Wildlife [CDFW] 2003); Species of Special Concern

Critical Habitat: Not Applicable (N/A)

Recovery Plan: N/A

Distribution

North America

Burrowing owl lives and breeds in the desert and grassland habitats from south central Canada through most of the western and central United States and Central America to the southern end of South America (Rosenberg et al. 1998). It also occurs in Florida and the Caribbean (Poulin et al. 2011).

California

In California, the range of western burrowing owl extends through the lowlands south and west from north central California to Mexico, with small, scattered populations occurring in the Great Basin and the desert regions of the southwestern part of the state (DeSante et al. 1996). Western burrowing owl is absent from the coast north of Sonoma County and from high mountain areas such as the Sierra Nevada and the ranges extending east from Santa Barbara to San Bernardino. Western burrowing owl populations have been greatly reduced or extirpated from the San Francisco Bay area (Trulio 1997; Wilkerson and Siegel 2010) along the coast to Los Angeles. They have also apparently disappeared from the Coachella Valley. The remaining major population densities of western burrowing owl in California are in the Central and Imperial valleys (DeSante et al. 1996; Wilkerson and Siegel 2010).

Placer County Plan Area

Historical

Data for burrowing owls in the Sierra foothill and valley portions of Placer County is sparse, despite the presence of large areas of annual grasslands in this location (CDFG 2003[1]). There is no information on historic population size and distribution of western burrowing owl that is specific to the Plan Area. Grinnell and Miller (1944) indicated that the species was "originally common" and even "abundant" in suitable habitat that includes the Plan Area.

Current

Western burrowing owl is considered rare in Placer County (Webb 2009). Breeding western burrowing owls have been documented at Swainson's Preserve in 2012, 2013, and 2015 (Wages pers. comm.). A pair with at least two nestlings was observed in 2012, a pair with at least four nestlings was observed in 2013, and two pairs with three nestlings each in 2015 (Wages pers. comm.).

Records of western burrowing owl in Placer County in the California Natural Diversity Database (September 2015) are likely of over-wintering birds. These records include: 1) one burrowing owl observed along Highway 65 south of Sheridan in 2011; 2) two burrowing owls observed along Nader Road off Highway 65 in 2011; 3) one burrowing owl observed in Redwing Preserve in 2005; 4) one adult observed at a burrow site on January 30, February 1, and February 18, 2008 on the Moore Ranch Wetland Restoration Project property 250 feet north of east Catlett Road and 0.4 mile west of Fiddyment Road southwest of Lincoln; 5) two individuals in moderately grazed, rolling grassland on the north side of Philip Road, approximately 0.75 mile west of Fiddyment Road, northwest of Roseville. These individuals were observed year-round in 1998, but none were observed on May 5, 2003; 6) two adults were observed April 29, 2008 at a burrow site in open grassland on the Swainson's Preserve, 0.43 mile south, southeast of the intersection of West Wise Road and Highway 65; and 7) at least one individual was observed at the Sterling Silver Stables in the southwest corner of the Plan Area in 2007. In addition to the California Natural Diversity Database records, a single western burrowing owl has been observed during the annual Lincoln Christmas Bird Count every year since the 2002 count, except in 2006 and 2008 when none were observed. Western burrowing owl was also observed at three locations in the Plan Area during the 2003 watershed surveys for the Placer Legacy program (Pandolfino pers. comm., Easterla pers. comm.).

Population Status & Trends

North America

Burrowing owl was once widespread and generally common over western North America. In recent decades a number of populations have declined or, in some cases, disappeared altogether (Poulin et al. 2011). The burrowing owl range has generally contracted southward and westward in North America (Wellicome and Holroyd 2001 as cited in Poulin et al. 2011). Burrowing owl breeding range has retreated from 1967 to 2008 in southern and northern California, Washington, southern Canada,

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[1] As of January 1, 2013, the California Department of Fish and Game (CDFG) was renamed the California Department of Fish and Wildlife. When this document cites reports prepared by the Department prior to 2013, the reference includes the prior department name of CDFG. Both CDFW and CDFG refer to the same agency.

eastern North and South Dakota, eastern Nebraska, eastern Kansas, and southern Texas (Macías-Duarte and Conway 2015). Burrowing owl breeding range is thought to have expanded toward unoccupied areas in southern Montana, eastern Oregon, central Nevada, and the four corners region of the United States (Macías-Duarte and Conway 2015). Burrowing owl is now endangered in Canada, a species with special protection in Mexico, and has declined in many parts of the United States (DeSante et al 1996, 1997; James and Espie 1997; Sauer et al. 2005; Poulin et al. 2011). In California, the species is a species of special concern; it is listed as endangered or threatened in a number of other states.

California

The California Department of Fish and Wildlife indicates that the California population of western burrowing owl is between 1,000 and 10,000 pairs (James and Espie 1997; Rosenberg et al. 1998) with a declining trend (Gervais et al. 2008; Wilkerson and Siegel 2010). Wilkerson and Siegel (2010) estimated the number of burrowing owl pairs statewide from 2006-2007 at 9,298 pairs. The population of burrowing owls was found to be highly concentrated in the Imperial Valley (68.9% of the statewide population) and to a lesser extent, the Southern Central Valley (12% of the statewide population) (Wilkerson and Siegel 2010). Christmas Bird Count data from 1959 – 1988 show declines in midwinter numbers of western burrowing owl in California (Sauer et al. 1996). In contrast, the numbers of western burrowing owl on Breeding Bird Survey Routes in California increased significantly from 1968 to 2004 (Sauer et al. 2005). Wilkerson and Siegel (2010) observed that the major patterns of burrowing owl occurrence across California appeared to be relatively unchanged since 1993, although non-significant declines were observed in numerous regions. The two urban areas in California with the sharpest declines were the San Francisco Bay Area and Bakersfield (Wilkerson and Siegel 2010). The primary factors cited in the decline are habitat loss, pesticides, predators, harassment, reduced burrow availability, and vehicle collisions.

Placer County Plan Area

There is no detailed information on population trends of western burrowing owl in Placer County because of the lack of baseline data; however, Webb (2009) describes the population as declining.

Natural History

The habitat requirements, ecological relationships, life history, and threats to western burrowing owl described below are summarized in diagram form in the Envirogram 1 Western Burrowing Owl.

Habitat Requirements

Western burrowing owl is found in open, dry grasslands, agricultural and range lands, and desert habitats often associated with burrowing animals and short vegetation (Poulin et al. 2011). It can also inhabit grass, forb, and shrub stages of piñon and ponderosa pine habitats. In addition to "natural" breeding habitats, areas such as agricultural fields, golf courses, cemeteries, road allowances, airports, vacant urban lots, and fairgrounds are regularly used (Poulin et al. 2011). Western burrowing owl requires burrows for roosting and nesting (CDFW 2012). In California, nest and roost burrows are most commonly dug by ground squirrels (e.g., *Otospermophilus beecheyi*), but the owl may also use the dens or holes of other species such as badger (*Taxidea taxus*) and coyote (*Canus latrans*) (Ronan 2002). In some instances, burrowing owls have been known to excavate their own burrows (Barclay 2007 as cited in CDFW 2012) or use natural rock cavities, debris piles, culverts, and pipes (Rosenberg et al. 1998).

Western burrowing owls can be found at elevations ranging from 200 feet below sea level to 9,000 feet above sea level. Foraging habitat is essential to burrowing owls (CDFW 2012). Foraging occurs primarily within 600 meters of their nests during the breeding season (CDFW 2012). Western burrowing owl commonly perches on fence posts or on mounds outside the burrow. It is active day and night, but is usually less active in the peak of the day (Rosenberg et al. 1998).

Reproduction

Western burrowing owls can begin breeding at 10 months of age (Poulin et al. 2011). Although typically monogamous, polygyny has been observed in western burrowing owl populations (Barclay and Menzel 2011). The breeding season for western burrowing owl in California is February to late August (Haug et al. 1993; Thompsen 1971; CDFW 2012), with some variances by geographic location and climatic conditions (CDFW 2012). The season tends to last later in the northern part of the range. Clutch size ranges from 1-12 eggs and averages about 7 eggs (Poulin et al. 2011). The incubation period is 28–30 days (Poulin et al. 2011). The female performs all of the incubation and brooding and is believed to remain continually in the burrow while the male does all the hunting (Poulin et al. 2011). Burrowing owls may use satellite or non-nesting burrows during the breeding season, moving young at 10-14 days, presumably to reduce the risk of predation (Desmond and Savidge 1998 as cited in CDFW 2012). Several studies have documented the number of satellite burrows used by young and adult burrowing owls during the breeding season as between one and 11 burrows with an average use of approximately five burrows (Haug 1985). The young fledge from 44 to 53 days but remain near the burrow and join the adults in foraging flights at dusk (Rosenberg et al. 1998).

Dispersal Patterns

Western burrowing owl tends to be resident where food sources are stable and available year-round. It disperses or migrates south in areas where food becomes seasonally scarce. In resident populations, nest-site fidelity is common, with many adults renesting each year in their previous year's burrow; young from the previous year often establish nest sites near (<1000 feet) their natal sites (Rosenberg et al. 1998). Western burrowing owls in migratory populations also often renest in the same burrow, particularly if the previous year's breeding was successful (Belthoff and King 1997). Other birds in the same population may move to burrows near their previous year's burrow. Differences in site fidelity rates may reflect differences in nest predation rates (Catlin et al. 2005). Despite the high nest fidelity rates, dispersal distances may be considerable for both juveniles (i.e., natal dispersal) and adults (i.e., postbreeding dispersal), but this also varies by location (Catlin 2004; Rosier et al. 2006). Distances of 32 miles to 93 miles have been observed in California for adult and natal dispersal, respectively (CDFW 2012). Holroyd et al. (2011) observed a burrowing owl disperse approximately 1,156 miles between two nesting attempts within the same breeding season. This is the longest distance ever recorded for breeding dispersal for any raptor within the same breeding season (Holroyd et al. 2011).

Longevity

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

Sources of Mortality

Predators of western burrowing owl includes prairie falcon (*Falco mexicanus*), barn owl (*Tyto alba*), red-tailed hawk (*Buteo jamaicensis*), Swainson's hawk (*Buteo swainsoni*), ferruginous hawk (*Buteo regalis*), northern harrier (*Circus cyaneus*), golden eagle (*Aquila chrysaetos*) (Poulin et al. 2011). In addition, mammals such as badgers, foxes (*Vulpes vulpes*), skunks (*Mephitis* spp.), weasels (*Mustela* spp.), opossums (*Didelphis virginiana*), coyote, and domestic dogs (*Canis lupus familiaris*) and cats (*Felis catus*) are major predators of burrowing owls (Poulin et al. 2011). Many owls are killed at night by traffic when flying low over roads (CDFW 2003; Klute et al. 2003), as well as by wind turbines (Smallwood et al. 2007) and barbed wire fences (Todd et al. 2003). Attempts to exterminate rodents by the use of poisons may also kill western burrowing owls (Rosenberg et al. 1998).

Behavior

Western burrowing owl tends to be an opportunistic feeder (Poulin et al. 2011). Large arthropods, mainly beetles and grasshoppers, comprise a substantial portion of its diet. Small mammals, especially mice, rats, gophers, and ground squirrels, are also important food items. Other prey animals include reptiles and amphibians, scorpions, young cottontail rabbits, bats, and birds such as sparrows and horned larks (*Eremophila alpestris*) (Poulin et al. 2011). Consumption of insects increases during the breeding season. Western burrowing owl hovers while hunting; after catching the prey it returns to perches on fence posts or the ground. Western burrowing owl is primarily active at dusk and dawn, but if necessary will hunt at any time of day (Rosenberg et al. 1998).

Movement and Migratory Patterns

Migration routes of burrowing owls have been poorly documented (Haug et al. 1993). Northern populations of western burrowing owl are usually migratory, while more southern populations (e.g., Florida and southern California) may move short distances (Coulombe 1971; Martin 1973; Botelho 1996; Poulin et al. 2011) or not at all (Brenckle 1936; Ligon 1961; Thomsen 1971; Haug et al. 1993). Burrowing owl primarily winters in California, Arizona, New Mexico, Texas, Louisiana, Florida, and Mexico (Sheffield 1997). Those western burrowing owls breeding farthest north appear to migrate the farthest south (James and Ethier 1989).

Ecological Relationships

Western burrowing owl in California is commensal with California ground squirrel in rangeland and agricultural areas. It may compete incidentally with other predators such as coyote, other owls and hawks, skunks, weasels, and badgers for rodents and a variety of insects (Rosenberg et al. 1998).

Threats

Conversion of grasslands to agriculture, other habitat destruction, and poisoning of ground squirrels have contributed to population reductions first noted in the 1940s. Habitat loss and degradation from rapid urbanization of farmland in the core areas of the Central and Imperial Valleys of California is the greatest threat to burrowing owls in California (Garvais et al. 2008).

The burrowing owl depends on colonies of burrowing mammals for nest sites. The reduction of such colonies by agriculture and control programs has limited access to nest burrows and contributed to the loss of burrowing owls (Klute et al. 2003; Poulin et al. 2011). In California, ground squirrel burrows are most often used by burrowing owls for nesting and cover; thus, ground squirrel control programs may affect owl numbers in local areas by eliminating a necessary resource (CDFW 2012).

Burrowing owls suffer direct losses from a number of sources (CDFW 2012). Vehicle collisions are a significant source of mortality especially in the urban interface and where owls nest alongside roads (Haug et al. 1993; CDFW 2003; Gervais et al. 2008). Road and ditch maintenance and disking to control weeds in fallow fields, among other activities, may destroy burrows (Catlin and Rosenberg 2006) which may trap or crush owls. Wind turbines at Altamont Pass Wind Resource Area are known to cause direct burrowing owl mortality (Thelander et al. 2003).

Context for a Regional Conservation Strategy

Western burrowing owl is present, but rare, in western Placer County. Populations in the Plan Area are on the eastern edge of the species' central range in California. Breeding western burrowing owls have been documented at Swainson's Preserve in 2012, 2013, and 2015 (Wages pers. comm.). A pair with at least two nestlings were observed in 2012 and a pair with at least four nestlings were observed in 2013 and two pairs with three nestlings each in 2015 (Wages pers. comm.). Records of western burrowing owl in Placer County in the California Natural Diversity Database (September 2015 are likely of over-wintering birds. These records are scattered throughout western Placer County.

In the region, western burrowing owl is known primarily from the south and southeast of western Placer County, although there are scattered records of occurrence in Yuba, Butte and Colusa counties. Although limited in occurrence in the Plan Area, protection of individual occurrences is not critical. Western burrowing owl has been successfully relocated ("passive relocation") and has also been found to utilize man-made burrows (Trulio 1995). The preservation of habitat in general however, is stressed. Loss of habitat and poisoning of ground squirrels are the top causes of decline of the species statewide. Lands prioritized for preservation/acquisition include annual grasslands and rangelands. In addition, agricultural lands often provide suitable habitat and their protection may benefit the species. The presence of ground squirrel burrows on these lands is critical in providing breeding habitat, although the species may forage in grasslands, rangelands, and agricultural lands devoid of burrows.

Modeled Species Distribution in the Plan Area

Model Assumptions

<u>Year-round habitat (Nesting and Overwintering)</u>

Modeled overwintering and nesting habitat for western burrowing owl includes these habitats within the western portion of the Plan Area below 200 feet in elevation: valley oak woodland, oak woodland savanna, vernal pool complex, annual grassland, alfalfa, pasture, and cropland.

Rationale

Western burrowing owls use open, dry grasslands and agricultural and range lands that have burrowing animals and short vegetation. Western burrowing owls forage in open grasslands, pasturelands, agricultural fields and field edges, and along the edges of roads and levees where vegetation is low. They require burrows for roosting and nesting and nest in open habitats with sparse vegetative cover and a high density of burrows. The land-cover types that characterize nesting and foraging habitat for western burrowing owl capture the general habitat requirements of western burrowing owl. This model, however, overestimates the extent of western burrowing owl habitat because the specific characteristics of western burrowing owl habitat are likely patchily distributed within the Plan Area.

Model Results

Species Map 3. *Western Burrowing Owl Modeled Habitat Distribution and Occurrence* shows the modeled potential year-round habitat for western burrowing owl overwintering and potential nesting within the Plan Area. Potential overwintering and nesting habitat occurs throughout the Plan Area, though it is primarily concentrated in the western Valley portion of the Plan Area below 200 feet in elevation. The known occurrences of this species fall within the modeled habitat.

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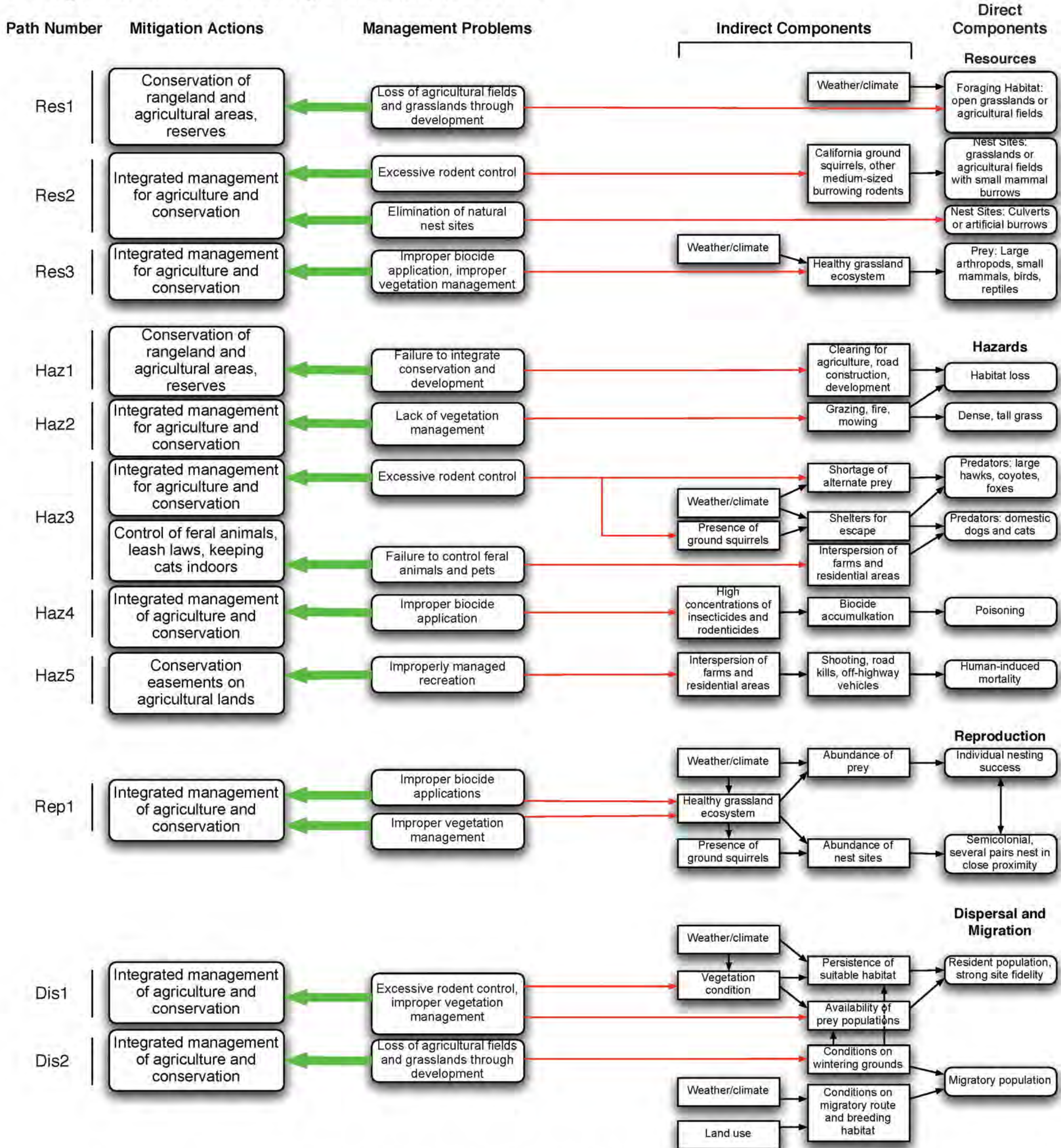
Envirogram 3 Western Burrowing Owl, *Athene cunicularia*



Envirogram 3 Western Burrowing Owl. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram Narrative

Western Burrowing Owl (*Athene cunicularia*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary. For example, resources are subdivided into foraging habitat, nest sites, and prey.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Western burrowing owls rely upon grasslands or agricultural fields for foraging habitat, and much of this has been lost to various kinds of development. Conservation of agricultural land and the creation of reserves can mitigate this loss to some degree.

Res2: The owls use burrows created by rodents, especially California ground squirrels (*Spermophilus beecheyi*), for nesting and roosting. Thus, the presence of medium-sized rodents is necessary for western burrowing owls. Excessive rodent control can eliminate these species, so land management that integrates agricultural production and conservation is necessary. Artificial nest sites also are used by the owls and can be used as a management technique to increase population sizes.

Res3: Prey of western burrowing owls includes large arthropods, small mammals, reptiles, amphibians, and small birds. During years with favorable weather, prey species are abundant in healthy grassland ecosystems and agricultural fields that do not have heavy biocide applications. Land management that integrates agricultural production and conservation helps provide these conditions.

Hazards

Haz1: Loss of natural grasslands to agriculture and other types of development has resulted in substantial loss of habitat for western burrowing owls. Conservation easements on agricultural land can mitigate these losses to some extent.

Haz2: In the absence of burning, mowing, or grazing, grass growth makes habitat unsuitable for the western burrowing owls, so vegetation management, such as properly managed livestock grazing, is necessary.

Haz3: Natural predators of western burrowing owls include larger raptors, foxes, and coyotes; additional predation pressure comes from feral and domestic dogs and cats. A healthy ground squirrel population provides refuges and lessens predation pressure on the owls. Predation pressure also is reduced by an abundance of alternative prey items, largely determined by weather patterns and the extent of rodent control. Land management that integrates agricultural production and conservation helps provide these conditions as does controlling feral cats and dogs and confining pets to yards or houses.

Haz4: Western burrowing owls can accumulate and be poisoned by various biocides. This usually occurs when these toxins are applied in excess. Integrated management for conservation and agricultural production should minimize these circumstances.

Haz5: Protecting western burrowing owls and their habitat from human disturbance such as OHV use and lessening direct mortality from casual shooting and road kills is important. The breeding period is an especially sensitive time because human disturbance can increase nest predation and nest abandonment and result in prolonged exposure of eggs to the elements, nestling starvation, early fledging, and predation upon fledglings.

These problems occur most frequently in the proximity of residential areas and in the absence of properly controlled recreational use of land. Conservation easements on agricultural land may be effective in minimizing these kinds of disturbances to the owls.

Reproduction

Rep1: Western burrowing owls are semi-colonial, and several pairs nest in close proximity. The nesting success of individual pairs depends upon prey abundance, which in turn depends on weather patterns and habitat condition. Excessive biocide application limits prey availability, and tall, rank vegetation inhibits the ability of the owls to hunt successfully. The presence of California ground squirrels or other medium-sized rodents determines the number of nest sites available to a colony. Again, appropriate conditions for the owls depend on the integration of production agriculture and conservation.

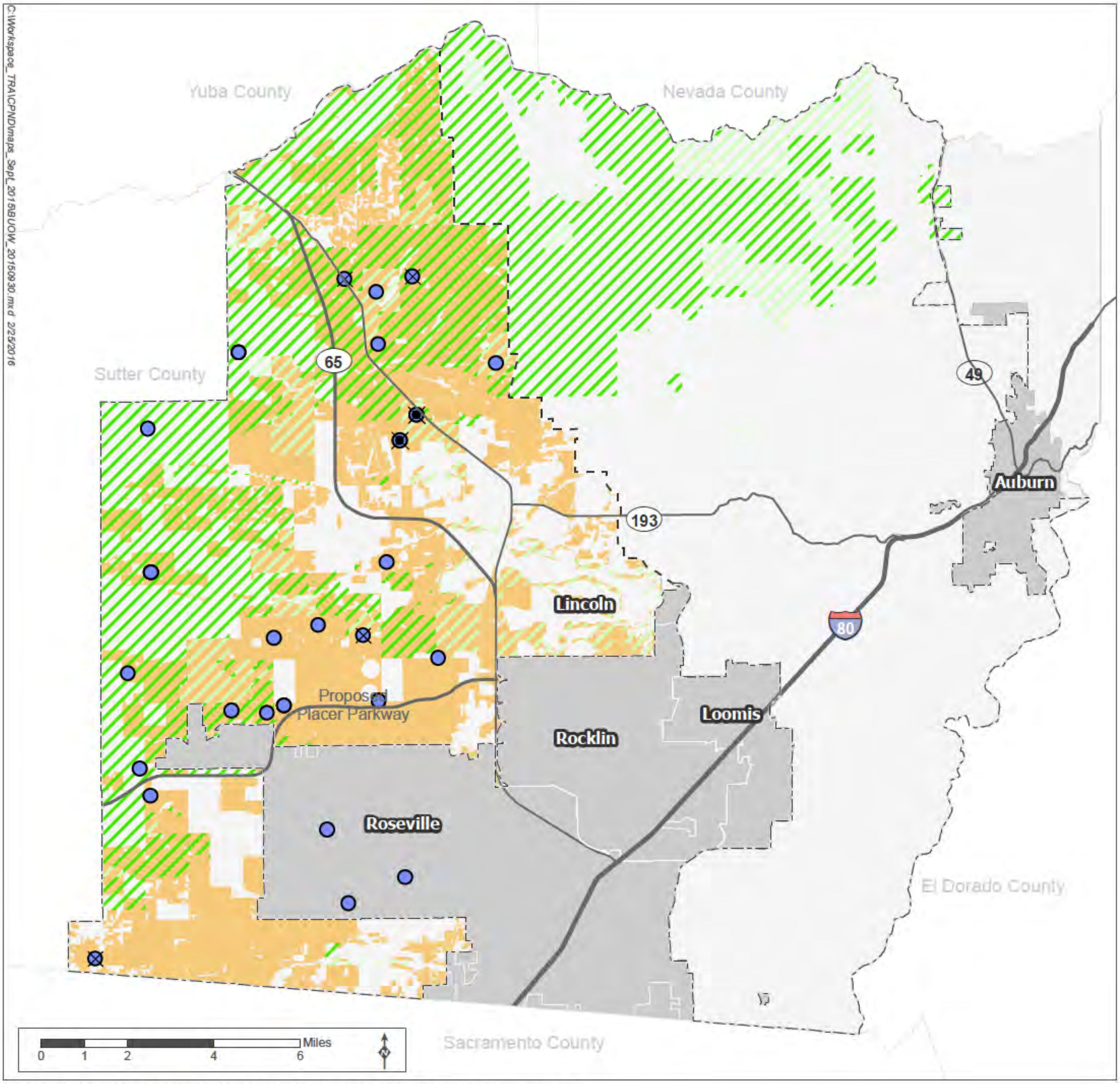
Dispersal and Migration

Dis1: A potential resident western burrowing owl population in Placer County is supplemented by additional individuals during winter. Resident populations remain in their breeding localities year around: the adults are highly philopatric, and young owls usually remain to breed close to their natal sites as long as adequate prey resources are available and habitat remains suitable. Weather conditions and integrated management both play major roles in maintaining resident owl populations. Management techniques that encourage stable prey availability may encourage the establishment of resident populations, and increased adult survival during winter will increase recruitment and population sizes.

Dis2: Migrant populations are those that have bred elsewhere but moved to areas of stable food supply in winter. Migration is a hazardous time, and the birds' physical state and energy reserves and the conditions along their migratory routes and in their breeding habitats are beyond the control of Placer County. However, maintaining the conditions that encourage resident populations also will be beneficial for migrant populations.

Summary

As predators near the top of their food web, western burrowing owls are highly sensitive to ecosystem conditions. They are also quite sensitive to various kinds of disturbance caused by human activities and feral and domestic animals, particularly during the breeding season. These factors suggest that the best strategy for western burrowing owl conservation in Placer County is to conserve range and crop land and manage it in an integrated fashion for both production and biodiversity conservation. The owls' requirements are such that they are very compatible with vernal pool-grassland ecosystem conservation as well.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2014; Placer Land Trust, 2010; eBird, 2012; Patrick Moeszinger, 2014

Occurrences
- Precise
- General
- Nest Location

Modeled Habitat
- Potential Nesting and Overwintering Habitat
- Non-habitat

- Existing Protected Area
- Reserve Aquisition Area
- Non-participating City
- Major Road
- Valley/Foothill Divide
- Area A Boundary

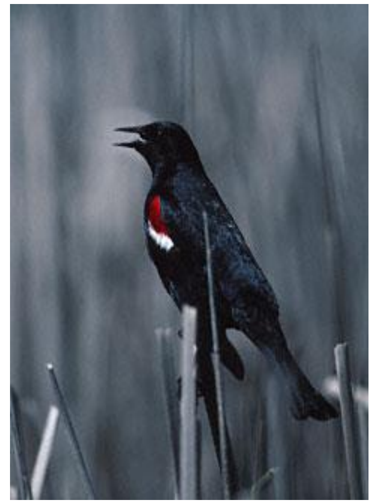
Species Map 3.

Western Burrowing Owl Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

MIG TRA

Tricolored Blackbird
(*Agelaius tricolor*)



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Status

Federal: Bird Species of Conservation Concern; Under Review for Federal Endangered Species Act Listing; Migratory Bird Treaty Act

State: Species of Special Concern; Threatened, California Endangered Species Act Listing (2017)

Critical Habitat: Not Applicable (N/A)

Recovery Plan: Conservation Plan for the Tricolored Blackbird (*Agelaius tricolor*) by The Tricolored Blackbird Working Group, dated January 2009

Distribution

North America

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

California

The historic breeding range for tricolored blackbirds included the Sacramento and San Joaquin Valleys, the foothills of the Sierra Nevada south to Kern County, the coastal slope from Sonoma County south to the Mexican border, and, sporadically the Modoc Plateau (Neff 1937). Historical surveys; however, did not include large areas of the species' currently known breeding range. Therefore, no the full extent of the tricolored blackbirds historical breeding range is not known.

The overall range of the tricolored blackbird has changed little since the mid-1930's (Beedy and Hamilton 1999), although more recent surveys have documented occurrences of tricolored blackbirds in areas that previously lacked surveys. Tricolored blackbirds are found at low elevation sites the entire length of the state (Dudek 2014). The largest number of birds has for decades been in the Central Valley (Neff 1937, Beedy and Hamilton 1999, Kyle and Kelsey 2011), but the Central Valley has recently seen a dramatic drop in abundance, down about 78% from 2011 to 2014 (Meese 2014). Small numbers of tricolored blackbirds are also found in coastal locations from Santa Barbara County north to Mendocino County, isolated sites in the western interior of southern California, and on the Modoc Plateau in northeastern California (Beedy 2008). Tricolored blackbirds are fairly common but localized breeders in the western Sierra Nevada foothills up to about 1,000 feet (Beedy and Pandolfino 2013), and uncommon breeders up to about 1,500 feet in Calaveras and Stanislaus Counties (Airola pers. comm. 2014).

The species breeds in large colonies, with breeding sites concentrated in the San Joaquin Valley, Sacramento–San Joaquin Delta, and the southern Sacramento Valley. The species also breeds along the California coast from Humboldt to San Diego counties; on the Modoc Plateau and western edge of the Great Basin (mostly Klamath Basin); in lowlands surrounding the Central Valley; and in western portions of San Bernardino, Riverside, and San Diego counties.

Following the breeding season tricolored blackbirds flock with other blackbird species and are concentrated in the Sacramento Valley. During winter, virtually the entire population of the species withdraws from Washington, Oregon (although a few remain), Nevada, and Baja California and wintering populations shift extensively within their breeding range in California (Beedy and Hamilton 1999). Major wintering concentrations occur in and around the Sacramento-San Joaquin River Delta and coastal areas, including Monterey and Marin counties, where they are often associated with dairies (Beedy 2008). Small flocks may also appear at scattered locations from Sonoma County south to San Diego County, and sporadically north to Del Norte County (Beedy and Hamilton 1999, Unitt 2004). Tricolored blackbirds are rare in the winter in the southern San Joaquin Valley and in the Sacramento Valley north of Sacramento County (Beedy and Hamilton 1999). The Southern California population segment south of the Tehachapi Mountains appears to be mostly confined to Southern California, although rarely, some birds will move out of the Central Valley into Southern California (Meese unpubl. data).

Placer County Plan Area

Historical

Neff (1937) found only two tricolored blackbird breeding colonies in Placer County. Both of these were in cattails (*Typha* sp.) along canals near the city of Lincoln in 1933 and 1936 and contained about 1,000 and 1,500 nests, respectively. A colony (unknown number of pairs) was found nesting in cattails in a wet pasture near Lincoln in 1971 (Hosea 1986; California Natural Diversity Database [CNDDB] 2015). Another colony of about 2,000 pairs was found in cattails around a marsh on the Chamberlain Ranch (north-northwest of Lincoln) in 1971 (DeHaven pers. comm.).

Current

Within the Plan Area, tricolored blackbirds occur in the lower elevations from 100 to 300 feet (Jones and Stokes 2004). It is often found in mixed flocks with other species of blackbirds, or may occur as single species flocks in annual grasslands, wetlands, and agricultural areas. Suitable foraging habitat for this species exists in most watersheds in the Plan Area that have not been extensively urbanized.

A statewide survey for tricolored blackbird was conducted in 2008 and again in 2014 (Meese 2014). In addition to the data collected in the statewide survey, data on tricolored blackbird distribution and occupancy is available on the Tricolored Blackbird Portal (http://tricolor.ice.ucdavis.edu/). Data found on the Tricolored Blackbird Portal includes both published and unpublished data from a variety of sources, including data collected by nonprofessional scientists. This account summarizes the published data and numbers from the 2014 statewide survey, as well as distribution and breeding data found on the Tricolored Blackbird Portal that provides valuable information on habitat usage in Placer County.

Meese (2014) groups Placer County into the Sierra Foothills region or the tricolored blackbird range, and data collected in the 2014 statewide survey suggests that this region may continue to support successful breeding by tricolored blackbirds. A total of 25,717 tricolored blackbird individuals were observed in the Sierra Foothills region in 2014 (Meese 2014). The dominant land use in areas occupied by tricolored blackbirds is ranching. Tricolored blackbirds in Placer County breed primarily in isolated stands of Himalayan blackberry and small cattail and bulrush marshes in stock ponds (Airola et. al. 2015).

A total of 17,600 birds were estimated from four active colonies in Placer County during the April 2014 statewide survey (Meese 2014). Tricolored blackbird colonies were detected at Gleason Ranch on Sunset Boulevard West (6,500 individuals), West Ferrari Ranch Road (1,800 individuals), Little Ben (7,500 individuals) and Orchard Creek (1,800 individuals, also known as Industrial Avenue #2). In 2014, Placer County supported a considerable portion of the tricolored blackbird population within the Sacramento Valley (Meese 2014). Specifically, 33% of the tricolored blackbird individuals counted in nine Sacramento Valley counties (Amador, Butte, Colusa, El Dorado, Placer, Sacramento, Sutter, Yolo, Yuba) occurred in Placer County. Although data from the 2014 statewide survey provides a good one-time snapshot of tricolored blackbird distribution and numbers in Placer County and statewide, additional surveys conducted in Placer County in 2014 are reported in the Tricolored Blackbird Portal.

The Tricolored Blackbird Portal documents 21 colony sites and aggregations in western Placer County, of which 15 are active (i.e., colonies were documented at a set within prior 10 years) and may potentially have breeding colonies, and one is listed as historical (Table 1). The Tricolored Blackbird Portal reports six colonies as being active in 2015, with approximately 12,715 tricolored blackbird individuals detected. These colonies are located at Gleason Ranch on Sunset Boulevard West, Little Ben, Markham Ravine #3, Orchard Creek, West Ferrari Ranch Road, and Dalby East of Highway 65 Lincoln Bypass (Table 1). A total of seven colonies were found to be active in 2014. In addition to the four colonies observed as part of the statewide survey (i.e., Gleason Ranch, West Ferrari Ranch Road, Little Ben, and Orchard Creek), individuals were also observed at Bear Valley Meadow, Yankee Slough, and Dalby East of Highway 65 Lincoln Bypass (Table 1).

A large, and regionally important, winter roost of blackbirds, including tricolored blackbirds, exists at Yankee Slough, northwest of Lincoln in western Placer County. In January 2014, this roost contained approximately 35,000 blackbirds of several species, several thousand of which appeared to be tricolored blackbirds (Dudek 2014). Yankee Slough also supports hundreds of breeding tricolored blackbirds (Meese pers. obs. 2014).

Population Status &Trends

North America/California

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

DeHaven et al. (1975a) estimated that the overall population size in the Sacramento and northern San Joaquin valleys had declined by more than 50% since the mid-1930s. They performed intensive surveys and banding studies in the areas surveyed by Neff (1937) and observed significant declines in tricolored blackbird numbers and the extent of suitable habitat in the period since Neff's surveys. Orians (1961a) and Payne (1969) observed colonies of up to 100,000 nests in Colusa, Yolo, and Yuba counties, but did not attempt to survey the entire range of the species.

The U.S. Fish and Wildlife Service (USFWS), the California Department of Fish and Wildlife, and California Audubon cosponsored intensive, volunteer tricolored blackbird surveys in suitable habitats throughout California in 1994, 1997, 1999, 2000, 2001, 2005, 2008,nd 2014, and 2017 (Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000; Kelsey 2008; Meese 2014; Meese 2017). Statewide, tricolored blackbird populations have fluctuated since 1994, declining by 62% from 1994 to 2001, increasing from 2001 to 2008, and then decreasing again in 2014 before slightly increasing in 2017 (Table 1). The primary causes of the long-term and more recent declines has been attributed to loss of native breeding habitat and the concentration of large colonies in agriculture fields where large proportions of the colonies are subject to reproductive failure (Hamilton et al. 1999; Hamilton 2000; Meese 2014). The widespread and ongoing conversion of native habitats to dairies, orchards, vineyards, rice, and other forms of agriculture and the use of effective and persistent insecticides may have created unsuitable breeding conditions in much of the core area of the species range (Meese 2015). Graves (2013) describes the range-wide population decline has not occurred uniformly among habitats and regions; a relatively recent agricultural crop (triticale) has supported large breeding populations in the San Joaquin Valley and resulted in an increased proportion of birds being within this region compared to records prior to the 1980s. However, this habitat is ephemeral and carries with it a high risk of failure through harvesting (Graves 2013). In 1994, full season survey results indicated that 70% of all tricolored blackbird nests and 86% of all foraging by nesting birds occurred on private agricultural land (Hamilton et al. 1995). Approximately 54% of all observed tricolored blackbird nesting efforts were associated with agricultural crops, primarily grain crops grown for silage at dairies (Beedy and Hamilton 1997).

Table 1. Total tricolored blackbirds counted in California in statewide surveys from 1994 – 2014.

Year	Number
1994	369,359
1997	237,928
1999	104,786
2000	162,508
2001	146,126
2005	257,802
2008	394,858
2014	145,135
2017	177,656

In 2017, a total of 177,656 birds were counted 37 counties from 44 counties and 884 locations surveyed. Of this total, 172,499 birds were observed at breeding colonies and 5,157 were observed in nonbreeding aggregations or as single birds (Meese 2017). A total of 145,135 birds were counted in 37 counties during the 2014 statewide survey (Meese 2014). Tricolored blackbirds were observed at a total of 143 locations out of 802 locations surveyed. The rate of decline in the number of tricolored blackbirds appears to be increasing. From 2008 to 2011 the number of tricolored blackbirds dropped by 34%, from 394,858 to 259,322 birds (Kyle and Kelsey 2011), and from 2011 to 2014 the number of tricolored blackbirds dropped by 44%, from 259,322 to 145,135 birds. This is despite the fact that in 2014, 75 new location records were added by 27 different Tricolored Blackbird Portal users as result of the statewide survey. In 2008, 180 sites were visited, in 2011, 608 sites were visited, and in 2014, 802 sites were visited. Despite this substantial increase in sites that were visited, the total number of tricolored blackbirds counted declined dramatically. In addition, the 2014 census reported a substantial downward trend in the sizes of the largest colonies over the past decade (Meese 2014).

In 2014, tricolored blackbird numbers were down markedly from the two previous statewide surveys in the San Joaquin Valley, especially in Kern and Merced counties, where the breeding birds had recently been most concentrated. Overall, the number of breeding birds in the San Joaquin Valley dropped 78% in 6 years, from 2008 to 2014, and the number of birds seen in counties along the Central Coast was less than 10% of that seen in 2008 (Meese 2014). In 2014, the largest nesting colonies occurred in Tulare, Madera, and Merced counties, but these colonies all supported drastically fewer numbers of tricolored blackbirds than in the previous two census surveys (Meese 2014). Meanwhile, Placer and Sacramento counties saw a marked increase in the number of tricolored blackbirds (Meese 2014).

The 2014 statewide survey also identified several important distribution and population trends for tricolored blackbird.

- The rate of decline in the number of tricolored blackbirds appears to be accelerating. The rate of mortality of adults far exceeds that of the recruitment of new breeding birds into the population and chronically low reproductive success since 2007 appears to be a major factor causing the disparity between mortality and recruitment (Meese 2013).
- The number of tricolored blackbirds has decreased steeply statewide, with declines most pronounced in the San Joaquin Valley and along the Central Coast. Meanwhile, the number of tricolored blackbirds in the Sierra Nevada foothills and Sacramento County have increased, suggesting either that tricolored blackbirds are moving into the foothills from other regions or are breeding relatively more successfully in the Sierra Nevada foothills than they are in the San Joaquin Valley or Central Coast (Meese 2014).
- A dramatic decline in the size of the largest colonies is associated with the decline in the number of tricolored blackbirds (Meese 2015)
- A large proportion of birds have become increasingly concentrated into relatively few colonies. Specifically, in 1994, 2000, 2008, and 2014 the top 10 counties accounted for 60%, 59%, 77.5%, and 90% of the total statewide population estimate, respectively.

However, based on the 2017 state-wide surveys, the decline in the number of tricolors observed since the 2008 survey appears to have ceased. From 2008 to 2014 the number of tricolors dropped by 64%, from 395,000 to 145,000 birds (Kelsey 2008, Meese 2014) but the number of birds increased by 22% from 2014 to 2017.

While the results of the 2017 Tricolored Blackbird Statewide Survey suggest that the rapid decline in abundance observed since at least 2008 has been arrested and that there has been an increase in abundance since 2014 of about 32,000 birds. Looking closely at these results shows that the majority of the increase from 2014 to 2017 is due to birds observed in the San Joaquin Valley, where the number of birds estimated increased by more than 44,000 (Meese 2017).

Placer County Plan Area

Tricolored blackbird populations in Placer County estimated from statewide surveys have fluctuated since 1994 (Table 2). Most of its historical nesting and foraging habitats are near the cities of Lincoln and Roseville. Rapid development in these areas may reduce their overall suitability for nesting by conversion of existing freshwater marshes, agricultural lands, and pastures to other land uses.

Table 2. Number of tricolored blackbirds counted in Placer County from 1994 – 2014 (Kelsey 2008, Meese 2014).

Year	Number
1994	1,000
1997	658
1999	4,500
2000	6,200
2001	2,800
2005	1,600
2008	12,050
2014	17,600
2017	960

Natural History

The habitat requirements, ecological relationships, life history, and threats to tricolored blackbird described below are summarized in diagram form in the envirogram (Envirogram 4 *Tricolored Blackbird*).

Habitat Requirements

Tricolored blackbird has three basic requirements for selecting its breeding colony site: open accessible water; a protected nesting substrate, including flooded, thorny, or spiny vegetation; and suitable foraging habitat providing adequate insect prey within a few miles of the nesting colony (Hamilton et al. 1995; Beedy and Hamilton 1997, 1999). Tricolored blackbird requires open water within 1,640 feet for colony settlement (Hamilton 2004). Almost 93% of the 252 breeding colonies reported by Neff (1937) were in freshwater marshes dominated by cattails (*Typha* spp.) and bulrushes (*Scirpus* sp.). The remaining colonies in Neff's study were in willows (*Salix* spp.), blackberries (*Rubus* sp.), thistles (*Cirsium* and *Centaurea* spp.), or nettles (*Urtica* sp.). In contrast, only 53% of the colonies reported during the 1970s were in cattails and bulrushes (DeHaven et al. 1975a).

Proximity to suitable foraging habitat appears to be extremely important for the establishment of colony sites, as tricolored blackbirds usually forage, at least initially, in the field containing the colony site (Cook 1996). However, often only a minor fraction of the area within the commuting range of a colony provides suitable foraging habitat (Beedy and Hamilton 1999). An increasing percentage of tricolored blackbird colonies in the 1980s and 1990s were reported in Himalayan blackberry (*Rubus discolor*) (Cook 1996), and some of the largest recent colonies have been in silage and grain fields (e.g., triticale) (Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000). In the Sacramento Valley, 67% of the colonies were found on Himalayan blackberry (Kelsey 2008). Other substrates observed to be used by tricolored blackbird for nesting include giant reed (*Arundo donax*), safflower (*Carthamus tinctorius*) (DeHaven et al. 1975a), tamarisk trees (*Tamarix* spp.), elderberry (*Sambucus* spp.) and poison oak (*Toxicodendron diversilobum*) (Beedy and Hamilton 1999). In addition, triticale, a vigorous wheat and rye hybrid grown to feed the dairy cows, has become an important nesting substrate accounting for nearly half of all early-season nesting and breeding sites and more than half of all known reproduction in 2005 (Hamilton and Meese 2006).

With the loss of a natural flooding cycle and most native wetland and upland habitats in the Central Valley, breeding tricolor blackbird now forages primarily in managed habitats. Ideal foraging conditions for tricolored blackbird is created when shallow flood-irrigation, mowing, or grazing keeps the vegetation at an optimal height (<6 inches) (Tricolored Blackbird Working Group 2009). Foraging habitats in all seasons include annual grasslands; wet and dry vernal pools and other seasonal wetlands; agricultural fields (e.g., rice, alfalfa, irrigated pastures, and ripening or cut grain fields); cattle feedlots; and dairies. Tricolored blackbird also forages occasionally in riparian scrub habitats and along marsh borders. Weed-free row crops and intensively managed vineyards and orchards do not serve as regular foraging sites (Beedy and Hamilton 1997, 1999).

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

Wintering tricolored blackbirds often congregate in huge, mixed-species blackbird flocks that forage in grasslands and agricultural fields with low-growing vegetation and at dairies and feedlots (Beedy 2008).

Foraging

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

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

Reproduction

Tricolored blackbird breeding extends from mid-March through early August (Beedy and Hamilton 1999). Autumnal breeding (i.e., September through November) has been documented at sites in the Central Valley (Orians 1960, Payne 1969). Tricolored blackbird is closely related to red-winged blackbird

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

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

Egg laying can begin as early as the second day after nest initiation but ordinarily starts about 4 days after the local arrival of tricolored blackbirds at breeding sites (Payne 1969). One egg is laid per day, and clutch size is typically 3-4 eggs (Payne 1969; Hamilton et al. 1995). Emlen (1941) and Orians (1961b) estimated the incubation period at 11 or 12 days, while Payne (1969) estimated it to be 11to 14 days. About 9 days generally elapse from hatching until the oldest nestling is willing to jump from the nest when disturbed. Young require about 15 days from this prefledging date until they are independent of their parents. Thus, one successful nesting effort for a reproductive pair takes about 45 days (Hamilton et al. 1995).

Low reproductive success has been recorded for tricolored blackbirds. Higher reproductive success has been found to be associated with greater abundance of favored insect groups in foraging habitats surrounding colonies. Meese (2013) documented widespread reproductive failures of entire colonies from 2006 to 2011 that appeared unrelated to nesting substrate. Instead, Meese (2013) found that insect abundance around these colonies was insufficient to support successful breeding, resulting in nestling starvation and failure of females to lay eggs.

Dispersal Patterns

DeHaven et al. (1975b) found that tricolored blackbird is unlikely to nest at the sites where they hatched or where they had nested the year before (n = 298 recoveries from 45,660 banded birds). However, breeding colonies often exhibit site fidelity and traditionally use many of the same areas year after year if these sites continue to provide essential resources such as secure nesting substrates, water, and suitable foraging habitats (Beedy et al. 1991; Hamilton et al. 1995; Beedy and Hamilton 1997; Hamilton 2000). As discussed in *Movement and Migratory Patterns* below, the distribution of tricolored blackbird in the Central Valley varies according to relatively predictable, seasonal movements. In Placer County, the species may number in the thousands at a colony site in one year and be absent the next year. Over the years of 2008, 2011, 2014 and 2015, 13 total sites were colonized, with an average of 5.75 of the 13 colonies occupied in any given year (Table 1).

Longevity

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

Sources of Mortality

Entire colonies (up to tens of thousands of nests) in cereal crops and silage are often destroyed by harvesting and plowing of agricultural lands in the San Joaquin Valley (Beedy and Hamilton 1999, Graves 2013). The concentration of a high proportion of the known population in a few breeding colonies increases the risk of major reproductive failures, especially in vulnerable habitats such as active agricultural fields. Harvesting of silage grains in locations where colonies have settled causes complete breeding failure of many thousands of birds for at least one breeding attempt (Tricolored Blackbird Working Group 2009).

Historical accounts documented the destruction of nest contents of entire nesting colonies by a diversity of avian, mammalian, and reptilian predators (Beedy and Hamilton 1999). Recently, especially in permanent freshwater marshes of the Central Valley, the contents of nests of entire colonies have been lost to black-crowned night heron (*Nycticorax nycticorax)* and common raven (*Corvus corax*). Some large colonies (up to 100,000 adults) may lose more than 50% of nests to coyotes (*Canis latrans*), especially in silage fields, but also in freshwater marshes when water is withdrawn (Hamilton et al. 1995).

Various poisons and contaminants have caused mass mortality of tricolored blackbird. McCabe (1932) described the strychnine poisoning of 30,000 breeding adults as part of an agricultural experiment. Neff (1942) considered poisoning to regulate numbers of blackbirds preying upon crops (especially rice) to be a major source of mortality. This practice continued until the 1960s, and thousands of tricolored and other blackbirds were exterminated to control damage to rice crops in the Central Valley.

Beedy and Hayworth (1992) observed a complete nesting failure of a large colony (about 47,000 breeding adults) at Kesterson Reservoir in Merced County; selenium toxicosis was diagnosed as the primary cause of death. At a Kern County colony, all eggs sprayed by mosquito abatement oil failed to hatch (Beedy and Hamilton 1999). Hosea (1986) attributed the loss of at least two colonies to aerial herbicide applications.

Behavior

Males defend only the immediate areas around the nests. Male territory size ranges from 19 square feet (Lack and Emlen 1939) to 35 square feet (Orians 1961b). Average size of recently established territories of six banded males at two different colonies was 35 square feet; volumetric territories in willows were calculated to be 300–400 cubic feet (Collier 1968). Some Himalayan blackberry colonies have nesting densities up to six nests/m^2 (0.56 nest/square foot) (Cook, pers. comm.; Hamilton pers. comm.). After one week of nest-building and egg-laying, males may cease territorial defense (Orians 1961b).

Tricolored blackbird generally forages within 3 miles of the colony site (Orians 1961a), but commutes distances of over 9 miles have been reported (Beedy and Hamilton 1999). Short-distance foraging (i.e., within sight of the colony) for nestling provisioning also is common. Both sexes are known to provision the nestlings (Beedy and Hamilton 1999).

Proximity to suitable foraging habitat appears to be extremely important for the establishment of colony sites, as tricolored blackbird always forages, at least initially, in the field containing the colony site (Hamilton and Meese 2006, Cook 1996). However, usually only a minor fraction of the area within the commuting range of a colony provides suitable foraging habitat. For example, within a 3 mile radius there may be low-quality foraging habitats such as cultivated row crops, orchards, vineyards, and heavily grazed rangelands in association with high-quality foraging areas such as irrigated pastures, lightly grazed rangelands, vernal pools, and recently mowed alfalfa fields (Beedy and Hamilton 1999; Cook 1999). Tricolored blackbird has been documented to travel more than 8 kilometers in search of animal prey with which to feed their young (Hamilton and Meese 2006).

Movement and Migratory Patterns

During the breeding season, tricolored blackbird exhibits itinerant breeding whereby individuals often move after their first nesting attempts and breed again at a different geographical location (Hamilton 1998). In the north Central Valley and northeastern California, individuals move after first nesting attempts, both successful and unsuccessful (Beedy and Hamilton 1997). Banding studies indicate that significant movement into the Sacramento Valley occurs during the postbreeding period (DeHaven et al. 1975b).

Wintering Tricolored Blackbird populations move extensively throughout their range during the non-breeding season. In winter, the number of tricolored blackbirds decreases in the Sacramento Valley and increases in the Sacramento–San Joaquin River Delta and north San Joaquin Valley (Neff 1937; Orians 1961a; Payne 1969; DeHaven et al. 1975b). By late October, large flocks also congregate in pasturelands in southern Solano County and coastal areas near dairies in Marin and Monterey Counties (Shuford and Gardali 2008, Beedy and Hamilton 1999). Other birds winter in the central and southern San Joaquin Valley. Concentrations of more than 15,000 wintering Tricolored Blackbirds may gather at one location and disperse up to 20 miles to forage (Neff 1937; Beedy and Hamilton 1999). Individual birds may leave winter roost sites after less than 3 weeks and move to other locations (Collier 1968), suggesting winter turnover and mobility. In early March/April, most birds vacate the wintering areas in the Central Valley and along the coast and move to breeding locations in the Sacramento and San Joaquin Valleys (DeHaven et al. 1975b).

Ecological Relationships

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

Population Threats

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

Habitat Loss and Alteration

Most native habitats that once supported nesting and foraging tricolored blackbird in the Central Valley has been lost or degraded. In 1939, only 560,500 of an original 4,000,000 acres (about 4%) of wetlands in the Central Valley were extant. By the mid-1980s, an estimated 480,000 acres of freshwater emergent marshes, or 85% of the total remaining freshwater wetlands in 1939, were reduced by one-half to about 243,000 acres (Beedy and Hamilton 1997). Much of the Central Valley has been altered by urbanization and unsuitable agricultural uses, including vineyards, orchards, and row crops (Frayer et al. 1989; Wilen and Frayer 1990). In Sacramento County, a historic breeding center of the species, the conversion of grassland and pastures to vineyards expanded from 7,536 acres in 1996 to 13,171 acres in 1998 (DeHaven 2000). The total vineyard lands in Sacramento County expanded further from over 16,500 acres in 2005 to over 21,200 in 2013 (Center for Biological Diversity 2015). Many former agricultural areas within the historical range of tricolored blackbird are now being urbanized; in western Placer County, where tricolored blackbird forages in the ungrazed annual grasslands associated with rural subdivisions, suitable habitat will be largely eliminated as current land conversion patterns continue.

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

Other Human Activities

Nests and nest contents in cereal crops and silage are often destroyed by agricultural operations (Hamilton et al. 1995; Beedy and Hamilton 1997). Harvest of grain silage is conducted in relation to moisture content of the forage, the timing of which coincides with tricolored blackbirds using the crops for nesting (USFWS 2000). This causes nest destruction and direct mortality, which in turn is threatening much of the remaining breeding population of the species (USFWS 2000). Harvesting of silage and plowing of weedy fields are currently the most common reasons tricolored blackbird nesting colonies are destroyed on agricultural lands. In 2014, it was reported 38% of all nesting substrate consisted of silage (e.g., triticale) (Meese 2014). The concentration of most of the tricolored blackbird reproductive effort into a few large colonies that are selecting grain silage as a nesting substrate has greatly increased the risk of threats to the species should the annual destruction of such a large proportion of nests continue unabated (Cook and Toft 2005).

Other factors that may affect the nesting success of colonies in agricultural areas include herbicide and pesticide applications and spraying for mosquito abatement (Beedy and Hamilton 1999). Beedy and Hamilton (1999) observed a colony sprayed by mosquito abatement operators in Kern County and all sprayed eggs failed to hatch. In addition, the loss of at least two tricolored blackbird colonies was attributed to herbicide applications (Beedy and Hamilton 1999). Beedy and Hayworth (1992) observed a complete nesting failure of a large colony (about 47,000 breeding adults) at Kesterson Reservoir in Merced County, and selenium toxicosis was diagnosed as the primary cause of death.

Predation

Predation is possibly a major cause of complete nesting failure at some tricolored blackbird colonies in the Central Valley. Historical accounts documented the reproductive failure of nesting colonies to predation of nest contents by a diversity of avian, mammalian, and reptilian predators. Heron and raccoon predation upon colonies nesting in marshes can destroy all or nearly all nests within colonies (Hamilton et al. 1995, Hamilton 2000). Entire colonies (>50,000 nests) have been lost to black-crowned night herons, common ravens, coyotes, and other predators, especially in permanent freshwater marshes of the Central Valley (Beedy and Hamilton 1999). More recent studies have documented wholesale reproductive failure of entire colonies due to predation by cattle egrets (Meese 2013).

Context for a Regional Conservation Strategy

Tricolored blackbird breeding colonies have been reported in western Placer County in recent years and are treated as present; however, because the distribution and abundance of breeding colonies varies annually, the current breeding population at a given colony site may be small or absent. Therefore, currently unoccupied colony sites that provide suitable habitat characteristics retain conservation value as sites that may be used in the future. Table 1 lists 21 tricolored blackbird colony sites and aggregations within Plan Area A, of which 15 are active or recently active (a colony site is assumed active if tricolored blackbirds were documented nesting at a site within the prior 10 years). Within a breeding season, surveys have found tricolored blackbirds at 2-6 colony sites in Plan Area A. Regular monitoring of colony sites has confirmed breeding at four sites in 2014 and five in 2015 (Airola pers. comm.). Of the 15 active or recently active colony sites found in Plan Area A, six are in the RAA, 3-4 are protected in Existing Protected Areas, and five are in the PFG.

In California, species occurrences are scattered throughout much of the state, with densities greatest in the Central Valley and surrounding lowlands. In the Sierra Nevada foothill region, tricolored blackbird has been recorded from all counties surrounding western Placer, including Sacramento, western Yuba, Butte, Sutter, Colusa, Glenn and Yolo counties. Placer County is; therefore, not highly significant in terms of the species' distribution and range. However, Placer County has been found to support a considerable proportion of the regional population (see above: *Distribution, Placer County Plan Area, Current*). The general decline of breeding colonies in the state and the rapid urbanization of previously occupied sites in the Plan Area lend value to remaining populations and suitable habitat.

A study on tricolored blackbirds was conducted in seven counties that contribute in some part to the Sierra Nevada foothills ecoregion, which includes grassland dominated regions of lower elevations in Placer County. The 2014 tricolored blackbird nesting population of foothills in these counties was 43,009 birds, of which 12,473 (29%) occurred in Placer County (Airola et. al. 2015). The relatively large number of birds that bred successfully in the Sierra Nevada foothills grasslands, which includes Placer County, in a year of historic drought when the number of breeding birds in the San Joaquin Valley was 78% lower than in 2008 (Meese 2014) suggests that the Sierra Nevada foothills region may play a significant role in tricolored blackbird species conservation (Airola et. al. 2015).

Tricolored blackbird colonies will breed at freshwater marsh dominated by cattails and bulrushes, or in other flooded or thorny vegetation such as willows, blackberries, thistles, or nettles at open and accessible water. The species will also use agricultural fields for nesting, such as silage and grain fields. Suitable foraging habitat within a few miles of the nesting colony is required. Tricolored blackbird will forage over annual grasslands, wet and dry vernal pools and other seasonal wetlands, agricultural fields, cattle feedlots and dairies. Ungrazed vernal pool grassland complexes and rice fields characterize the landscape in much of the species' breeding range and preferred foraging habitats in western Placer County. For the conservation of tricolored blackbird in the Plan Area, acquisition and protection of the habitats described above, including current and past colony sites, is of highest priority. Meese et. al. (2015) states that locations with relatively higher average reproductive success should be preferred targets for conservation investments and that land uses within a 5 kilometer radius of the nesting sites should be considered equally as important as nesting vegetation.

Modeled Species Distribution in the Plan Area

Model Assumptions

<u>Nesting Habitat</u>

Tricolored blackbird nesting habitat includes the marsh complex land cover type below 300 feet elevation.

<u>Foraging Habitat</u>

Modeled foraging habitat for tricolored blackbird emphasizes the open cover below 300 foot elevation which is mapped as vernal pool complex, annual grassland, pasture, alfalfa, and cropland. While tricolored blackbird may forage in rural residential, urban golf courses, urban parks, and urban wetland, those rural residential and urban/suburban land-cover types are not included in modeled foraging habitat for tricolored blackbird.

Rationale

Tricolored blackbirds breed and overwinter in the Plan Area; therefore, foraging habitat may be used year-round. Tricolored blackbirds have three basic requirements for selecting their breeding colony sites: open accessible water; a protected nesting substrate, including either flooded, thorny, or spiny vegetation; and suitable foraging habitat providing adequate insect prey within a few miles of the nesting colony. Tricolored blackbirds require open water within 1,500 feet for colony settlement. Historically, most breeding colonies were in freshwater marshes dominated by cattails and bulrushes (*Schoenoplectus* sp.), with a small percentage in willows (*Salix* spp.), blackberries (*Rubus* sp.), thistles (*Cirsium* and *Centaurea* spp.), or nettles (*Urtica* sp.). Documented tricolored blackbird colonies in the Plan Area are mostly located in large stands of Himalayan blackberry. The scale of the land cover model does not delineate stands of Himalayan blackberry, so Himalayan blackberry stands are not included as nesting habitat in this model Such stands are occasionally found within annual grassland and valley foothill riparian land-cover types, so nesting habitat should be suitably captured by including these layers as potential nesting habitat. These stands comprise a small percentage of the total amount of these land-cover types; therefore, the total acreage for nesting habitat is an overestimate.

Model Results

Species Map 4. *Tricolored Blackbird Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for tricolored blackbird in the Plan area. Potential tricolored habitat is distributed throughout much of the Plan Area below 300 foot elevation; there is considerably more secondary habitat than primary habitat. All of the known occurrences of nesting colonies fall within the modeled habitat, the majority of them within primary habitat. Those that do not occur in stands of Himalayan blackberry not distinguished in the GIS land-cover layers.

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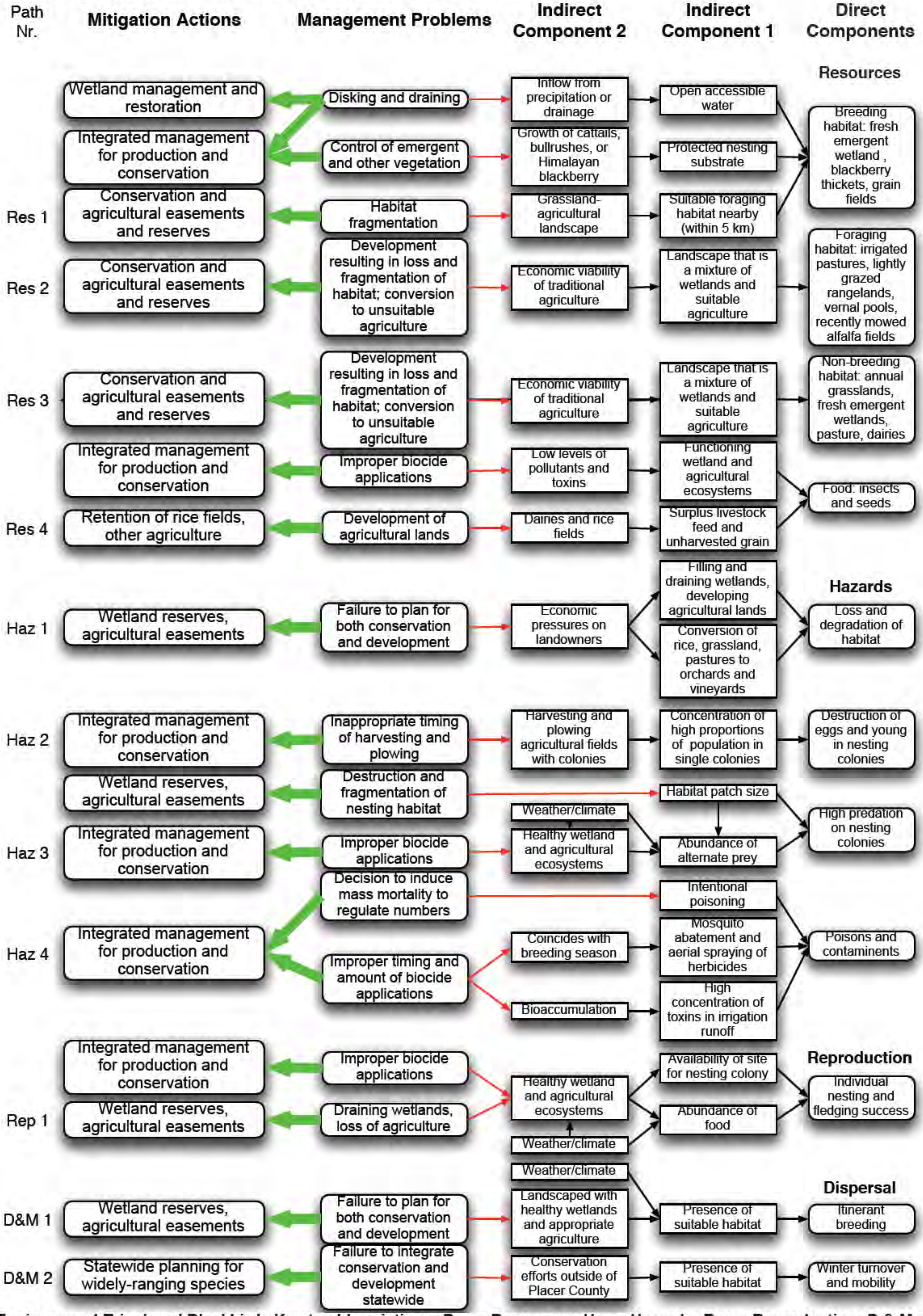
Envirogram 4 Tricolored Blackbird, *Agelaius tricolor*



Envirogram 4 Tricolored Blackbird. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; D & M = Dispersal and Migration.

Envirogram Narrative

Tricolored Blackbird (*Agelaius tricolor*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary. For example, resources are subdivided into breeding habitat, foraging habitat, non-breeding habitat, and food.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1**:** Breeding habitat for tricolored blackbird requires open accessible water, protected nesting substrate, and suitable foraging habitat within three miles. Disking and draining of wetlands has reduced the first requisite, which could be mitigated by better wetland management or restoration. Cattails and bulrush originally were the preferred nesting substrate, but upland shrubs and vines (including the introduced Himalayan blackberry) can substitute provided that they are not eliminated by vegetation management. Suitable foraging habitat can be found in large, contiguous grassland-agricultural landscapes that have not been converted to orchards, vineyards, or row crops. Conservation and agricultural easements can ensure that these large habitat patches will still be present in Placer County.

Res2**:** Foraging habitat includes fresh emergent wetlands, irrigated pastures, lightly grazed rangelands, vernal pools, and recently mowed alfalfa fields. Such habitat can be found in large, contiguous grassland- agricultural landscapes that have not been developed or converted to orchards, vineyards, or row crops. The persistence of such landscapes in Placer County depends on the continued viability of traditional agriculture. Conservation and agricultural easements may ensure that suitable landscapes will still be present in Placer County.

Res3**:** Non-breeding habitat is similar to foraging habitat, but requires the presence of more grain and seeds because insects are less available (see path Res4). The presence of suitable non-breeding habitat has the same requirements as foraging habitat (path Res2).

Res4**:** Tricolored blackbird eats primarily insects during the breeding season and mostly seeds and grain at other times of the year. The presence of adequate insect prey depends on wetland and agricultural ecosystems that have low levels of pollutants and toxins—a result of proper management. An integrated approach to agricultural production and conservation can help insure adequate insect abundance. Much of the grain consumed by the tricolored blackbird during the non-breeding season comes from surplus livestock feed or unharvested rice. Thus, dairies and rice fields are important to this species, and these enterprises must be conserved through conservation and agricultural easements.

Hazards

Haz1**:** Continued loss of habitat is a hazard faced by the tricolored blackbird. Filling and draining wetlands and developing agricultural lands are one source of loss; conversion of rice fields, grasslands, and pastures to row crops, orchards, and vineyards are another. Both of these problems result from economic pressures on landowners that are the legacy of a failure to plan for both conservation and development. Wetland reserves and agricultural easements can help alleviate these pressures.

Haz2**:** Destruction of eggs and young in entire nesting colonies can be catastrophic for the species' annual reproductive success because such a large proportion of its total population can be found in a single colony. Harvesting and plowing during the nesting season is a cause of the destruction; it can be mitigated by integrating production agriculture with conservation. Easements or other kinds of economic incentives may be necessary.

Haz3**:** Entire nesting colonies have been decimated by predators, including herons, ravens, and coyotes. Lack of alternate prey and easy access to breeding colonies because of habitat fragmentation facilitate such mass predation events. Maintaining large, unfragmented wetland and agricultural ecosystems through reserves, easements, and proper management practices can minimize this hazard.

Haz4**:** Various poisons and contaminants are another hazard for the tricolored blackbird. Entire colonies have been eliminated by intentional poisoning with strychnine in the past. Aerial spraying for mosquito abatement or weed and pest control still can have severe consequences for breeding colonies either directly or indirectly through its effects on insect prey. Various toxins such as selenium that are found in irrigation tailwater can bioaccumulate and cause mortality and reproductive failure as well. Integrated management for agricultural production and conservation, especially on the timing and management of pesticides, can help minimize this hazard.

Reproduction

Rep1**:** Individual nesting and fledging success depends on the presence of an appropriate habitat for colonial nesting and sufficient food to sustain the colony. While food abundance depends to some extent on weather conditions, the presence of healthy wetland and agricultural ecosystems is critical. Wetland reserves, agricultural easements, and proper management of pesticides and land use help ensure that reproduction will succeed.

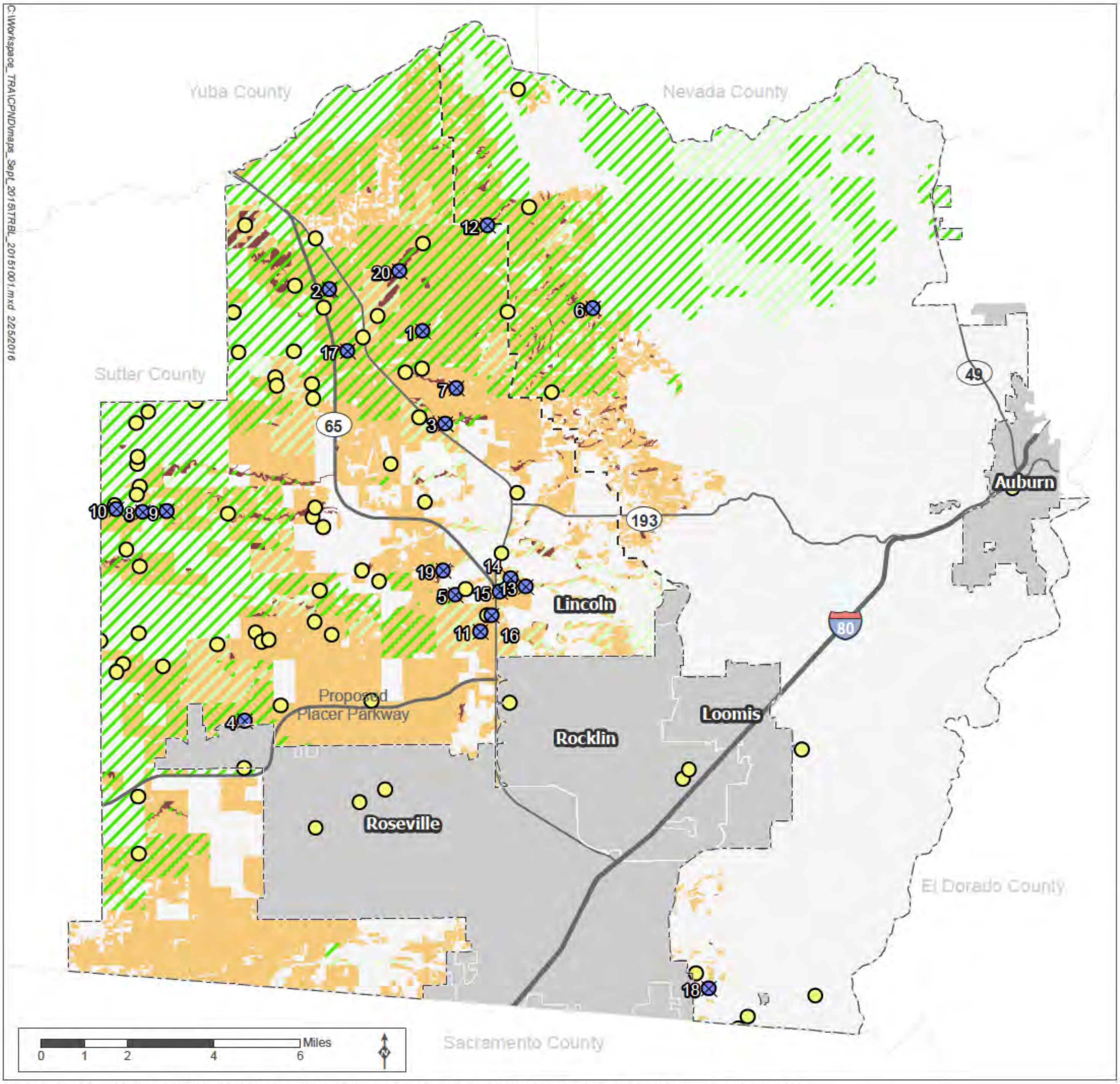
Dispersal and Migration

D&M1: Tricolored blackbird is an itinerant breeder; areas that supported breeding colonies one year may be empty in others. Habitat suitability depends to some extent on weather conditions, but primarily on landscapes that are mixtures of healthy wetlands and appropriate agriculture. Past planning failures that are responsible for the disappearance of such landscapes can be mitigated partially by creating wetland and agricultural reserves now.

D&M2: Tricolored blackbird colonies wander throughout the greater Central Valley-Delta region during the non-breeding season as well as remain near breeding areas. Clearly, the survival of this species requires habitat throughout the entire region, and that will depend upon conservation activities outside of Placer County. Statewide planning efforts are necessary for the continued existence of this species.

Summary

Tricolored blackbird, originally a wetland species, adapted to agricultural areas after most of the Central Valley's wetlands were lost. The continued existence of this species will not only depend on wetland conservation and restoration but also on conserving rice farms, dairies, and ranches and managing these operations in ways that are compatible to the species' needs. Agricultural/conservation easements in Placer County should be very specific about these management goals.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2015; eBird, 2012; Tricolored Blackbird Portal, 2015; eBird, 2015; Placer Land Trust, 2010; Restoration Resources, 2005

Occurrences
- Nest Colony Sites - Tricolored Blackbird Portal
- Other Sightings - eBird and Other Sources

Modeled Habitat
- Nesting Habitat
- Foraging Habitat
- Non-habitat
- Existing Protected Area
- Reserve Aquisition Area
- Non-participating City
- Major Road
- Valley/Foothill Divide
- Area A Boundary

Colony #	Colony Name	Colony #	Colony Name
1	Chamberlain Ranch	11	Orchard Creek
2	Dalby East of Hwy 65 Lincoln Bypass	12	Riosa Road
3	Gladding Highway 65	13	Rodeo Ground Open Space 1
4	Gleason Ranch on Sunset Blvd West	14	Rodeo Ground Open Space 2
5	Industrial Avenue	15	Rodeo Ground Open Space 3
6	Little Ben	16	Twelve Bridges and Hwy 65 West
7	Manzanita Road	17	Waltz East Hwy. 65 Bypass
8	Markham Ravine #1	18	Wellington Way
9	Markham Ravine #2	19	West Ferrari Ranch Road
10	Markham Ravine #3	20	Yankee Slough

MIG | TRA

Species Map 4.

Tricolored Blackbird Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

Giant Garter Snake
(*Thamnophis gigas*)

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Status

Federal: Threatened (USFWS 1993)

State: Threatened

Recovery Plan: Draft Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*) (USFWS 1999); Recovery Plan for the Giant Garter Snake (USFWS 2017).

Critical Habitat: No critical habitat rules have been published for the giant garter snake.

Distribution

California

Giant garter snake is endemic to California, found only in the Sacramento and San Joaquin Valleys (Fitch 1941; Hansen and Brode 1980; Rossman and Stewart 1987; USFWS 1999). Records of giant garter snakes coincide roughly with the historical distribution of the large flood basins, freshwater marshes, and tributary streams of the Central Valley of California (Hansen and Brode 1980). The distributional range of this species probably extended from Butte County in the north to Buena Vista Lake in Kern County in the south. The eastern and western boundaries of the range are believed to be the foothills of the Coast Ranges and the Sierra Nevada (USFWS 1999). Rossmann et al. (1996) described an elevation range for giant garter snake of 0–400 feet. Occurrence records in the southern Sacramento Valley occurred between 10–40 feet elevation (Hansen 1986). Agricultural and flood control activities have extirpated the species from the southern one-third of its range in the former wetlands associated with the historic Buena Vista, Tulare, and Kern lakebeds (Hansen and Brode 1980; Hansen 1986, 1988; CDFG 1992[1]; USFWS 1999). Today, populations of giant garter snake are found in the Sacramento Valley and isolated portions of the San Joaquin Valley (USFWS 1999; USFWS 2006; CNDDB 2015). Recent records indicate 13 recognized populations distributed from the vicinity of Chico in Butte County to near Burrel in Fresno County (Hansen and Brode 1980; Rossman and Stewart 1987; USFWS 1999; USFWS 2006; Wood et al. 2015; CNDDB 2015). This range is currently divided into three recovery units including the Northern Sacramento Valley Recovery Unit (Butte, Colusa, and Sutter Basins), Southern Sacramento Valley Recovery Unit (American, Yolo, and Delta Basins), and San Joaquin Valley Recovery Unit (San Joaquin and Tulare Basins) (Wood et al. 2015). The recovery units are presumed to be distinct from one another based on ecological and geographical characteristics and unique recovery actions needed within them (USFWS 1993; 2006).

[1] As of January 1, 2013, the California Department of Fish and Game (CDFG) was renamed the California Department of Fish and Wildlife. When this document cites reports prepared by the Department prior to 2013, the reference includes the prior department name of CDFG. Both CDFW and CDFG refer to the same agency.

Placer County Plan Area

Historical

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

Current

There are no current records of giant garter snake within the Placer County Plan Area. However, suitable habitat occurs in the drainage network associated with agricultural fields in the western section of the County, from approximately Sheridan south to the area of Baseline Road and South Brewer Road (USFWS 1999; USFWS 2006; Dudek 2014). Several locations within this area are used for growing rice, and the associated agricultural ditches and wetlands/sloughs containing emergent vegetation in conjunction with suitable adjacent upland habitat could be used by giant garter snake during both the active and inactive seasons (Dudek 2014).

A total of 19 occurrences of giant garter snake have been reported within five miles to the west and south of the Placer County line in the Sutter and Natomas Basins of Sutter and Sacramento Counties (CNDDB 2015). The closest occurrence was recorded in the Natomas Basin of Sacramento County approximately 1.5 miles to the southwest of the Placer County line. Another population occurs in Auburn Ravine, west of the Plan Area in Sutter County (Paquin et al. 2006).

Population Status & Trends

California

The current distribution and abundance of giant garter snake is reduced and declining due to loss, degradation, and fragmentation of habitat (USFWS 1999). Despite the loss of 93% of historic wetlands throughout the Central Valley, giant garter snakes continue to persist in relatively small, isolated patches of highly modified agricultural wetlands (Wood et al. 2015). Giant garter snake have become increasingly fragmented in recent decades and persist in small clusters of populations primarily in agricultural canals and drains associated with rice agriculture and remnant managed wetlands (Halstead et al. 2010). Prior to 1970, the species was known from 17 populations (Hansen and Brode 1980). At the time of listing in 1993, 13 of these populations were extant; only three of these populations are currently considered stable and safe from threats. Populations of giant garter snake have been nearly extirpated from the San Joaquin Valley Recovery Unit where only a few isolated populations remain within the San Joaquin Basin. They are presumed extirpated further south of the San Joaquin Basin in the Tulare Basin: Buena Vista Lake, Kern Lake, and Tulare Lake (Dickert 2005 as cited in Wood et al. 2015).

Giant garter snake populations north of the Sacramento-San Joaquin Delta are believed to be relatively stable compared to the San Joaquin Valley where populations appear to be in notable decline (USFWS 2012). The previous USFWS status review for giant garter snake found that, of the 13 populations in the listing, the population at Burrell/Lanare in the San Joaquin Valley is likely extirpated and that several locality records in the San Joaquin Valley and within the Sacramento-San Joaquin Delta are threatened with extirpation (USFWS 2006). Surveys conducted since 2006 strongly indicate that populations at Burrell/Lanare and at Liberty Farms in Yolo County are extirpated (Hansen 2008 as cited in USFWS 2012). The other populations listed in the previous status review all appear to be extant. Giant garter snakes are known to be extant in Butte County, Glenn County, Colusa County, Sutter County, Sacramento County, Yolo County, Solano County, San Joaquin County, Contra Costa County, Merced County, and Fresno County (USFWS 2012).

Placer County Plan Area

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

Natural History

The habitat requirements, ecological relationships, life history, and threats to giant garter snake described below are summarized in diagram form in the Envirogram 5 Giant Garter Snake.

Habitat Requirements

Giant garter snake inhabits agricultural wetlands and associated waterways. These include irrigation and drainage canals, rice fields, marshes, sloughs, ponds, small lakes, low-gradient streams, and adjacent uplands (USFWS 1999; USFWS 2012). Features of these habitats important to giant garter snakes include: sufficient water during the snake’s active season (early spring through mid-fall) to maintain an adequate prey base; emergent vegetation such as cattails (*Typha* spp.) and bulrushes (*Scirpus* spp.) for escape cover and foraging habitat; upland habitat with grassy banks and openings to waterside vegetation for basking; and adjacent upland areas for cover and refuge from floodwaters during the species’ inactive season (Hansen 1980; Hansen 1988; Brode and Hansen 1992; Hansen and Brode 1993; USFWS 2012). Studies suggest that permanent wetlands with emergent vegetation harbor the greatest densities of giant garter snakes, and that wetlands that do not provide water during giant garter snakes inactive season (April to October) cannot support large populations of the giant garter snake (Wylie et al. 1997). In addition, irrigated pastures provide indirect habitat for giant garter snake because the pastures require early summer flooding of pastures and frequent irrigation—often from a maze of irrigation canals (Paquin et al. 2006; Paquin, pers. comm.). Giant garter snake primarily occurs where a dense network of canals exists among rice agriculture and wetlands (Halstead et al. 2010).

Giant garter snake is absent from larger rivers; wetlands with sand, gravel, or rock substrates; and from riparian woodland areas lacking suitable basking sites or suitable prey populations (Hansen 1980; Rossman and Stewart 1987; Brode 1988; Hansen 1988; USFWS 1999). Instead, Giant garter snake typically inhabits stagnant or slow-moving waterbodies with abundant emergent vegetation (Halstead et al. 2010).

Although many wildlife refuges within the range of giant garter snake contain wetlands, those that use “wet-soil management” do not provide suitable habitat for giant garter snakes (Paquin et al 2006). In wet-soil management, the wetlands are left to dry in the summer months in order to promote the growth of wetland plant species that provide food for overwintering waterfowl (Paquin et al 2006). Therefore, this type of management does not provide enough aquatic habitat during the snake’s active season.

According to the Draft Recovery Plan for the Giant Garter Snake (USFWS 1999), the ideal concept of a marsh managed as giant garter snake habitat should have shallow and deep water and variations in topography, including some higher ground resembling the ditch banks, or "islands", similar to a rice check. Rice fields contain warm shallow water with sheltering emergent vegetation (i.e. rice plants), which is present within the fields during the giant garter snake active season in the spring, summer, and early fall. During the late summer when rice fields contain large numbers of mosquito fish and Pacific chorus frogs (*Pseudacris regilla*), rice fields may provide important nursery areas for newborn giant garter snakes (Brode and Hansen 1992, Hansen and Brode 1993). The habitat and its associated water conveyance system, if managed properly, provides the giant garter snake ease of movement; protection from predators; warmth to aid metabolism, gestation, and digestion; and a source of food.

The diverse habitat elements of ricelands; the rice fields, tail water marshes, the ditch and drain components of the water conveyance system, delivery canals, and associated levees, all contribute structure and complexity to this man-made ecosystem. Giant garter snakes can survive in this artificial ecosystem because the spring and summer flooding and fall dry-down of rice culture coincides fairly closely with the biological needs of the species (USFWS 1999). Giant garter snake utilizes ricelands extensively and depends on them for habitat. In the spring, when the rice is planted and the fields are flooded with several inches of water, they contain prey species such as small fish or frogs attract giant garter snakes. In the summer, while the flooded rice continues to grow, giant garter snake continues to use rice fields as long as their prey are present in sufficient densities. In the late summer and fall, when the water is drained from the rice fields, giant garter snake moves off the fields to other adjacent habitats. Rice is harvested at this time and female garter snakes have just borne young and need food to regain their body weight; in the fall, the snake can get a good supply of food from the rice lands because prey are concentrated in the rice drains. In the winter, while the rice fields are fallow, giant garter snakes are dormant.

Within rice fields and the irrigation canals, giant garter snake also basks in openings in vegetation, created by riprap placed around water control structures. Giant garter snake uses small mammal burrows and other soil crevices above prevailing flood elevations during the winter (i.e., November to mid-March). Giant garter snake typically selects burrows with sunny exposures along south and west facing slopes (USFWS 1999). Small mammal burrows, crayfish burrows, and soil crevices provide retreats from extreme heat for giant garter snake during the active season (Hansen and Brode 1993). Wintering sites varied from canal banks and marsh locations, to riprap along a railroad grade near the marsh (Wylie et al. 1997). Wintering locations of radio-telemetered snakes tended to be in the vicinity of spring capture sites.

Individuals have been found using burrows as far as 164 ft from marsh edges during the active season, and as far as 820 ft from the edge of wetland habitats while overwintering, presumably to reach hibernacula above the annual high water mark (Hansen 1986, Wylie et al. 1997, USFWS 1999).

Reproduction

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

Foraging Behavior

Giant garter snake feeds primarily on fish and amphibians, taking advantage of pools that trap and concentrate prey (Brode 1988; Hansen 1980; Hansen 1988; Hansen and Brode 1993). Prey species include bullfrog (*Lithobates catesbeianus*), Pacific chorus frog, carp (*Cyprinus carpio*), mosquitofish

(*Gambusia affinis*), and blackfish (*Othodox microlepidotus*) (Fitch 1941; Fox 1952; Cunningham 1959; Hansen 1980; Brode 1988; Hansen and Brode 1993; Rossman et al. 1996).

Dispersal Patterns

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

Demography

Giant garter snake is about 8 inches long at birth. It typically doubles in size by one year of age (USFWS 1999); males usually reach sexual maturity in three years and females in five years. Sex ratios of adult females to males vary from 1:1 to 2:1, but this variance may be a function of capture methods employed in different studies (Hansen and Brode 1993; Wylie et al. 1997; USFWS 1999). Adult females are on average longer and heavier than males; males can reach 32.3 inches in snout-vent length (mean = 26.2 inches) and females can reach 42.5 inches snout-vent length (mean = 34.9 inches). Males weigh up to 10.2 ounces (mean = 4.9 ounces) and females weigh up to 27.7 ounces (mean = 15.3 ounces) (USFWS 1999).

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

Longevity

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

Sources of Mortality

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

Behavior

Home range estimates for giant garter snake based on radio telemetry data vary with location; estimates averaged 47 acres in Gilsizer Slough (n = 27; range: 2.0–640 acres);131 acres in Colusa National Wildlife Refuge (n = 29; range: 3.2–2,792 acres); and 23 acres at Badger Creek (n = 8; range: 10.4–202.6 acres) (USFWS 1999).

Movement, Migratory, and Activity Patterns

Giant garter snake is most active from early spring through mid-fall; activity is dependent on local weather conditions (Brode 1990; Hansen and Brode 1993). Giant garter snake begins to emerge from winter retreats around April 1. By the beginning of May, all giant garter snakes have usually emerged and are actively foraging. By about October 1, giant garter snakes begin seeking winter retreats. Foraging and other activities are sporadic at this time and dependent on weather conditions. By November 1, most snakes are in winter retreats and will remain there until spring. During winter, giant garter snake is generally inactive, although some individuals may bask or move short distances on warmer days (USFWS 1999). During the active season, giant garter snake generally remains near wetland habitats but can move more than 800 feet from the water (Hansen 1988; Wylie et al. 1997) during the day. Some individuals may move up to five miles over a period of several days if the conditions of their habitat become unsuitable (e.g., as a result of flooding) (Wylie et al. 1997).

As discussed above, giant garter snake uses burrows in the summer as much as 164 feet away from the marsh edge, whereas, overwintering snakes use burrows as far as 820 feet from the edge of marsh habitat (Wylie et al. 1997).

Genetic studies from six watersheds in the Sacramento Valley found significant genetic variation between watersheds with low interpopulation and interregion gene flow (Paquin et al. 2006). Studies also reveal that gene flow appears to be restricted across the major watershed basins, which lends support for naming the basins separate populations (USFWS 2012).

Ecological Relationships

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

Threats

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

The introduction of nonnative predators, including bullfrog, largemouth bass (*Micropterus salmoides*), and catfish (*Ictalurus* spp.), has been responsible for eliminating many species of native fishes and aquatic vertebrates in the western United States (Minkley 1973; Moyle 1976; Holland 1992). Exotic species have probably had detrimental effects on the giant garter snake through direct predation (sensu

Bury and Whelan 1984; Treanor 1993) and competition for smaller forage fish (CDFG 1992; 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. 1986; Saiki and Lowe 1987; USFWS 1993). Preliminary studies have documented potential bioaccumulative effects on giant garter snake or its prey species caused by agriculturally derived contaminants (Saiki et al. 1992, 1993). Disease and parasitism, potentially exacerbated by compromised immune response ability as a result of contaminant exposure, may also pose a threat to this species (USFWS 1999).

Populations across the Central Valley have been affected by diversion of water (i.e., dams, levees, and irrigation systems) and the expansion of agriculture for over a century. This has resulted in the loss of over 93% of historic wetlands in the Central Valley (USFWS 2006). Microsatellite analyses conducted by Wood et al. (2015) indicate that reductions in population size (i.e., genetic bottlenecks) have occurred in about half of the populations sampled in the Central Valley. Genetic evidence of bottlenecks was also observed in several northern populations, indicating that giant garter snake declines are not limited to the San Joaquin Valley (Wood et al. 2015). Small effective population sizes and geographic isolation leave these populations susceptible to stochastic events (i.e., disease and prolonged drought) and the deleterious consequences of genetic drift, both of which can lead to extinction of this species (Wood et al. 2015).

Climate change will likely adversely affect the giant garter snake (Halstead et al. 2010). Climate change models predict that the climate in the Sierra Nevada mountains will become drier (Hayhoe et al. 2004; Barnett et al. 2008), potentially shrining the area of habitats suitable for giant garter snake through drying of wetlands and cessation of rice agriculture as the cost of water increases (Halstead et al. 2010).

Conservation Considerations

Status of Recovery Planning

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

Compatible Land Uses

Rice fields currently provide a significant amount of giant garter snake habitat; however, flooding makes thousands of acres uninhabitable, and burning the fields in winter leaves snakes exposed to increased predation and thermal stress upon spring emergence. Establishing management practices that are compatible with giant garter snake ecology should enhance the perpetuation of the species. By changing the timing of water management and the method and timing of ditch and field maintenance, rice farmers can minimize impacts on this species (Engles 1994).

Context for a Regional Conservation Strategy

There are no records of giant garter snake in western Placer County; however, the species has been recorded in the region and specifically in neighboring Sutter and Sacramento counties and suitable habitat is present within the Plan Area. Specifically, Dudek (2014) identified suitable habitat for giant garter snake within the Plan Area from approximately Sheridan south to the area of Baseline Road and South Brewer Road (USFWS 1999; USFWS 2006; Dudek 2014). Several locations within this area are used for growing rice, and the associated agricultural ditches and wetlands/sloughs containing emergent vegetation in conjunction with suitable adjacent upland habitat could be used by giant garter snake during both the active and inactive seasons (Dudek 2014).

Records of giant garter snake are restricted to the Sacramento and San Joaquin Valleys. The widest range is within the Sacramento Valley, where there are historical or current records of giant garter snake from nine counties. As the western boundary of the Plan Area touches into the region of highest giant garter snake density based off of California Natural Diversity Database records, conservation of potential habitat within western Placer County is stressed. For the conservation of giant garter snake within the Plan Area, agricultural wetlands and associated waterways are of highest conservation and/or acquisition priority.

Modeled Species Distribution in the Plan Area

Model Assumptions

<u>Aquatic Habitat</u>

Modeled habitat includes the following land-cover types below 100 feet in elevation: ponds, fresh emergent marsh, flooded rice, and riverine (only smaller, low-gradient streams, tributaries, and canals).

<u>Upland Habitat</u>

Modeled habitat includes the following land-cover types below 100 foot elevation and within 200 feet of the edge of wetland habitats: annual grassland, pasture, alfalfa, irrigated pasture, unidentified croplands, vernal pool complex, and row crop.

Rationale

Giant garter snakes require sufficient water during the snake's active season (early spring through mid-fall) to maintain an adequate prey base; emergent vegetation for escape cover and foraging habitat; adjacent upland habitat with grassy banks and openings to waterside vegetation for basking; and adjacent upland areas for cover and refuge from floodwaters during the species' inactive season. They are known to inhabit agricultural wetlands and associated waterways including irrigation and drainage canals, rice fields, marshes, sloughs, ponds, small lakes, low-gradient streams, and adjacent uplands. Giant garter snakes inhabit small mammal burrows and other soil crevices above prevailing flood elevations throughout the winter dormancy period (November to mid-March). Individuals have been found using burrows as far as 165 ft from marsh edges during the active season, and as far as 820 ft from the edge of wetland habitats while overwintering, presumably to reach hibernacula above the annual high water mark (Hansen 1986, Wylie et al. 1997, USFWS 1999). Changing agricultural regimes, development, and other shifts in land use create an ever-changing mosaic of available habitat. Giant garter snakes move around in response to these changes in order to find suitable sources of food, cover, and prey. Connectivity between regions is therefore extremely important for providing access to available habitat and for genetic interchange. In an agricultural setting, giant garter snakes rely largely upon the interconnected network of canals and ditches that provide irrigation and drainage to provide this connectivity. Primary habitat includes breeding, foraging, and movement habitat because breeding habitat could not be differentiated from foraging and movement habitat at the resolution of the GIS land-cover data. Also, giant garter snake may use breeding habitat for foraging and movement. Upland habitats were modeled to include suitable land-cover types within 200 feet of the edge of wetland habitats as described in the 1997 Biological Opinion for USACE projects with relatively small effects on the giant garter snake in within Butte, Colusa, Glenn, Fresno, Merced, Sacramento, San Joaquin, Solano, Stanislaus, Sutter and Yolo Counties, California (USFWS 1997).

Model Results

Species Map 5. *Giant Garter Snake Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for giant garter snake within the Plan Area. The majority of the modeled habitat occurs in the far western portion of the Plan Area that supports flooded rice and other suitable agricultural lands.

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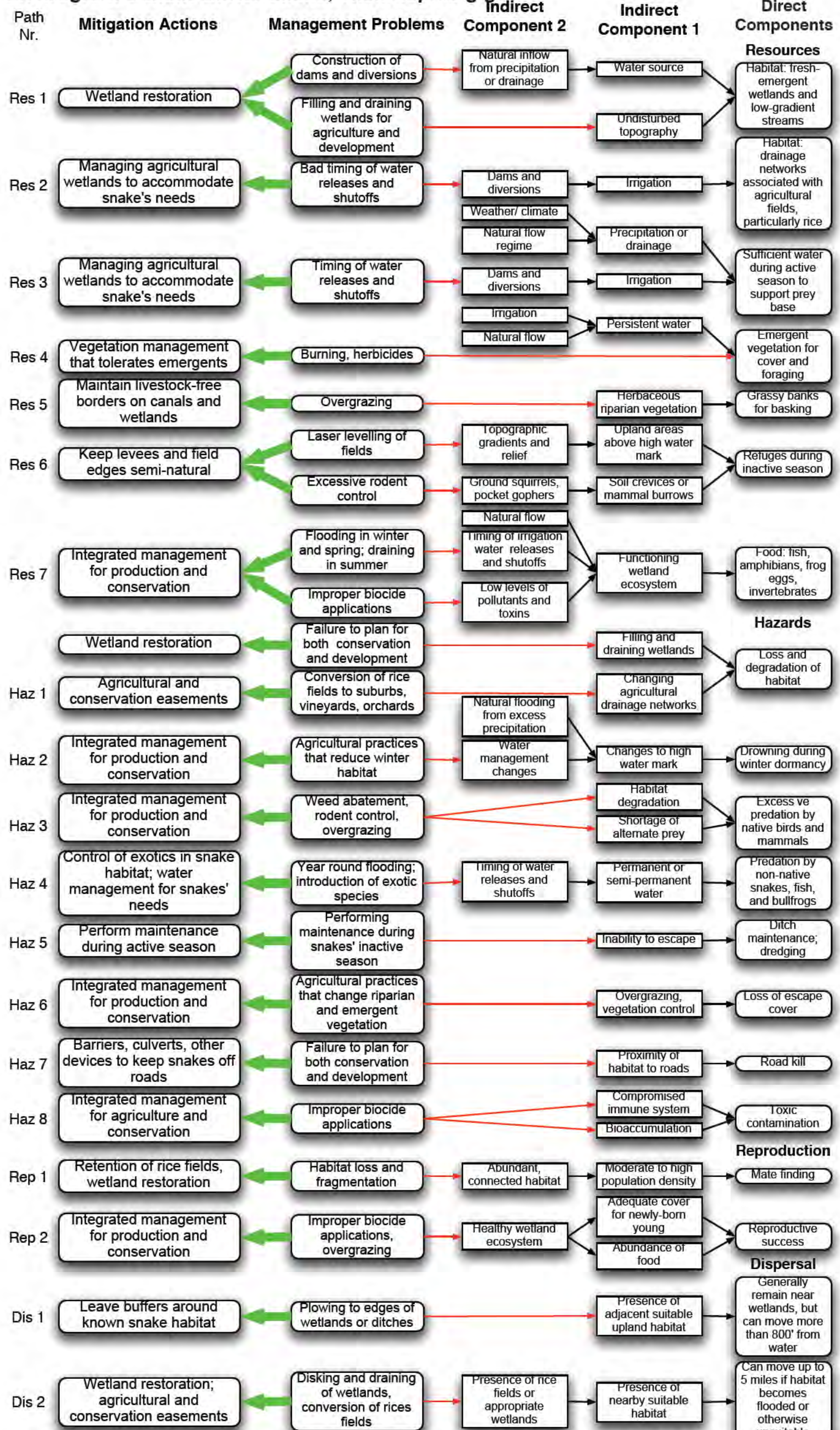
Envirogram 5 Giant Garter Snake, *Thamnophis gigas*



Envirogram 5 Giant Garter Snake. Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram Narrative

Giant Garter Snake (*Thamnophis gigas*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Giant garter snake originally was found in fresh emergent wetlands, ponds, small lakes with appropriate shoreline, and low gradient streams in the Central Valley. These areas required a particular topography and a water source, either precipitation or natural drainage. The construction of dams and diversions and the filling and draining of wetlands for agriculture and development has eliminated most of this habitat type. Substantial wetland restoration will be required to mitigate the loss.

Res2: Giant garter snake now mostly inhabits drainage networks associated with agriculture, particularly rice fields. The dams and diversions that helped destroy its original habitat have made possible the irrigation that maintains this new one. Bad timing of water releases and shutoffs can make these fields unsuitable for giant garter snake, however, so water management must take the giant garter snakes' needs into account.

Res3: During its active season the giant garter snake needs enough water in its habitat to support a prey base. The water can come either from precipitation and drainage and a natural flow regime or from irrigation. The suitability of irrigation water depends on the timing of water releases and shutoffs as in path Res2.

Res4: Giant garter snake needs emergent vegetation for cover and foraging habitat. This requires persistent water during the giant garter snake's active season, either from natural flow or irrigation. Burning or treating emergent vegetation with herbicide results in unsuitable habitat for the giant garter snake; emergent vegetation must be allowed to grow during the giant garter snake's active season.

Res5: Giant garter snake also requires grassy banks for basking. Thus, herbaceous riparian vegetation should not be overgrazed, and livestock should be excluded from the edges of fields and ditches.

Res6: During the inactive season, giant garter snake hibernates in mammal burrows or crevices above the high water line. This means that levees or natural topographic features must be present in otherwise level—or leveled—areas and that some rodent burrowing must be tolerated in giant garter snake habitat.

Res7: Giant garter snake feeds on fish, amphibians and their eggs, and invertebrates. The presence of these organisms requires a functioning wetland ecosystem with unpolluted water persisting during the giant garter snakes' active season. Proper pesticide application and timing of irrigation releases are critical.

Hazards

Haz1: Loss and degradation of habitat, either by filling and draining natural wetlands or by converting rice fields and other suitable agricultural areas to suburbs, vineyards, and orchards, is the major hazard to giant garter snake. Active wetland restoration and agricultural and conservation easements can help mitigate this loss.

Haz2: Drowning during winter dormancy is another hazard faced by the giant garter snake. Drowning occurs when normal high water marks are exceeded either by natural floods or by modifying water management practices. Agricultural practices that accommodate the giant garter snake's needs should be encouraged and made part of conservation easements.

Haz3: Excessive predation levels by native species can occur when alternate prey items are not available or the giant garter snake habitat has been degraded (usually by loss of cover). These problems can occur as a result of a variety of management actions including weed abatement, rodent control, and overgrazing. Integrated management for production and conservation could minimize these hazards.

Haz4: Predation by non-native snakes, fish (mostly cetrarchids), and bullfrogs is another hazard for giant garter snake. These introduced species live in permanent or semi-permanent waters, so shutting off irrigation water during the giant garter snake's inactive season, along with control efforts on the exotics, can help eliminate this problem.

Haz5: Giant garter snake can be killed during ditch maintenance or dredging if these activities occur during their inactive season. However, if these activities are conducted during the giant garter snake's active season they usually can escape.

Haz6: Loss of escape cover, through vegetation management or overgrazing, is another hazard for the giant garter snake. Integrated management for production and conservation needs to include protection of riparian and emergent vegetation during the giant garter snakes' active season.

Haz7: Snakes are killed by vehicles when roads are close to their habitat. Conservation areas should be well isolated from development; if this is not feasible, culverts and barriers should be installed to separate snakes from automobiles.

Haz8: Toxic contamination has been shown to be another hazard to the giant garter snake. Contaminants bioaccumulate and can result in weakened immune systems. Over-application of pesticides and the concentration of toxin-bearing runoff must be addressed in areas inhabited by this species.

Reproduction

Rep1: Population density should be adequate for mate finding in abundant, well connected habitat, but habitat loss and fragmentation have been severe in Placer County. Retention of rice fields and wetland restoration can help mitigate this problem.

Rep2: Giant garter snake bears live young (ovoviviparity). Reproductive success depends upon adequate food and escape cover for young giant garter snakes, both of which require healthy wetland ecosystems. Improper pesticide applications, overgrazing, and other activities that degrade these ecosystems must be addressed in management plans associated with conservation easements on farmland.

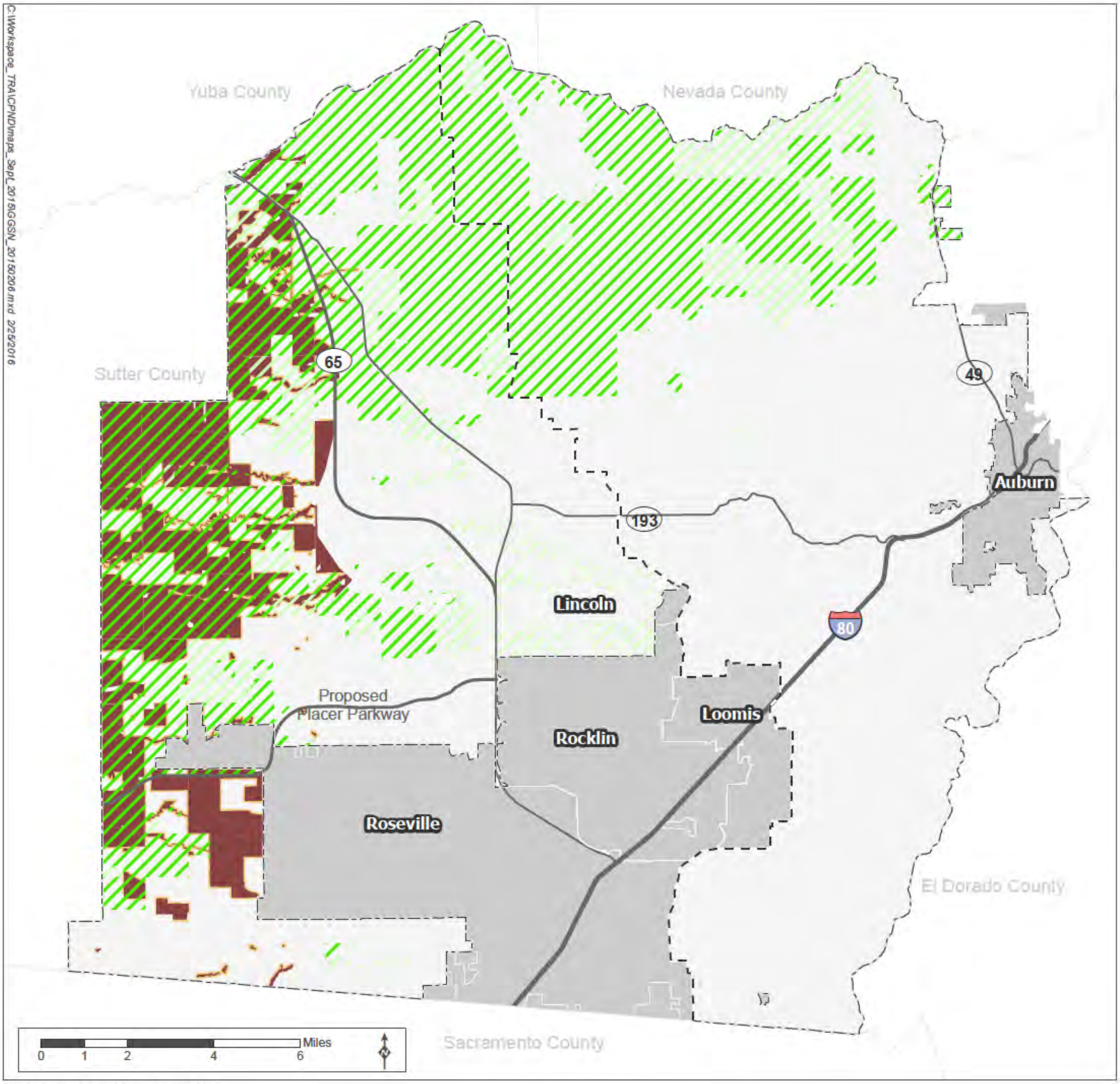
Dispersal

Dis1: Giant garter snake usually remains close to water, but has been known to venture 800 feet into adjacent upland areas. Thus, substantial buffers should be maintained around wetlands or rice fields known to support the snakes.

Dis2: If a habitat patch becomes unsuitable, giant garter snakes can move up to five miles to find a suitable one. Thus, the proximity of other wetlands or rice fields, managed to be compatible with the giant garter snake's needs and connected by suitable dispersal habitat, are critical to giant garter snake conservation.

Summary

The giant garter snake now depends almost entirely on agriculture, particularly rice growing, for its continued existence. Managing rice fields in ways compatible with the needs of giant garter snakes is quite possible, and these management prescriptions should be spelled out in agricultural/conservation easements. Restoring large fresh emergent wetlands would lessen the giant garter snake's dependence on agriculture.



Source: Placer County, 2014; MIG | TRA, 2015

Occurrences	Modeled Habitat		
None	Aquatic Habitat	Existing Protected Area	Major Road
	Upland Habitat	Reserve Aquisition Area	Valley/Foothill Divide
	Non-habitat	Non-participating City	Area A Boundary

MIG TRA

Species Map 5.

Giant Garter Snake Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

Western Pond Turtle
(*Emys marmorata*)

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Status

Federal: U.S. Fish and Wildlife Service was petitioned to list the species under the Endangered Species Act in 1992 (USFWS 1992), but the petition was rejected.

State: Species of Special Concern

Critical Habitat: Not Applicable (N/A)

Recovery Plan: N/A; though considered in the Draft Recovery Plan for the Giant Garter Snake (*Thamnophis gigas*) (USFWS 1999). A recovery plan has been developed by the State of Washington (Hayes et al. 1999).

Taxonomy

The western pond turtle was originally named and described *Emys marmorata* by Baird and Girard (1852) from a specimen found in the Puget Sound area. In 1945, Seeliger named and described two separate subspecies based on differing morphological characteristics: northwestern pond turtle (*Clemmys marmorata marmorata*) and southwestern pond turtle (*Clemmys marmorata pallida*). The northwestern subspecies was described by Seeliger (1945) as ranging from Puget Sound south to the Sacramento Valley in California. The southwestern subspecies was found to be from Monterey County south to Baja California Norte with intergradation occurring between the American River drainage and the Transverse Ranges in the San Joaquin Valley of California (Seeliger 1945).

A genetic study by Spinks and Shaffer (2005) suggested the existence of four unnamed clades within *Emys marmorata*, with the following geographical distribution: 1) from the Transverse Mountains (Los Angeles Mountains or range) south to Baja California Norte; 2) the San Joaquin Valley and associated foothills; 3) Ventura and Santa Barbara Counties in Central Coastal California; and 4) all remaining populations to the north. However, a more recent genetic study conducted by Spinks and Shaffer (2014) clarified that two primary clades exist and that subpopulations within each clade have been incorrectly interpreted as intergrades. As a result, Spinks and Shaffer (2014) proposed using the name *Emys marmorata* for all populations north of the San Francisco Bay area plus populations from the Great Central Valley north including an apparently introduced population in Nevada. Thus, *Emys pallida* is used for those populations inhabiting the central coast range south of the San Francisco Bay area to the species' southern range boundary, including the Mojave River. *Emys marmorata* and *Emys pallida* show very limited intergradation in a few populations in the northern central coast range and adjacent Sierra Nevada foothills, although at all intergrade sites pure individuals of the locally prevalent species were also found (Spinks and Shaffer 2014). In addition, although populations from Baja California are included in *Emys pallida*, these animals may also represent a distinct species pending results from additional analyses (Spinks and Shaffer 2014).

Distribution

North America

Western pond turtle is distributed along the North American Pacific Coast from Washington State to Baja California in Mexico. The species inhabits a variety of aquatic systems, mainly west of the Cascade-Sierra Nevada-Peninsula Mountains. Western pond turtle occurs from Puget Sound in Washington south through Oregon to the American River drainage in central California, and generally west of the Cascade-Sierra crest to the American River drainage. In the Central Valley, western pond turtle historically inhabited the vast permanent and seasonal wetlands of the area, with the Tulare Lake basin as a major population center (Hayes et al. 1999).

California

Historically, this species occurred in most Pacific slope drainages between the Oregon and Mexican borders and in only two drainages on the desert slope (i.e., the Mojave River in San Bernardino County and Andreas Canyon in Riverside County) (Jennings and Hayes 1994). Today, western pond turtle occurs in 90% of its historic range in the Central Valley and west of the Sierra Nevada mountains, but in greatly reduced numbers (Jennings and Hayes 1994; Germano and Bury 2001). It currently occurs from the Oregon border south to the San Francisco Bay Area and east through San Joaquin and Tuolumne County. The southwestern pond turtle is known from Santa Clara County south to the Mexican border.

Placer County Plan Area

<u>Historical</u>

Western pond turtle occurred in suitable habitat throughout the American River drainage, including the Placer County Plan Area (USFWS 1999).

<u>Current</u>

Four occurrences of western pond turtles have been documented within the Plan Area and vicinity (CNDDB 2015). This includes an occurrence at three locations on Coon Creek in Hidden Falls Park, within a reservoir north-northwest of Newcastle, on the western edge of Folsom Lake, and on the southern border of Placer County in the Baldwin Reservoir (CNDDB 2010).

Population Status & Trends

North America

Western pond turtle was once abundant in California, Oregon, and locally in Washington, but is declining in numbers throughout its range, particularly in Washington, northern Oregon, southern California and Baja California (Holland and Bury 1998; Hayes et al. 1999). Loss, degradation, and fragmentation of habitat are the primary factors contributing to the decline of the species (Hayes et al. 1999).

California

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

Placer County Plan Area

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

Natural History

The habitat requirements, ecological relationships, life history, and threats to western pond turtle described below are summarized in diagram form in the Envirogram 6 Western Pond Turtle.

Habitat Requirements

Western pond turtle inhabits a variety of aquatic habitats from sea level to elevations of 6,500 feet. It is found in rivers, streams, lakes, ponds, wetlands, reservoirs, brackish estuarine waters, canals and even sewage ponds (Holland 1994; Jennings and Hayes 1994; Germano and Bury 2001). Hatchling and young turtles (i.e., 1 year) require shallow water areas (i.e., less than 11.8 inches deep) dominated by emergent aquatic reeds, such as Juncus (*Juncus* sp.) and sedge (*Carex* sp.) (Holland 1991). Western pond turtle uses aquatic habitats primarily for foraging, thermoregulation, and avoidance of predators; it requires emergent basking sites, and has been observed to avoid areas of open water lacking them (Holland 1994). Basking sites can include rocks, logs, or emergent vegetation, and are used by the turtle for thermoregulation. Western pond turtle can be found in waters with temperatures as low as 34ºF, and rarely in water with temperatures exceeding 102–104ºF (Jennings and Hayes 1994).

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

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

Reproduction

Western pond turtles first breed at 10 to14 years of age (UFWS 1999). Mating generally occurs in late April or early May (Jennings and Hayes 1994). Most females lay eggs in alternate years. Clutch size ranges from 1 to 13 eggs, with larger females generally laying larger clutches (Holland 1985a, 1991a).

Females move inland 39–1,319 feet to upland habitat to nest from May through July, although this can occur as late as early August (Jennings and Hayes 1994). The eggs are best suited for development in dry, warm places because of their thin shells. Females typically dig the nest in soil with high clay or silt content on an unshaded slope (Jennings and Hayes 1994). Proximity of the nesting site to aquatic habitat is reliant on availability, but is generally within 650 feet of aquatic habitat, although it can be up to 1,320 feet away (Storer 1930; Jennings and Hayes 1994). Incubation lasts 80–100 days, and the normal hatch success is approximately 70%. Nest predation rates are high and complete failure of nests is common. In southern California, juveniles emerge from the nest in early fall (Holland 1994). Most hatchlings overwinter in the nest and move to water in March–April, although some leave the nest in September (Holland 1985a, 1991a, 1991b).

Demography

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

Dispersal Patterns

Males generally move greater distances than females or juveniles (Bury 1972a), but there is little movement between drainages (Holland 1991b). Measured home ranges of western pond turtle average 2.5 acres for males, 0.7 acre for females, and 1 acre for juveniles (Bury 1972a). Western pond turtles rarely move between drainages (Holland 1991a). Turtles may move up to 820 feet from aquatic habitat to overwinter under dense vegetation, logs, or leaf litter (Holland 1991a).

Foraging Behavior

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

Longevity

The maximum recorded age for western pond turtle is 39–40 years, but the expected longevity for this species probably reaches 50–70 years (Holland 1991a). On average, adult males have a higher probability of survivorship than adult females (Holland 1991a).

Sources of Mortality

Western pond turtle is preyed upon by a wide variety of native and introduced predators, including raccoon (*Procyon lotor*), spotted skunk (*Spilogale putorius*), river otter (*Lontra canadensis*), black bear (*Ursus americanus*), coyote (*Canis latrans*), bullfrog (*Rana catesbeiana*) and largemouth bass (*Micropterus salmoides*) (Moyle 1973; Holland 1991a; Hayes et al. 1999). Bobcat (*Lynx rufus*), great blue heron (A*rdea herodias*), black-crowned night-heron (*Nycticorax nycticorax*), golden eagle (*Aquila chrysaetos*), red-shouldered hawk (*Buteo lineatus*), and giant garter snake (*Thamnophis gigas*) are also believed to be predators of western pond turtle (Holland 1994). Prolonged drought, contaminants, disease, and parasites also contribute to mortality in western pond turtle populations (Frye et al. 1977; Hayes et al. 1999).

Behavior

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

Western pond turtle commonly forages during late afternoon or early evening. It also basks intermittently throughout the day in order to maintain a body temperature of 75–90ºF. In general, this species typically becomes more active in water that consistently reaches 60ºF (Jennings and Hayes 1994). Extreme heat is avoided by moving to cooler areas on the bottom of pools. Western pond turtles tend to avoid water temperatures greater than 104ºF (Jennings and Hayes 1994).

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

Movement and Migratory Patterns

During spring or early summer, females move overland up to1,319 feet to find suitable sites for egg laying (Hays et al. 1999). Other long-distance movements may occur in response to drying of local water bodies or other factors. The species is capable of moving long distances (at least one mile overland) to find water; however, no mass migrations have been observed (Pilliod et al. 2013). In addition, movement patterns appeared to be independent of each other (Pilliod et al. 2013). Pilliod et al. (2-13) also found that western pond turtles make two types of movements during the winter, including short movements (less than 33 feet) within a vegetation patch and longer movements (approximately 330 feet) to new habitat patches. Genetic analysis suggests that movement of this species occurs within drainages (Spinks and Shaffer 2005).

Studies have been conducted on western pond turtles at intermittent sites and perennial sites (Bondi and Marks 2013). These studies found that turtles from intermittent sites migrated from the river substantially earlier than those from the perennial site and initiated terrestrial estivation in mid-summer, apparently in response to the declining water levels. In contrast, those turtles in the perennial sites did not migrate from the river until early fall, with the onset of declining air and water temperatures. Overall, turtles from the intermittent site spent significantly less time in the water compared to those in the perennial site. As a result, these turtles had lower body condition and were smaller, most likely due to less time available each year for aquatic foraging (Bondi and Marks 2013).

Ecological Relationships

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

Threats

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

Flow regime has a profound influence on western pond turtle movement ecology and morphology (Bondi and Marks 2013). Changes in the nature and timing of water releases from reservoirs may adversely affect downstream habitat by eliminating or altering basking sites, refugia, foraging areas, and hatchling microhabitat (Holland 1991a; USFWS 1999). The reservoirs themselves generally provide poor habitat for turtle because of the lack of emergent aquatic vegetation and basking sites, high recreational disturbance, and the presence of exotic predatory fish species. Water diversions for agriculture can also have negative impacts on turtle populations by resulting in very low or no flows for miles of stream habitat during summer months. Agricultural diversions have resulted in the elimination of pond turtle from such streams and isolation of turtle populations located in other portions of affected drainages (Holland 1991a).

Roads can create barriers to dispersal movements of western pond turtle and contribute to the isolation of populations. Contaminants from road materials, leaks, and spills could further degrade aquatic habitats used by this species. Corridors from aquatic habitat to historical and long-term nesting sites can be blocked by roads and development (Holland 1991a).

Additional threats include habitat degradation from cattle grazing; instream and streamside sand and gravel mining operations; removal of basking sites (e.g., logs, snags, and rocks) for aesthetic purposes or to facilitate recreational use; and collection of turtles for food or for the pet trade. Incidental collection of turtles, exposure to diseases from introduced exotic species, introduced predators, such as the bullfrog (*Rana catesbeiana*), will eat hatchling and young western pond turtles (Holland 1994), indiscriminate shooting, construction of highway barriers in upland nesting/migration corridors, off-road vehicle activity, boat activity, and increased exposure to contaminants are also likely to contribute to population declines in western pond turtle (Bury 1972b; Holland 1991a). Finally, extended drought and associated fire can also result in significant mortality of western pond turtle (Holland 1991a).

Context for a Regional Conservation Strategy

There are four known occurrences of western pond turtle in the Plan Area and the subspecies may be present in other locations not yet surveyed. In the state, the subspecies is primarily scattered throughout areas east and west of the Central Valley, with greatest concentrations in the San Francisco Bay area. In the Placer County region, western pond turtle has been recorded from all surrounding counties with the exception of those east of the Sierra Nevada, with concentrations greatest in Sacramento County. Given the species' broad distribution in northern California, Placer County is not of particular significance in the western pond turtle's range and distribution. However, the species' overall decline throughout the state dictates that protection of all remaining, intact habitat is important for the species' persistence. Within the Plan Area, protection or acquisition of aquatic habitats including rivers, streams, lakes, ponds, wetlands and reservoirs with emergent basking sites and upland refugia and nesting sites is of highest priority.

Modeled Species Distribution in the Plan Area

Model Assumptions

Aquatic Habitat

Western pond turtle aquatic habitat is defined by fresh emergent wetlands, seasonal wetland, riverine/riparian, and ponds.

Upland Nesting Habitat

Nesting habitat (nesting, burrowing habitat) is defined as any land cover type within 150 feet of aquatic habitat, except for urban/suburban, rural residential, agricultural types, barren, and disturbed land cover types.

Rationale

Western pond turtles are found in rivers, streams, lakes, ponds, wetlands, reservoirs, and brackish estuarine waters up to 6,500 feet above sea level (Holland 1994; Jennings and Hayes 1994). Western pond turtles use aquatic habitats primarily for foraging, thermoregulation, and avoidance of predators; they require emergent basking sites, and have been observed to avoid areas of open water lacking them (Holland 1994). Basking sites can include rocks, logs, or emergent vegetation, and are used by the turtles for thermoregulation.

Western pond turtles overwinter in both aquatic and terrestrial habitats. Terrestrial overwintering habitat consists of burrows in leaf litter or soil. Typically, western pond turtles dig nests on unshaded slopes. Nesting has been reported to occur up to 1,391 feet from water (Jennings and Hayes 1994), but is usually closer, averaging 92 feet from aquatic habitat (Rathbun et al. 2002). To remain conservative, modeled nesting habitat included a buffer of 150 feet, which should account for most possible nesting sites in the Plan Area. To account for long-distance dispersal to nest sites or movement between water bodies, the distance of 1,200 feet from all aquatic habitats was used to model movement and secondary habitat. Though this is not all inclusive of the documented 1,391 foot dispersal by Jennings and Hayes (1994), it likely still overestimate the actual upland habitat use by this species.

Model Results

Species Map 6. *Western Pond Turtle Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for western pond turtle within the Plan Area. Primary habitat is distributed throughout the Plan Area along streams. Movement habitat is found throughout the Plan Area adjacent to streams and other primary aquatic habitats. The documented occurrences of western pond turtle in the Plan Area generally correspond to modeled habitat.

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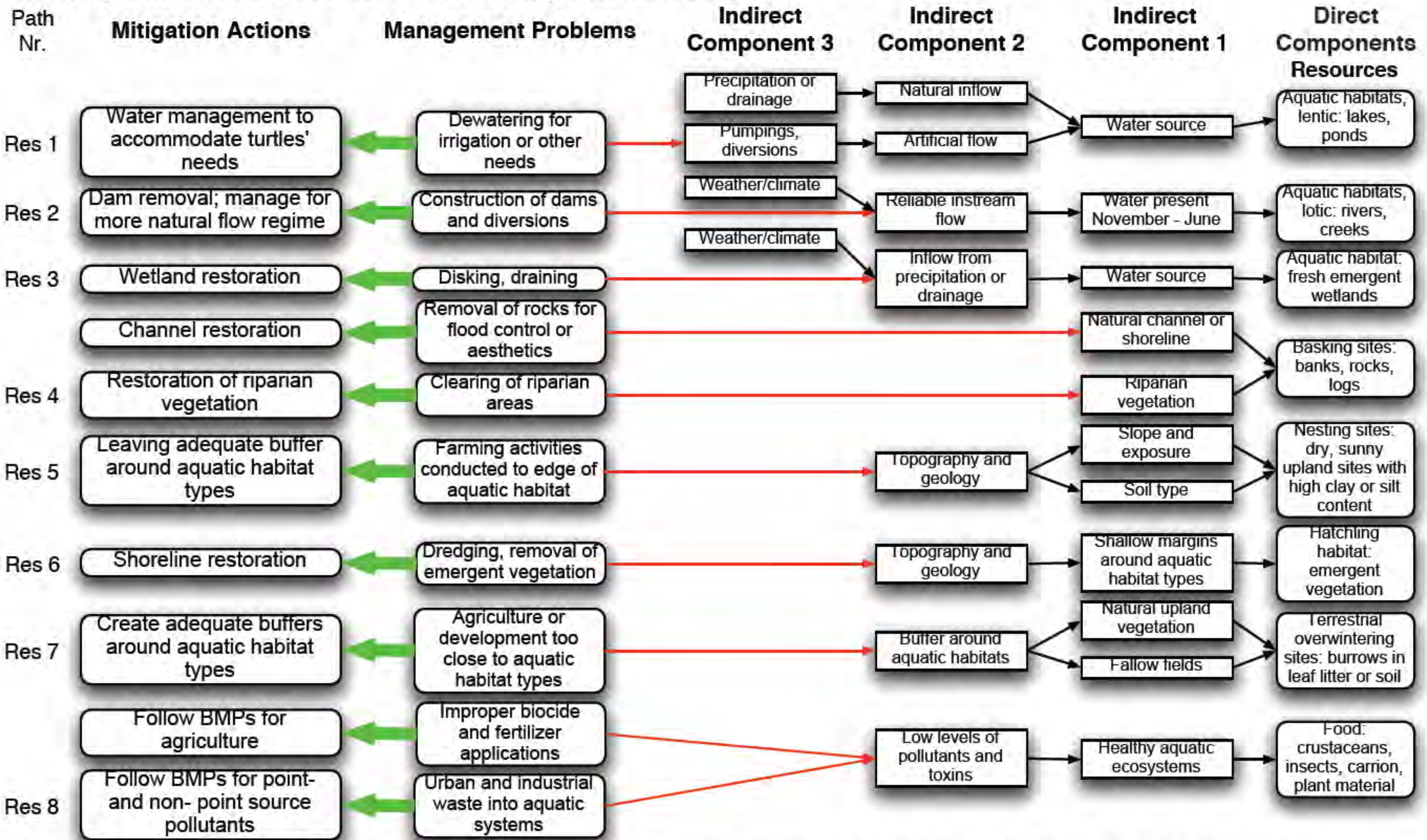
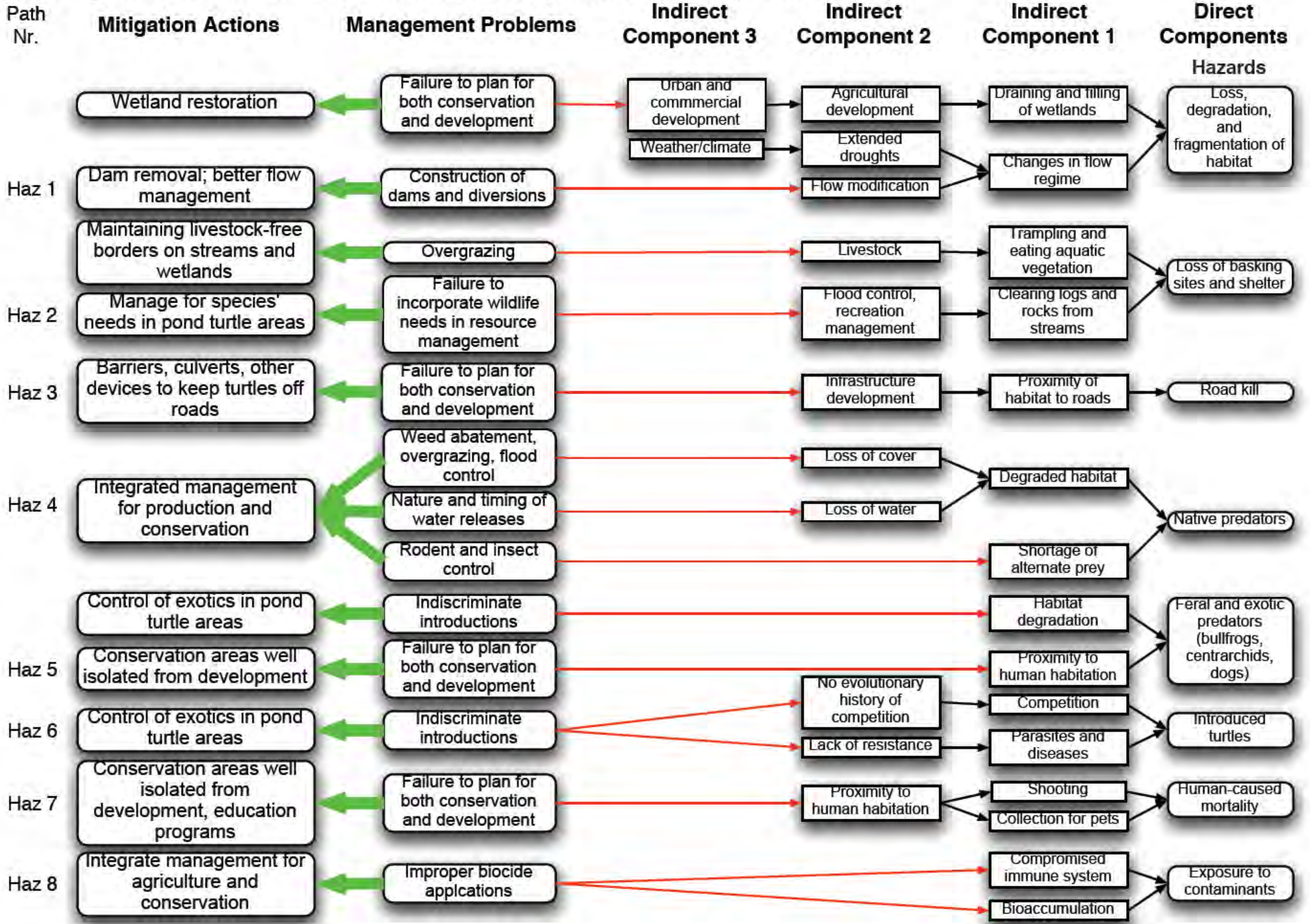
Envirogram 6 Western Pond Turtle, *Emys marmorata*

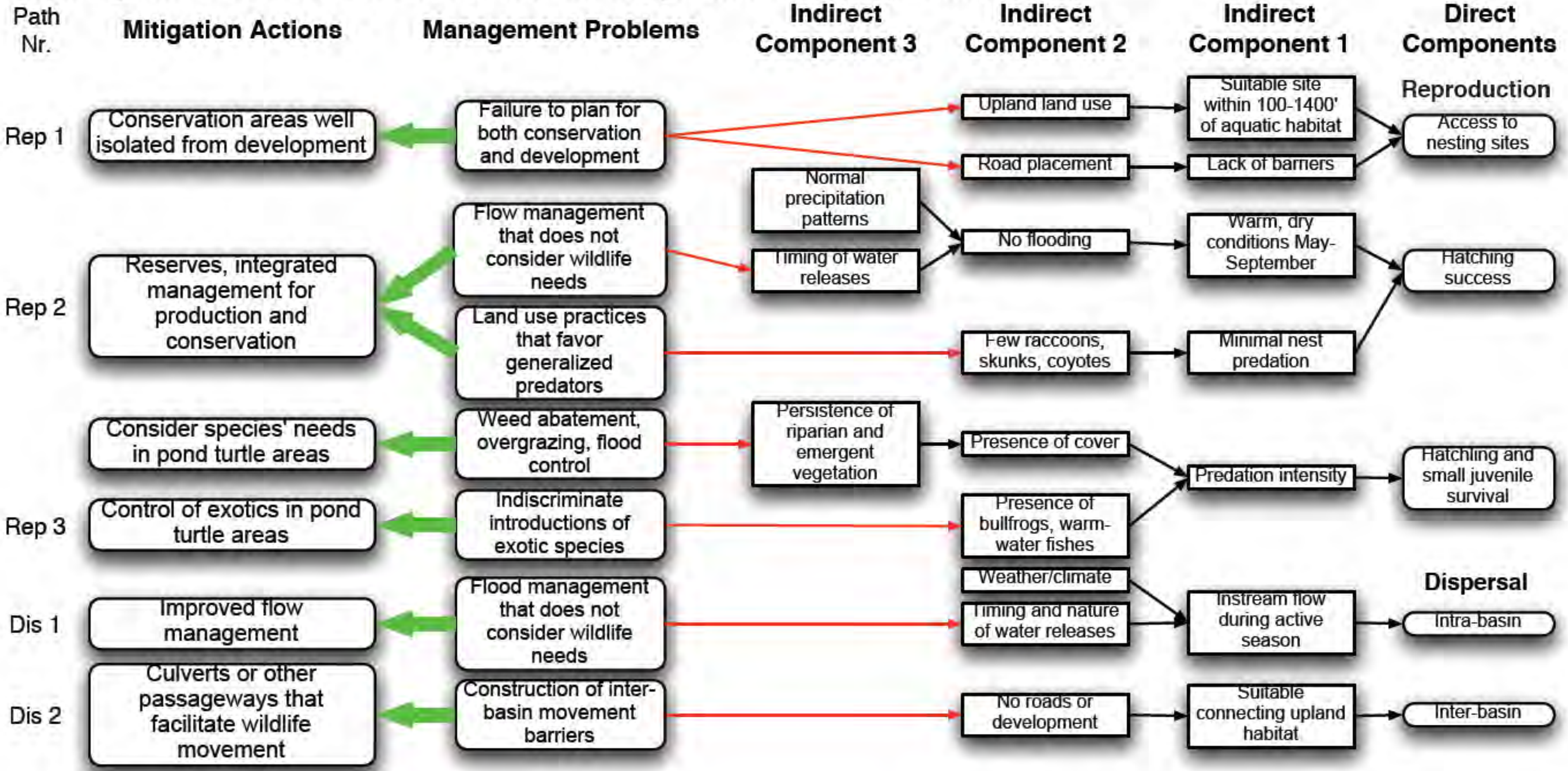


Envirogram 6 Western Pond Turtle. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram 6 Western Pond Turtle, *Clemmys marmorata* (continued 2)



Envirogram 6 Western Pond Turtle, *Clemmys marmorata* (continued 3)



Envirogram Narrative

Western Pond Turtle (*Emys marmorata*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1**:** Lentic (i.e., still water) habitats for the western pond turtle include lakes and ponds, both of which require a water source, either from natural inflow resulting from precipitation or drainage, or artificial inflow from pumping or diversions. If the artificial inflow stops because of dewatering for irrigation or other needs, the habitat may no longer exist. Managing water resources with the needs of the turtle in mind may help mitigate this problem.

Res2**:** Lotic (i.e., moving water) habitats for the western pond turtle include rivers and creeks that have water present in them between November and June. Normal precipitation patterns result in reliable instream flow during these months if the stream is not dammed or diverted. Removing these structures or managing streams for a more normal flow regime potentially can restore lost habitat for the turtle.

Res3**:** Western pond turtle also occurs in fresh emergent wetlands that result from natural precipitation or drainage. Disking and draining has destroyed most of these wetlands, but many can be restored.

Res4**:** Western pond turtle needs basking sites such as gently sloping banks, rocks, or logs. These features are found along lentic or lotic habitat types with natural channels and shorelines and riparian vegetation. The removal of rocks and logs for flood control and the clearing of riparian areas have resulted in the loss of basking sites; the restoration of natural banks and channels and restoration vegetation can restore them.

Res5**:** Western pond turtle lays its eggs in dry, sunny, upland sites with high clay or silt content. The presence of appropriate nesting sites depends on both slope and exposure and soil type, which in turn are related to local topography and geology. Farming activities that are conducted close to the edges of aquatic habitat types can result in the loss of nesting habitat. Leaving adequate buffers around aquatic habitat types can help ensure that good nesting sites are available.

Res6: Hatchling western pond turtles require emergent vegetation for cover. The presence of emergent vegetation depends upon the presence of shallow margins around northwestern pond turtle aquatic habitats, which partly depends on the microtopography of the site and partly on past management activities. Shoreline restoration can re-create appropriate bank morphology and vegetation.

Res7**:** Western pond turtle overwinters in burrows in leaf litter or soil. Appropriate sites are found in both natural upland vegetation and some fallow agricultural fields. Buffers around aquatic habitat types also help ensure the presence of good overwintering sites.

Res8**:** Western pond turtle feeds on a variety of aquatic invertebrates including crustaceans and insects, carrion, and some plant material. These items are abundant in healthy aquatic ecosystems with low levels of pollutants and toxins. Keeping agricultural, industrial, and urban pollutants out of aquatic systems by following Best Management Practices can help maintain ecosystem health.

Hazards

Haz1**:** The major hazard faced by western pond turtle is the loss, degradation, and fragmentation of its habitat. Draining and filling wetlands for agricultural development followed by poorly planned urban and commercial development is a major cause for the decline or loss of populations. Some of these wetlands may be restorable, however. Another source of habitat loss and degradation has been changes to natural flow regimes through both natural (extended droughts) and human (various types of flow modifications) causes. Removal of dams and diversions and better flow management can help ameliorate the latter.

Haz2**:** Loss of basking sites and shelter is another hazard for the western pond turtle. Trampling and eating aquatic vegetation by livestock and clearing logs and rocks from streams for flood control or recreation enhancement are largely responsible. Elimination of overgrazing by keeping livestock out of streams and wetlands and managing aquatic habitat types for wildlife as well as other uses can improve the availability of basking sites.

Haz3**:** Western pond turtle can be killed on roads when moving from aquatic to nesting or overwintering habitat. New roads built in proximity to western pond turtle habitat should be made “turtle-friendly” with barriers to discourage crossing and culverts to facilitate dispersal.

Haz4**:** Various native species prey on western pond turtle (e.g., raccoon, skunk, otter, and coyote on adults and herons and giant garter snake on young). Excessive predation usually results from degraded habitat, particularly loss of cover from weed abatement, overgrazing, or flood control, and lack of water from bad timing of water releases. Heavy predation pressure on the western pond turtle also can occur during periods when alternate prey are scarce because of rodent or insect control. These problems can be ameliorated to some extent by integrated management for production and conservation.

Haz5**:** Feral and exotic predators such as bullfrog and centrarchid fishes prey on adults and young western pond turtle. Habitat degradation and proximity to human habitation increase predation pressure from these species. Isolating turtle conservation areas from development and control of exotics can ameliorate this pressure to some extent.

Haz6**:** Introduced turtles (e.g., painted turtle, *Chrysemys picta*) compete with western pond turtles for food and other resources. Western pond turtles are especially vulnerable to such competition because it evolved in the absence of other turtles. Exotic species also are sources of parasites and diseases to which the western pond turtle apparently has little resistance. Control of exotic species in western pond turtle habitat is a necessary conservation tactic.

Haz7**:** Direct human-caused mortality such as shooting or collection for pets is another hazard for the western pond turtle. These problems are exacerbated near human habitation, suggesting that conservation areas for the western pond turtle should be well isolated from development and that education on the conservation of native species should be expanded.

Haz8**:** Exposure to contaminants, resulting in either bioaccumulation or a compromised immune system, is another hazard for the western pond turtle. Many contaminants enter aquatic ecosystems because of improper pesticide applications; these can be ameliorated by integrated management for both agriculture and conservation.

Reproduction

Rep1**:** Western pond turtle needs access to nesting sites. Suitable sites must be present with no barriers within 1,200 feet of aquatic habitats. These factors largely depend on upland land use and road placement. Conservation areas well away from development seem to be the best way to insure access to nest sites.

Rep2**:** Hatching success depends on warm, dry conditions from May to December. Such conditions require that the nest not be flooded either from irrigation or abnormal precipitation. Flow management that considers the needs of the western pond turtle is critical. Hatching success also depends on minimal nest predation; the major predators seem to be raccoon, skunk, and coyote. Integrated management for both agriculture and conservation can help ensure that favorable conditions are maintained.

Rep3**:** Reproductive success also depends on the survival of hatchlings and juveniles. Heavy predation by bullfrog and warm-water fish, exacerbated by the absence of cover provided by riparian and emergent vegetation, results in poor survival. These problems result from indiscriminate introductions of exotic species and vegetation management that does not consider wildlife needs. Control of exotics and management for factors that favor the western pond turtle can help ameliorate these problems.

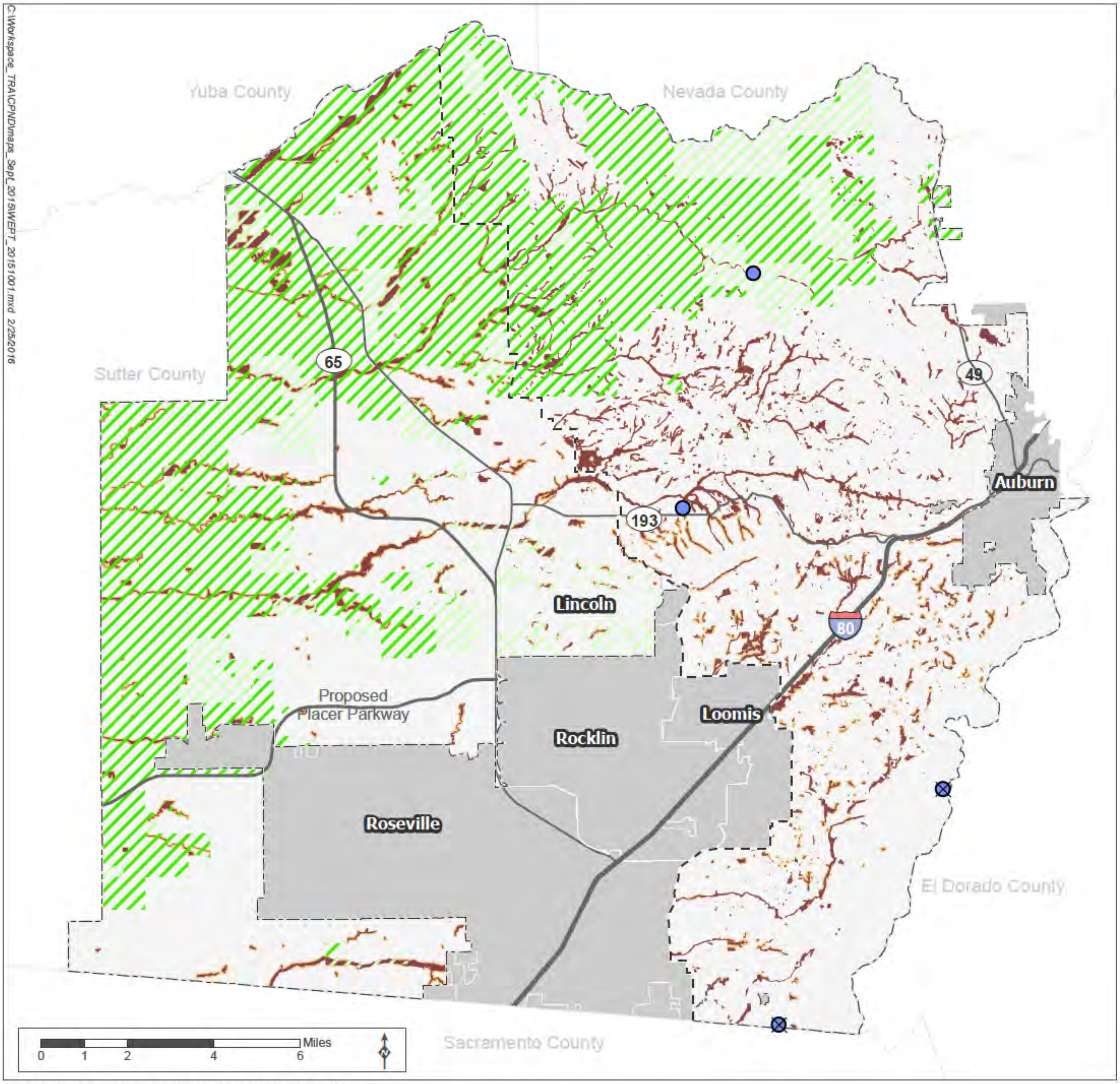
Dispersal

Dis1**:** Long-range dispersal within a watershed depends on adequate instream flow during the active season. Suitable precipitation and appropriate timing of water releases facilitates these movements.

Dis2**:** Dispersal among watersheds is rare and depends on suitable upland habitat with few roads and development. Making landscapes more permeable by removing barriers and installing devices that facilitate dispersal may allow some intra-basin movement.

Summary

Western pond turtle is a conservation challenge not only because it relies on both aquatic and terrestrial habitats but also because so much of its habitat has been lost in Placer County. Its life history strategy—many breeding attempts over a long life—is very vulnerable to increases in adult mortality rates. Because it evolved in the absence of other turtles, it also lacks evolutionary experience with introduced turtle species and their parasites and diseases. Reserve areas well isolated from intense human activity and managed primarily for the western pond turtle and compatible species seem to be the best conservation strategy.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2015; CDFG, 2005

MIG | TRA

Species Map 6.

Western Pond Turtle Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

Foothill Yellow-legged Frog
(*Rana boylii*)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Status

Federal: Under review for listing under the Endangered Species Act (USFWS 2015)

State: Species of Special Concern

Critical Habitat: Not Applicable (N/A)

Recovery Plan: N/A

Distribution

North America

Historically, foothill yellow-legged frog occurred in most Pacific coast drainages west of the Sierra/Cascade crest from the Santiam River in Marion County, Oregon to the San Gabriel drainage in Los Angeles County, California (Jennings and Hayes 1988). Records also exist for an isolated population in the Sierra San Pedro Martir in Baja California, Mexico (Loomis 1965).

California

Historically, foothill yellow-legged frog occurred from west of the crest of the Cascade Mountains in Oregon south to the Transverse Ranges in Los Angeles County, and in the Sierra Nevada foothills south to Kern County (Zweifel 1955; Stebbins 1985). An isolated population was reported in Sierra San Pedro Martir, Baja Mexico (Loomis 1965). The current range excludes coastal areas south of northern San Luis Obispo County and foothill areas south of Fresno County where the species is apparently extirpated (Jennings and Hayes 1994). Fellars (2009) found uneven distribution of foothill yellow-legged frog in California, with 30% of streams in the south coast range (south of San Francisco) and 12% of the Sierra Nevada foothills streams inhabited.

Placer County Plan Area

<u>Historical</u>

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

<u>Current</u>

Jennings and Hayes (1994) reported that foothill yellow-legged frog is widely scattered on the western slope of the northern Sierra Nevada and considered it threatened in this area. Brian Williams (pers. comm.) conducted surveys for foothill yellow-legged frogs in 2002 along tributaries of the American River and believes that foothill yellow-legged frog is likely to be widespread throughout the foothill portions of Placer County. There are twenty recent records (1998–2008) for this species in the central portion of Placer County outside of the Plan Area (California Natural Diversity Database 2009). However, there are no documented occurrences of foothill yellow-legged frog within the Plan Area or in close proximity to Auburn, Lincoln, Loomis, Newcastle, or Rocklin (Dudek 2014).

The bulk of the CNDDB records are 5 or more miles from the eastern boundary of the Plan Area. The closest recorded occurrences to the Plan Area are either on the North Fork American River or on a tributary to the river upstream of the confluence with the Middle Fork American River (Dudek 2014). The closest documented occurrence of foothill yellow-legged frog to the Plan Area is a specimen from 1952 collected at the North Fork American River confluence with the Middle Fork American River, approximately 1 mile east-northeast of the eastern boundary of the Plan Area (Dudek 2014). The closest extant occurrence of foothill yellow-legged frog to the Plan Area is located just downstream of the Clementine Reservoir, approximately 2.5 miles east of the northeastern boundary of the Plan Area. Additional occurrences are located at Dog Bar Bridge along the Bear River (just over the border in Nevada County), the Bear River upstream of Rollins Reservoir and upstream of Lake Combie, further upstream on the North Fork American River, and along the Middle Fork American River (just south of the border in El Dorado County) (California Natural Diversity Database 2015; Dudek 2014).

Dudek (2014) conducted an assessment of potentially suitable habitat for foothill yellow-legged frog in the Plan Area in August 2014. Based on this assessment, moderate to moderately high-quality breeding, larval development, and juvenile/adult habitat for foothill yellow-legged frog in the Plan Area is limited and high-quality habitat does not appear to be present. They found that, in general, vegetation encroachment and lack of suitable substrates within many stream reaches are likely the primary reason for the limited habitat within the Plan Area. Overall, the upper reaches of Coon Creek were found to provide the most suitable habitat for foothill yellow-legged frog in the Plan Area, although the portion of the Bear River within the Plan Area may also provide some potentially suitable habitat for this species. In addition, a few streams within other watersheds in the Plan Area have potentially suitable habitat for foothill yellow-legged frog, although it is generally limited in extent and isolated from other potential stream areas. Streams with isolated, short reaches of potentially suitable habitat within the Plan Area include Mormon Ravine and the upper reaches of South Fork Dry Creek.

Population Status & Trends

California

Foothill yellow-legged frog has disappeared from 54 percent of its range (Kupferberg et al. 2012). Foothill yellow-legged frog has become rare in the west slope drainages of the Sierra Nevada and southern Cascade Mountains east of the Sacramento–San Joaquin Valleys. It has not been observed since the mid-1970s at 19 historical localities on the western slope of the southern Sierra Nevada (Jennings and Hayes 1994).

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

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

Placer County Plan Area

Foothill yellow-legged frog may have occurred historically in the eastern portion of Placer County near Auburn; however, these populations were determined to be extirpated (Jennings and Hayes 1994). There are no recent records for foothill yellow-legged frogs in the western portion of the County within the Plan Area (Jennings and Hayes 1994; California Natural Diversity Database 2009; Dudek 2014).

Natural History

The habitat requirements, ecological relationships, life history, and threats to foothill yellow-legged frog described below are summarized in diagram form in the Envirogram 7 Foothill Yellow-legged Frog.

Habitat Requirements

Foothill yellow-legged frog occupies rocky streams in valley-foothill hardwood, valley foothill hardwood-conifer, valley foothill riparian, ponderosa pine, mixed conifer, coastal scrub, mixed chaparral, and wet meadow habitat types (Zeiner et al. 1988) from sea level to 6,370 feet (Jennings and Hayes 1994). It is nearly always found within a few feet of water. Foothill yellow-legged frog is frequently found in moving but not swiftly flowing water (Stebbins 1954). The species is most common along streams with rocky bottoms but has also been found along streams with mud bottoms (Stebbins 1951). Foothill yellow-legged frog requires permanent streams or, at a minimum, streams where pools persist through the dry season (Stebbins 1951). Foothill yellow-legged frog exhibits fidelity to breeding sites, using the same areas for reproductive activity annually for many years (Kupferberg 1996; Wheeler 2006). Foothill yellow-legged frogs are usually absent from habitats where introduced aquatic predators, such as sunfish (*Lepomis* spp.) and bullfrogs (*Lithobates catesbeianus*), are present (Jennings and Hayes 1994).

Reproduction

Foothill yellow-legged frog breeds from mid-March to May after the high-water stage in streams has passed and less sediment is being conveyed (Stebbins 1954). Breeding sites are typically shallow, low velocity areas close to shore (Lind et al. 1996). Foothill yellow-legged frog will use the same areas for reproduction from one year to the next (Kupferberg 1996). Eggs have been observed in early and mid-May in streams in southern California, indicating that oviposition occurs later in the south than in the north (Stebbins 1951). Eggs are deposited in clusters near margins of steams in shallow water. The egg clusters are attached to stones, vegetation, or the bank itself (Stebbins 1954.) Warm edge-water habitat is especially important for developing tadpoles. The embryos have a critical thermal maximum temperature of 26º Celsius. Tadpoles metamorphose in approximately three to four months (Storer 1925; Stebbins 1951).

Dispersal Patterns

Females tend to move greater distances than males during and following the breeding season (Wheeler et al. 2006). Little information is available regarding the distances foothill yellow-legged frog will travel. Twitty et al. (1967) observed that newly metamorphosed foothill yellow-legged frogs consistently moved upstream during fall and winter over a three-year period.

Longevity

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

Sources of Mortality

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

Behavior

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

Normal home range of foothill yellow-legged frogs is probably not more than 33 feet in the longest dimension (Zeiner et al. 1988). If it follows the pattern of other ranid frogs, males probably defend territories during the breeding season (Martof 1953; Emlen 1968).

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

Movement and Migratory Patterns

Adult foothill yellow-legged frogs are primarily diurnal and occupy small home ranges. Foothill yellow-legged frogs are highly aquatic and spend most of their life in or near streams. During periods of high water conditions, foothill yellow-legged frog may make occasional long-distance movements (up to 165 feet) (Zeiner et al. 1988). At a study site in Del Norte County, in northwestern California, females tended to move greater distances (from breeding to non-breeding sites) than males, and that males tended to remain at their breeding sites following reproductive activity (Wheeler et al. 2006). In contrast, Van Wagner (1996, as cited in Wheeler 2006) found that males and females moved similar distances.

Ecological Relationships

Bullfrog is known to prey upon foothill yellow-legged frog (see *Population Threats* below). Several species of garter snakes, including red-sided garter snake (*Thamnophis sirtalis parietalis*), western terrestrial garter snake (*T. elegans*), and Oregon garter snake (*T. couchii hydrophilus*), are predators of post-hatching stages of foothill yellow-legged frogs (Zweifel 1955; Jennings and Hayes 1994). Oregon garter snakes have been observed to feed more frequently on tadpoles, whereas the other two species of garter snakes have been observed to feed more frequently on post-metamorphic individuals

(Jennings and Hayes 1994). Rough-skinned newt (*Taricha granulosa*) has been recorded preying on foothill yellow-legged frog eggs (Evenden 1948). In addition, when Centrarchid fishes were offered *Rana* tadpoles and eggs, they ate them readily (Werschkul and Christensen 1977). Fish, mammals (e.g., raccoons), and birds are likely to prey on one or more stages of foothill yellow-legged frog (Zweifel 1955). Foothill yellow-legged frog coexists with Cascades frog (*Rana cascadae*) and red-legged frog (*Rana aurora*) at some localities; however, different microhabitat preferences may limit competition (Zeiner et al. 1988).

Threats

Habitat Alteration and Degradation

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

Drought, Flooding, and Water Management

Scouring floods that occur approximately every 500 years have been implicated in declines of foothill yellow-legged frog in southern California (Hayes and Jennings 1986). Poor timing and high-flow releases of water from upstream reservoirs can have the same effect by scouring eggs from their oviposition substrates (Kupferberg 1996; Lind et al. 1996). Eggs are washed away when streams experience high-flow velocities for several days in a row (Kupferberg 1996; Bondi et al. 2013). The magnitude and timing of spring pulse flows can affect survival of foothill yellow-legged frog embryos. Large magnitude pulses decrease egg survival and smaller magnitude pulses late in the breeding season can cause higher mortality because egg jelly adhesion and cohesion may be diminished (Kupferberg 2008). Decreased water flows can also have a negative result by stranding eggs or forcing foothill yellow-legged frog into permanent pools where it may be more susceptible to predation (Moyle 1973; Kupferberg 1996). Results of surveys in Oregon by Borisenko and Hayes (1999) suggest a direct correlation between the absence of frogs and the presence of dams and grazing activities. Large dams substantially alter habitat by changing the hydrology and geomorphology of the water system, resulting in degraded habitat for foothill yellow-legged frog (Borisenko and Hayes 1999). Presence of dams in the upstream watershed is associated with an absence of foothill yellow-legged frogs, suggesting that flow alteration is associated with lower abundance of this species (Kupferberg et al. 2012).

Introduction of Nonnative Predators

One of the primary factors in decline of the foothill yellow-legged frog in the Sierra Nevada is the introduction of nonnative predators (Jennings 1996). At least 60 species of fish, many of them predatory, have been introduced in western North America over the past 120 years (Jennings 1988). Fish are known to prey upon various life stages of anurans (Hayes and Jennings 1986) and have been implicated in the decline of populations of frogs in some areas (Cory 1963; Knapp and Mathews 2000). Locations where introduced fishes were abundant contained few foothill yellow-legged frogs, indicating that the presence of these fish had a negative influence on the abundance of the frog (Moyle 1973; Hayes and Jennings 1986; Jennings and Hayes 1994).

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

Pesticides and Herbicides

Pesticides, herbicides, and other toxins are known to be harmful to various life stages of ranid frogs (Hayes and Jennings 1986). Research has demonstrated that amphibians absorb pesticides in aquatic and terrestrial systems; the toxins disrupt their nervous systems and cause death by respiratory failure (Yosemite Association 2001). Pesticide drift has recently been implicated as a potential cause of declining populations of four ranid species in California, including foothill yellow-legged frog. Davidson et al. (2002) compared the spatial pattern of declines for eight species of California amphibians with the amount of upwind agricultural land use. Predominant winds flow from the coast through the Central Valley and into the foothills and mountains of the Sierra Nevada. Declines of four ranid species were strongly associated with upwind agricultural use, suggesting that this group of amphibians may have a particular sensitivity to agrochemicals.

Pathogens

Recent attention has been given to the role pathogens may play on localized population fluctuations of amphibians. The chytrid fungus has been found in several species of amphibians in California (U. S. Fish and Wildlife Service 2000; Vredenburg 2001). Chytrid fungus causes deterioration of the mouthparts of tadpoles and has been implicated in amphibian declines in North America (Oulett et al. 2005), Australia, Central and South America, and Europe (Vredenburg 2001). The fungus has been found in a number of amphibians whose populations are declining, and current studies indicate that the chytrid fungus may be playing a role in that decline (Adams et al 2017).

Context for a Regional Conservation Strategy

There are no current records of foothill yellow-legged frog in western Placer County, although the species has been recently recorded in Placer County and is assumed to be present or potentially present within the foothill region of the Plan Area. Moderate to moderately high-quality breeding, larval development, and juvenile/adult habitat for foothill yellow-legged frog is present (although limited) in the Plan Area and high-quality habitat does not appear to be present, including in the upper reaches of Coon Creek, portions of the Bear River, Mormon Ravine, and the upper reaches of South Fork Dry Creek (Dudek 2014).

In California, the range of recorded yellow-legged frog extends along western California from Del Norte County south to San Luis Obispo County, and east of the Central Valley from Siskiyou County to Tulare County. The species has become rare on the western slopes of the Sierra Nevada, and although there are records from counties north and south of Placer (including Nevada, Sierra and Plumas counties to the north and El Dorado and Amador counties to the south) the significant reduction of yellow-legged frog in the region heightens the value of existing populations and suitable habitat. Human-induced changes such as introduced non-native predators, pesticide run-off, hydrologic changes resulting from dams, and urbanization are the leading causes of population declines. Within the Plan Area, suitable habitat where these threats are minimized is of greatest conservation or acquisition priority.

Modeled Species Distribution in the Plan Area

Model Assumptions

Year-round Habitat

Modeled year-round habitat for foothill yellow-legged frog is defined by riverine land-cover above 500 feet in elevation.

Rationale

Foothill yellow-legged frog occurs from sea level to 6,370 feet (Jennings and hayes 1994); however, the CDFW has modeled the foothill yellow-legged frog habitat in California and restricts it to above 500 feet in elevation in the Plan Area (Hooper pers. comm.). Foothill yellow-legged frogs occupy, and are nearly always found within a few feet of, rocky streams that run through oak woodlands. Foothill yellow-legged frogs are frequently found in moving but not swiftly flowing water (Stebbins 1954). Foothill yellow-legged frogs require permanent streams or, at a minimum, streams where pools persist through the dry season (Stebbins 1951).

Model Results

Species Map 7. *Foothill Yellow-legged Frog Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for foothill-yellow legged frog within the Plan Area. Modeled potential habitat is limited primarily to the western portion of the Bear River, Coon Creek and upper tributaries, Auburn Ravine, Pleasant Grove Creek, and Dry Creek and its upper tributaries.

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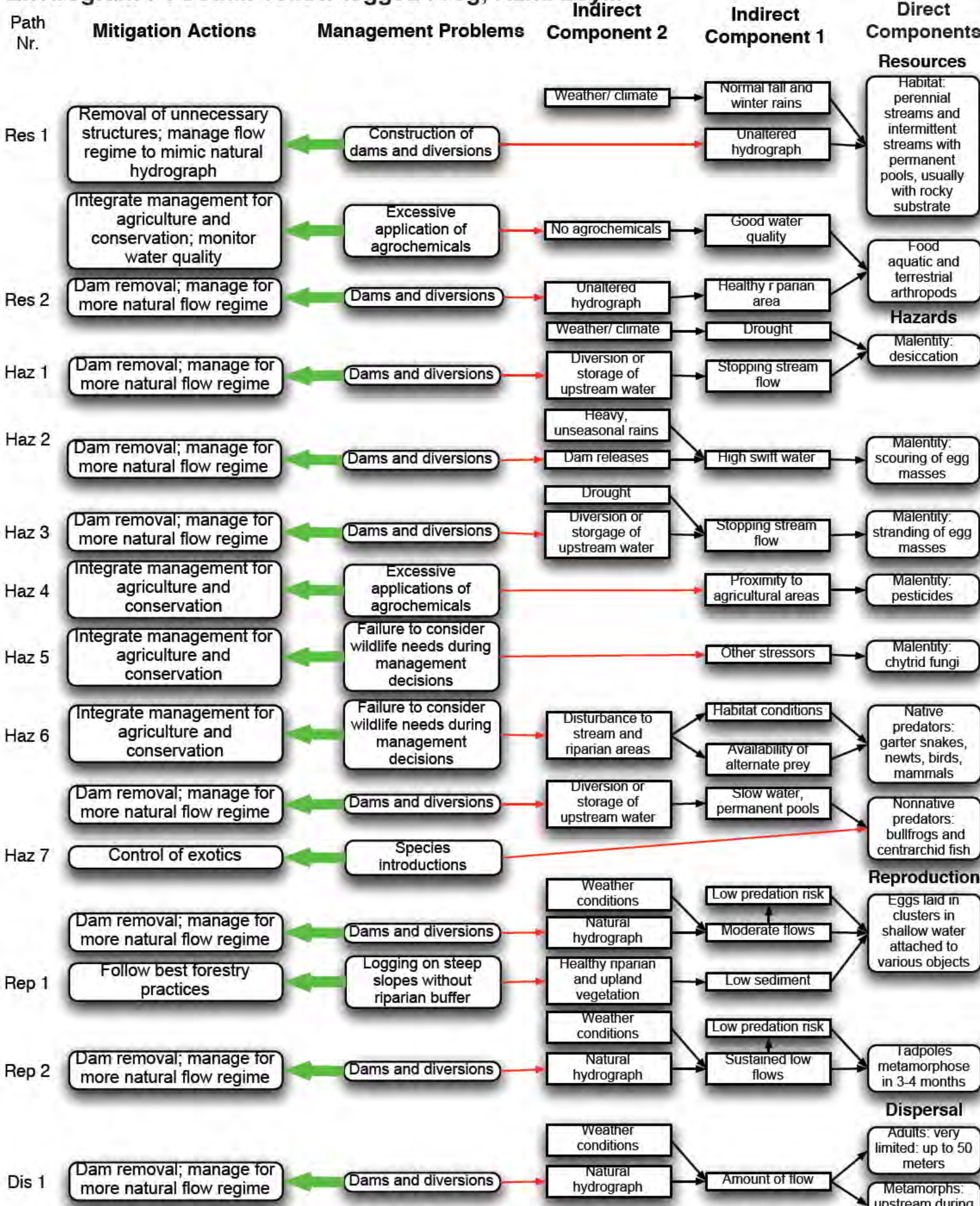
Envirogram 7 Foothill Yellow-legged Frog, *Rana boylii*



Envirogram 7 Foothill Yellow-legged Frog. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram Narrative

Foothill Yellow-legged Frog (*Rana boylii*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Foothill yellow-legged frog inhabits both perennial streams and intermittent streams with permanent pools; preferred streams usually have rocky substrates. Suitable conditions for the species depend on adequate rainfall and a natural hydrograph. The construction of dams and diversions has made many streams unsuitable, but the removal of unnecessary structures and managing the flow regime to mimic historic conditions can mitigate this problem to some extent.

Res2: Foothill yellow-legged frog preys on both aquatic and terrestrial arthropods. Aquatic arthropods require good water quality, and agrochemicals entering streams because of excessive application are a major threat to water quality. Best management practices that integrate agriculture and conservation can protect water quality; water quality monitoring is necessary to ensure that these practices are being followed. A healthy riparian zone resulting from an unaltered hydrograph is favorable for terrestrial arthropods; the removal of unnecessary structures and managing the flow regime to mimic historic conditions encourages good riparian conditions.

Hazards

Haz1: Larval and adult foothill yellow-legged frogs can desiccate when their streams dry up from drought or blocked stream flow. The removal of unnecessary structures and managing the flow regime to mimic historic conditions can help mitigate this problem.

Haz2: High, swift water resulting from unseasonal heavy rains or dam releases can scour egg masses from their attachments. Mitigation is the same as for Haz1.

Haz3: Egg masses can be stranded in pools that make them susceptible to predation and high temperatures. Stranding results from drought or blocked stream flow. Mitigation is the same as for Haz1.

Haz4: Pesticide poisoning has been implicated in amphibian mortality; this is most likely to happen in streams that flow through agricultural areas where agrochemicals are applied to excess. Best management practices that integrate agriculture and conservation can help mitigate this problem.

Haz5: Chytrid fungi also have been implicated in amphibian mortality. Infections usually result when the amphibians are stressed from other factors; multiple stressors can be alleviated by management practices that integrate agriculture and conservation.

Haz6: A number of native predators eat all life stages of foothill yellow-legged frog. Predation pressure is minimized when habitat conditions are favorable to foothill yellow-legged frog (see Res1) and alternate prey items are available (see Res2). These conditions require management practices that integrate agriculture and conservation.

Haz7: Non-native predators, particularly bullfrogs and centrarchid fishes (e.g. bass, sunfish), are very effective predators on foothill yellow-legged frog, particularly larvae. These species thrive in altered flow regimes and are often introduced into farm ponds or dammed streams for food or sport fishing. Removal of unnecessary structures and managing the flow regime to mimic historic conditions provides conditions that are less favorable to exotic species, and introductions of these species should not occur within or near conservation areas. Control of bullfrogs and other non-native predators also may be necessary.

Reproduction

Rep1: Eggs are laid in clusters in shallow water and are attached to various objects to keep them from floating away. Anything that increases sedimentation or changes the flow regime such as dams and diversions, drought or deluge, or logging too close to streams can make conditions unsuitable for eggs and may increase predation risk (see Haz6 and Haz7). Removal of unnecessary structures, managing the flow regime to mimic historic conditions, and following best forestry practices mitigate such conditions.

Rep2: Tadpoles require three to four months of low flow to develop. Anything that changes the flow regime such as dams and diversions or drought or deluge can make conditions unsuitable for tadpoles and may increase predation risk (see Haz6 and Haz7). Removal of unnecessary structures and managing the flow regime to mimic historic conditions can help mitigate these problems.

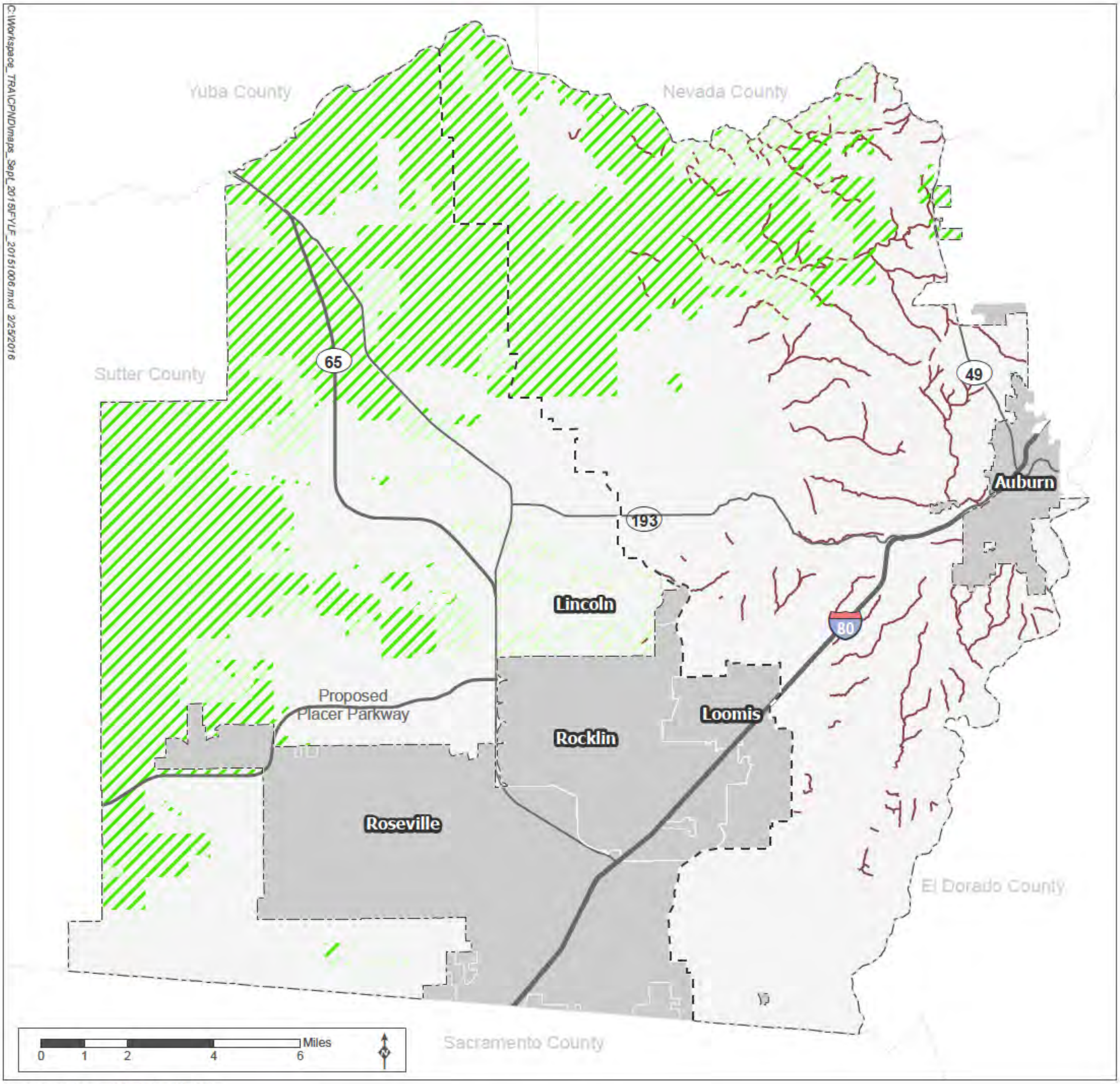
Dispersal

Dis1: Foothill yellow-legged frog is apparently very sedentary, with adults and metamorphs moving only very short distances (150 feet). Dispersal is facilitated by natural flow conditions, which in turn depends on an unaltered hydrograph. Removal of unnecessary structures and managing the flow regime to mimic historic conditions can help facilitate dispersal.

Summary

Almost all of the foothill yellow-legged frog's life history is tied to pre-settlement conditions—high water in the spring following winter rains, low flows in the summer and fall, and healthy riparian vegetation.

Deviations from these conditions brought about by human activities result in increased rats of predation, reproductive failure, and food shortages. Dams and diversions, destruction of riparian vegetation, and introduced predators all have reduced populations of this species considerably. Reversals of these problems are necessary for this species to recover.



Source: Placer County, 2014; MIG | TRA, 2015

Occurrences	Modeled Habitat		
None	Year-round Habitat	Existing Protected Area	Major Road
	Non-habitat	Reserve Aquisition Area	Valley/Foothill Divide
		Non-participating City	Area A Boundary

MIG TRA

Species Map 7

Foothill Yellow-legged Frog Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

California Red-legged Frog (*Rana draytonii*)

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Status

Federal: Threatened.

Critical Habitat: Critical habitat designated on April 12, 2001 (USFWS 1996); Critical habitat

designation revised on March 17, 2010 (USFWS 2010).

State: Species of Special Concern

Other: None

Recovery Plan: Recovery Plan for the California red-legged Frog (USFWS 2002)

Distribution

California

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

The species' current coastal distribution extends from Sonoma to Los Angeles counties; it also occurs in isolated locations in the Sierra Nevada/Cascade, the northern Coast, and the northern Transverse Ranges. It is relatively common in the San Francisco Bay area. California red-legged frog is believed to be extirpated from the floor of the Central Valley (USFWS 2002). Two populations were recently discovered in the southern Transverse and Peninsular Ranges, where the species was believed to be extirpated (USFWS 2001).

Placer County Plan Area

<u>Historical</u>

Historically, the California red-legged frog occupied sections of the western slope of the Sierra Nevada from Shasta to Tulare counties. The species occurred in at least 30 foothill drainages bordering the Central Valley; however, no specific location information is available for Placer County (USFWS 2002). Museum specimens and photographs obtained for the Sierra Nevada represented 21 localities where California red-legged frog occurred that extended from French Creek in east central Butte County southeast to O'Neals in Madera County (Barry and Fellers 2013). There are three historical records for California red-legged frog in Placer County (Barry and Fellers 2013; Jennings and Hayes 1994; California Natural Diversity Database 2015; Williams pers. comm.). One of these occurrences from 1946 was reportedly near the Placer County Superior Courthouse in Auburn within the Plan Area (Barry and Fellers 2013). One other occurrence from 1916 was located near Michigan Bluff within the Plan Area (Barry and

Fellers 2013; California Natural Diversity Database 2009). The other occurrence was from 1939 and was located near Dutch Flat (Barry and Fellers 2013), 15–20 miles outside of the Plan Area.

Current

Most of the known California red-legged frog populations on the western slope of the Sierra Nevada foothills have been eliminated or fragmented, with only ten occurrences discovered since 1991, including one extant historical population, six new populations, and three new single specimen occurrences (Barry and Fellers 2013). These recent occurrences extend from Jack Creek in east central Butte County southeast to Cuneo Creek in Mariposa County (Barry and Fellers 2013). Of these occurrences, one large population of California red-legged frog is present within the Plan Area near Michigan Bluff at Big Gun Diggings. This population is the only historical population of California red-legged frog in the Sierra Nevada known to be extant (Barry and Fellers 2013). Previously undesignated as critical habitat (USFWS 2001), 1,243 acres of the Michigan Bluff area are currently designated as critical habitat unit PLA-1 (USFWS 2010). Big Gun Diggings, now called the Big Gun Conservation Bank, was acquired by Westervelt Ecological Services in 2007 and is now privately held as a California red-legged frog habitat mitigation bank (Barry and Fellers 2013; Westervelt Ecological Services 2012). The site includes six mine tailing ponds situated on a bluff northeast of the Middle Fork of the American River.

Population Status and Trends

California

California red-legged frog has sustained a 70% reduction in its geographic range as a result of several factors acting singly or in combination (Jennings et al. 1992). Before 1960, California red-legged frog populations were densely distributed throughout the California Coast Range and the species was widespread in the coastal southern California foothills (Jennings and Hayes 1994). By 1970, all but a handful of populations were extirpated from southern California and local declines and extirpations occurred from Monterey to Ventura counties (Jennings and Hayes 1994; USFWS 2002). Habitat loss and alteration, over-exploitation, and the introduction of exotic predators were significant factors in the species' decline in the early- to mid-1900s. Reservoir construction, expansion of introduced predators, grazing, and prolonged drought fragmented and eliminated many of the Sierra Nevada foothill populations (USFWS 1996); however, studies conducted since 1990 in the Sierra Nevada foothills have identified new populations and also documented surviving populations that were previously reported as extirpated (Barry and Fellers 2013; Jennings and Hayes 1994; USFWS 2002). The Sierra Nevada distribution of California red-legged frog seems to have declined very little, if at all, since the 1960's (Barry and Fellers 2013). California red-legged frog still occurs in Baja California (USFWS 2001); however, information on the species' status in that portion of the range could not be located. California red-legged frog remains relatively widespread in the coastal mountains north of Point Conception, but it is only common in the San Francisco Bay area (USFWS 2002).

Placer County Plan Area

Population status and trends in Placer County are difficult to determine because there is little information available on locations of California red-legged frog. There are only three historical records for California red-legged frog in Placer County. Two of these are located within the Plan Area, including one near Placer County Superior Courthouse in Auburn and one near Michigan Bluff (Barry and Fellers

2013). Of these occurrences, only one historical occurrence within the Plan Area is currently extant. This population is a large population of California red-legged frog near Michigan Bluff at the Big Gun Conservation Bank (Barry and Fellers 2013). Only one additional location near Ralston Ridge in Placer County was found to have California red-legged frog; however, this occurrence was located at least 10 miles from the Plan Area.

Natural History

The habitat requirements, ecological relationships, life history, and threats to California red-legged frog described below are summarized in diagram form in the Envirogram 8 California Red-legged Frog.

Habitat Requirements

California red-legged frog has been found at elevations from sea level to about 5,000 feet. It uses a variety of habitat types including various aquatic, riparian, and upland habitats (USFWS 2002). California red-legged frog can use many aquatic systems, provided a permanent water source, ideally free of nonnative predators, is nearby (USFWS 2001). However, individual frogs may complete their entire life cycle in a pond or other aquatic site that is suitable for all life stages (USFWS 2001). California red-legged frog breeds in aquatic habitats such as marshes, ponds, deep pools and backwaters in streams and creeks, lagoons, and estuaries. Breeding adults are often associated with dense, shrubby riparian or emergent vegetation and areas with deep (>27 inches) still or slow-moving water (USFWS 2001; 2002). However, this subspecies often successfully breeds in artificial ponds with little or no emergent vegetation and has been observed in stream reaches that are not covered in riparian vegetation. An important factor influencing the suitability of aquatic breeding sites is the general lack of introduced aquatic predators (USFWS 2002).

California red-legged frog spends a substantial amount of time resting and feeding in riparian and emergent vegetation. The moisture and camouflage provided by the riparian plant community may provide good foraging habitat and may facilitate dispersal in addition to providing pools and backwater aquatic areas for breeding. Dispersal sites typically provide forage or cover opportunities and include boulders or rocks and organic debris such as downed trees or logs; industrial debris; and agricultural features such as drains, watering troughs, spring boxes, and abandoned sheds (USFWS 2001). California red-legged frog also uses small mammal burrows and moist leaf litter (Jennings and Hayes 1994). Incised stream channels with portions narrower and deeper than 18 inches may also provide habitat (USFWS 1996). Use of this habitat type by California red-legged frog is most likely dependent on year-to-year variations in climate and habitat suitability and varying requisites per life stage (USFWS 2001).

During summer, California red-legged frog generally remains in or near water. If water is not available it often disperses from the breeding habitat to forage and seek summer habitat (USFWS 2002). This habitat may include shelter under boulders, rocks, logs, industrial debris, agricultural drains, watering troughs, abandoned sheds, or hay-ricks. California red-legged frog will also use small mammal burrows and moist leaf litter and incised streamed channels (Jennings and Hayes 1994; USFWS 1996, 2002). This summer movement behavior, however, has not been observed in all California red-legged frog populations studied.

Reproduction

California red-legged frog breeds from November through March, although earlier breeding has been recorded in southern localities (Storer 1925). Males have paired vocal sacs and call in air (Hayes and Krempels 1986). Males appear at breeding sites two to four weeks before females (Storer 1925). Female California red-legged frog deposit egg masses on emergent vegetation so that the masses float on the surface of the water (Hayes and Miyamoto 1984). Egg masses contain about 2,000–5,000 moderate-sized (0.08–0.11 inches in diameter), dark reddish brown eggs (Storer 1925; Jennings and Hayes 1985). Eggs hatch in 6–14 days (Storer 1925). Larvae generally undergo metamorphosis 3.5–7 months after hatching (Storer 1925; Wright and Wright 1949; Jennings et al. 1992); however, California red-legged frog tadpoles have recently been observed to overwinter in some areas (Fellers et al. 2001). Survival from hatching to metamorphosis has been estimated as ranging from less than 1% (Jennings et al. 1992) to 1.9% (Cook 1997). In one pond study fewer than 5% of California red-legged frog larvae reached metamorphosis when bullfrog (*Lithobates catesbeiana*) tadpoles were placed in the ponds with them, whereas 30–40% reached metamorphosis in ponds that did not have bullfrog tadpoles (Lawler et al. 1999). Males attain sexual maturity by two years and females by three years of age (Jennings and Hayes 1985).

Dispersal Patterns

California red-legged frog often disperses from its breeding habitat to utilize various aquatic, riparian, and upland aestivation habitats in the summer; however it is also common for individuals to remain in the breeding area on a year-round basis (USFWS 2001). Bulger et al. (2003) postulated that the observed under-representation of subadults, or postmetamorphs (males < two years, females < three years), at breeding locations signifies a largely terrestrial existence, and that this age class likely contributes significantly more to regional metapopulation persistence than adults. Other ranid species such as the northern leopard frog (*Rana pipiens*) and the wood frog (*Rana sylvatica*) have been found to disperse radially from their natal site and travel upwards of three miles in the two to three years between metamorphosis and first breeding (Dole 1965; Berven and Grudzien 1990). Bulger et al. (2003) found that 11 – 22% of a Santa Cruz County coastal population of California red-legged frog migrated. Fellers and Kleeman (2007) found 66% of studied females and 25% of studied males migrated. Both studies found migration events to be almost exclusively between a breeding location and the nearest non-breeding habitat, and that frogs mostly traveled between these habitats using straight-line trajectories, regardless of habitat type (Bulger et al. 2003; Fellers and Kleeman 2007). Bulger et al. (2003) documented three frogs traveling over 9,240 feet between the same breeding and non-breeding location when the closest breeding location was only 1,980 feet away.

Longevity

In a long term tagging study conducted at Waddell Creek and Lagoon, the oldest frogs recorded were two males at 11 and 12 years (Smith pers. comm.). Females were captured far less often; the oldest female at the Waddell site was found to be 8 years (Smith pers. comm.)

Sources of Mortality

California red-legged frog is eaten by native predators such as raccoon (*Procyon lotor*), great blue heron (*Ardea herodias*), and garter snake (*Thamnophis* sp.), as well as by nonnative predators such as bullfrog (USFWS 2002). Hayes and Jennings (1986) suggested a variety of fish species may feed on California red-legged frog and foothill yellow-legged frog (*Rana boylii*). Smallmouth bass (*Micropterus dolomieui*) are known to eat larval and postmetamorphic red-legged frog (Kiesecker and Blaustein 1998). Calef (1973) looked extensively into predation on red-legged frog and found that *Odonata* (dragonfly and damselfly) larvae and giant diving beetles (*Lethocerus americanus*) prey on red-legged frog eggs. He also found that larvae were regularly preyed upon by salamanders and newts. Fungal infection or localized desiccation is reported as sources of egg mortality (Calef 1973).

Behavior

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

California red-legged frog has a varied diet that includes both invertebrates and vertebrates (USFWS 2002). The tadpoles are believed to eat algae; however, this has not been studied (Jennings et al. 1992). In general, amphibian larvae feed on bacteria, protozoans, free-floating algae, and other small particles suspended in water (Stebbins and Cohen 1995). Invertebrates were found to be the most common food item consumed by adult frogs. However, larger California red-legged frogs will eat vertebrates, such as Pacific tree frog (*Hyla regilla*) and California mice (*Peromyscus californicus*) (Hayes and Tennant 1985). Vertebrates represented more than half the prey mass eaten by larger frogs examined by Hayes and Tennant (1985). Feeding activity probably occurs along the shoreline and on the surface of the water as evidenced by the types of foods eaten (Hayes and Tennant 1985).

Movement and Migratory Patterns

Although percentages vary by location, if perennial breeding habitat is present, some portion of any given red-legged frog population will likely be migratory (Bulger et al. 2003; Fellars and Kleeman 2007). Although most adult migration occurs during the rainy months (November – April), California red-legged frog is known to move throughout the year, often times despite the persistence of a breeding site (Bulger et al. 2003; Fellars and Kleeman 2007). Adult movement during the dry season (May – October) is generally associated with the drying up of seasonal breeding ponds (Fellars and Kleeman 2007), although adult frogs have been found to persist in the desiccation cracks of dried ponds (Cook 2004).

During the wet season, dispersing California red-legged frog can travel long distances (over 2 miles) over land, mostly in point-to-point, straight-line trajectories, between breeding and non-breeding locations, through a diversity of pristine and modified habitats including pastureland, fallow and planted agricultural land, forestland, fields, and grasslands (Bulger et al. 2003; Fellars and Kleeman 2007). Non-migratory frogs generally stay year-round at a breeding location and rarely travel more than 100 feet from water (Bulger et al. 2003; Fellars and Kleeman 2007). During dry periods, California red-legged frog generally remains in or near water (USFWS 2002).

Ecological Relationships

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

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

Threats

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

Habitat Loss, Degradation, and Fragmentation

The primary factors that have led to declining populations of California red-legged frog are loss, degradation, and fragmentation of habitat. Urbanization, conversion of land to agriculture, overgrazing, and timber harvesting are some of the activities that have resulted in habitat loss, degradation, and fragmentation. The rate of urbanization in California is considered a significant threat to California red-legged frog. Urbanization often isolates or fragments existing habitat. In addition, adjacent habitat may become more susceptible to exotic invasions. Drainages in the area of developments often experience changes in hydroperiod due to water diversions and/or wastewater effluent. Conversion of intermittent drainages to perennial ones may create suitable conditions for nonnative predators (USFWS 2002). Land use changes such as conversion of natural lands to agriculture, livestock grazing in riparian areas, timber harvesting, and historic placer mining not only changed or eliminated existing habitat but contributed significantly to the siltation of aquatic habitats (Jennings et al. 1992; Jennings 1996; Fisher and Shaffer 1996; USFWS 2002).

Introduced Predators

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

In addition to predation, bullfrog may also have a competitive advantage over California red-legged frog (Hayes and Jennings 1986). Bullfrog is larger, has more generalized food habits, and has an extended breeding season; moreover, bullfrog eggs and larvae are unpalatable to predatory fish (Walters 1975). Bullfrog also can interfere with red-legged frog reproduction; both California and northern red-legged frog have been observed in amplexus with both male and female bullfrog (USFWS 2002).

Water Management

Hydrologic alteration has been associated with decreases in the distribution and abundance of California red-legged frog (Kupferberg et al. 2012). Rapid changes in flow can dislodge or strand clutches of eggs and are associated with low survival of clutches to hatching (Kupferberg et al. 2012). In contrast, water diversions and impoundments have altered habitat and made it less suitable for many ranid species (Jennings 1996). Water diversions change the hydrology of drainage systems; these changes can isolate populations and restrict the use of or eliminate dispersal corridors. Water impoundments have a similar effect where they create a barrier to dispersal (Jennings 1996; USFWS 2002). Additionally, reservoirs provide suitable habitat for bullfrog and are typically stocked with nonnative fish. These nonnative species frequently disperse upstream and downstream of reservoirs into California red-legged frog habitat (USFWS 2002).

Pesticides

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

Pathogens

Recent attention has been given to the role pathogens may play on localized population fluctuations of amphibians. The chytrid fungus has been found in several species of amphibians in California (USFWS 2002; Vredenburg 2001). Chytrid fungus causes deterioration of the mouthparts of tadpoles and has been implicated in amphibian declines in Australia, Central and South America, and Europe (Vrendenburg 2001). The fungus has been found in a number of amphibians whose populations are declining and current studies indicate that the chytrid fungus may be playing a role in that decline (Adams et al 2017).

Context for a Regional Conservation Strategy

There are only three historical records for California red-legged frog in Placer County. Two of these are located within the Plan Area, including one near Placer County Superior Courthouse in Auburn and one near Michigan Bluff (Barry and Fellers 2013). Of these occurrences, only one historical occurrence within the Plan Area is currently extant. This population is a large population of California red-legged frog near Michigan Bluff at Big Gun Diggings (Barry and Fellers 2013).

Previously undesignated as critical habitat (USFWS 2001), 1,243 acres of the Michigan Bluff area are currently designated as critical habitat unit PLA-1 (USFWS 2010) within the Plan Area. In addition, Big Gun Diggings, which is located within the Plan Area, was acquired by Westervelt Ecological Services in 2007 and is now privately held as California red-legged frog habitat mitigation bank (Barry and Fellers

2013; Westervelt Ecological Services 2012). The site includes six mine tailing ponds situated on a bluff northeast of the Middle Fork of the American River.

Much of Placer County land that is privately owned has not been surveyed and may support unidentified populations of California red-legged frog. Where suitable habitat persists, California red-legged frog has some potential for occurrence in the Plan Area. If additional extant populations of California red-legged frog were discovered in the Plan Area, protection of associated habitat would be critical due to the species' rarity in the County and in the region. In general, maintenance of suitable aquatic habitats with adjacent upland areas are of highest priority for conservation of California red-legged frog, and ponds that provide potential breeding habitat should be protected when feasible within the Reserve System.

Modeled Species Distribution in the Plan Area

Model Assumptions

Aquatic Habitat (Breeding and Foraging Habitat)

Modeled breeding habitat for California red-legged frog is defined by the following land-cover types: lacustrine (excluding the largest reservoirs such as Camp Far West, Folsom), fresh emergent wetlands, seasonal wetlands, riverine, valley foothill riparian, stock ponds, urban riparian, and urban wetland at elevations above 200 feet.

Upland Habitat (Upland Refugia and Movement Habitat)

Upland refugia habitat is defined as all oak woodland land-cover types, annual grassland, and pasture within 100 feet of modeled breeding habitat. Movement habitat is defined as all oak woodland, annual grassland, pasture, valley foothill riparian, all agricultural land-cover types, urban riparian, urban wetland, and landscape and golf course ponds beyond 100 feet but within one mile of modeled breeding habitat.

Rationale

Modeled habitat was restricted to elevations above 200 feet in the Plan Area because California red-legged frogs are believed to be extirpated from the floor of the Central Valley. Historically, the California red-legged frog occupied sections of the western slope of the Sierra Nevada. California red-legged frogs use a variety of aquatic habitats for breeding, including streams, deep pools, backwaters within streams and creeks, and ponds (USFWS 2002). Breeding adults are often associated with deep (> 2 feet) still or slow moving water and dense, shrubby riparian or emergent vegetation (Hayes and Jennings 1988). California red-legged frogs also breed in stock ponds and other artificial impoundments that are managed to provide suitable habitat.

If aquatic breeding habitats dry up during the summer, California red-legged frog often disperse to other areas with water or to temporary shelter or aestivation sites (U.S. Fish and Wildlife Service 2002). Temporary shelter and aestivation sites include shelter under boulders, rocks, logs, and leaf litter. Dispersal and migration movements can be straight-line movements or along migratory corridors such as riparian habitats (Bulger et al. 2003; Fellers and Kleeman 2007); however, the distance moved, and habitats moved through, is site-dependent (e.g., proximity of breeding and non-breeding habitat), and influenced by local landscape (Bulger et al 2003; Fellers and Kleeman 2007). Because the actual movement patterns of California red-legged frog in western Placer County landscapes is generally not known, movement habitat was conservatively modeled to include suitable land-cover types within a radius of one mile from all potential breeding sites.

Model Results

Species Map 8. *California Red-legged Frog Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for California red-legged frog in the Plan Area. Potential suitable habitat is located in the foothills portion of the Plan Area above 200 feet in elevation. Due to the abundance of aquatic primary habitat in the eastern portion of the Plan Area, the associated upland refugia and movement habitat is extensive.

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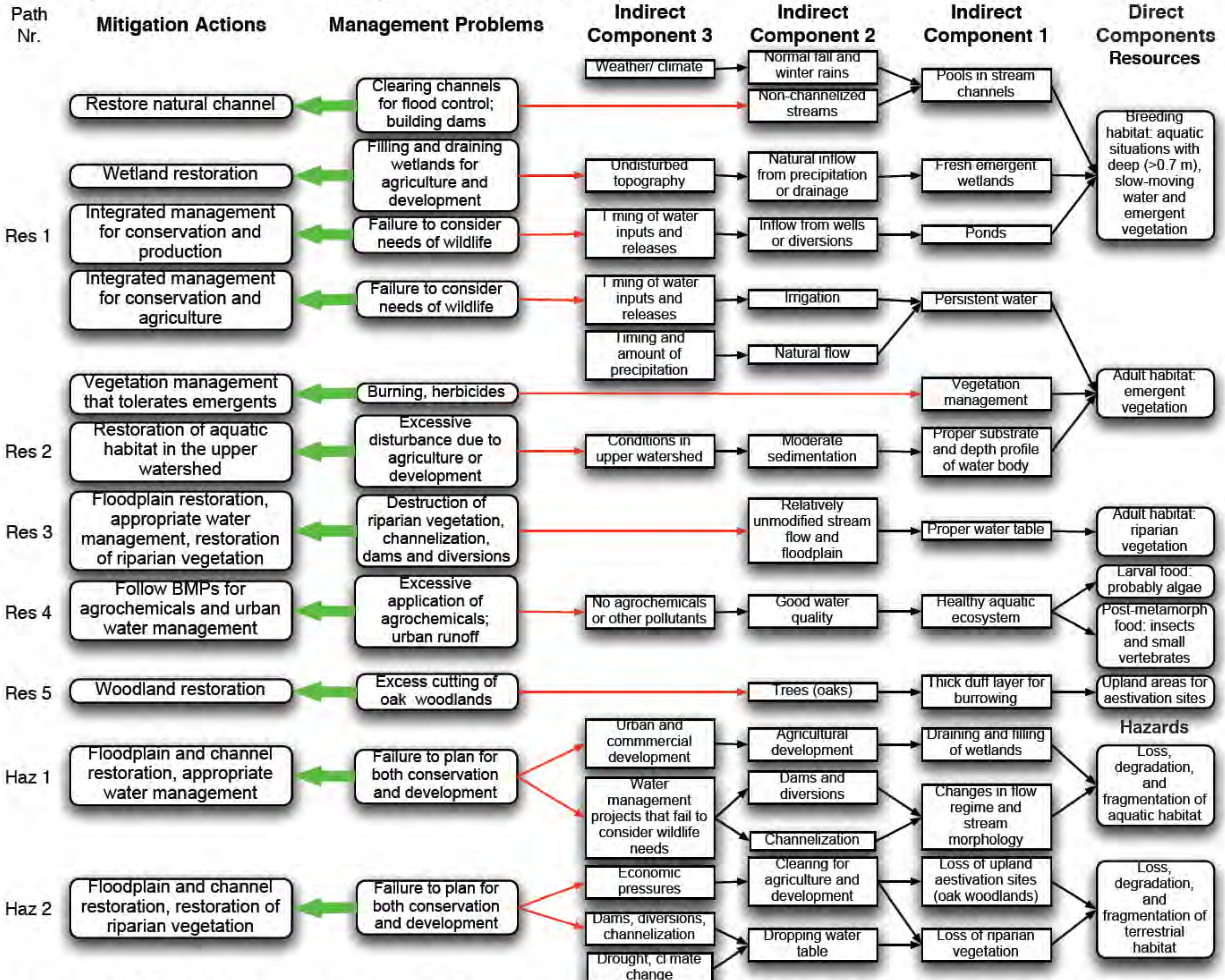
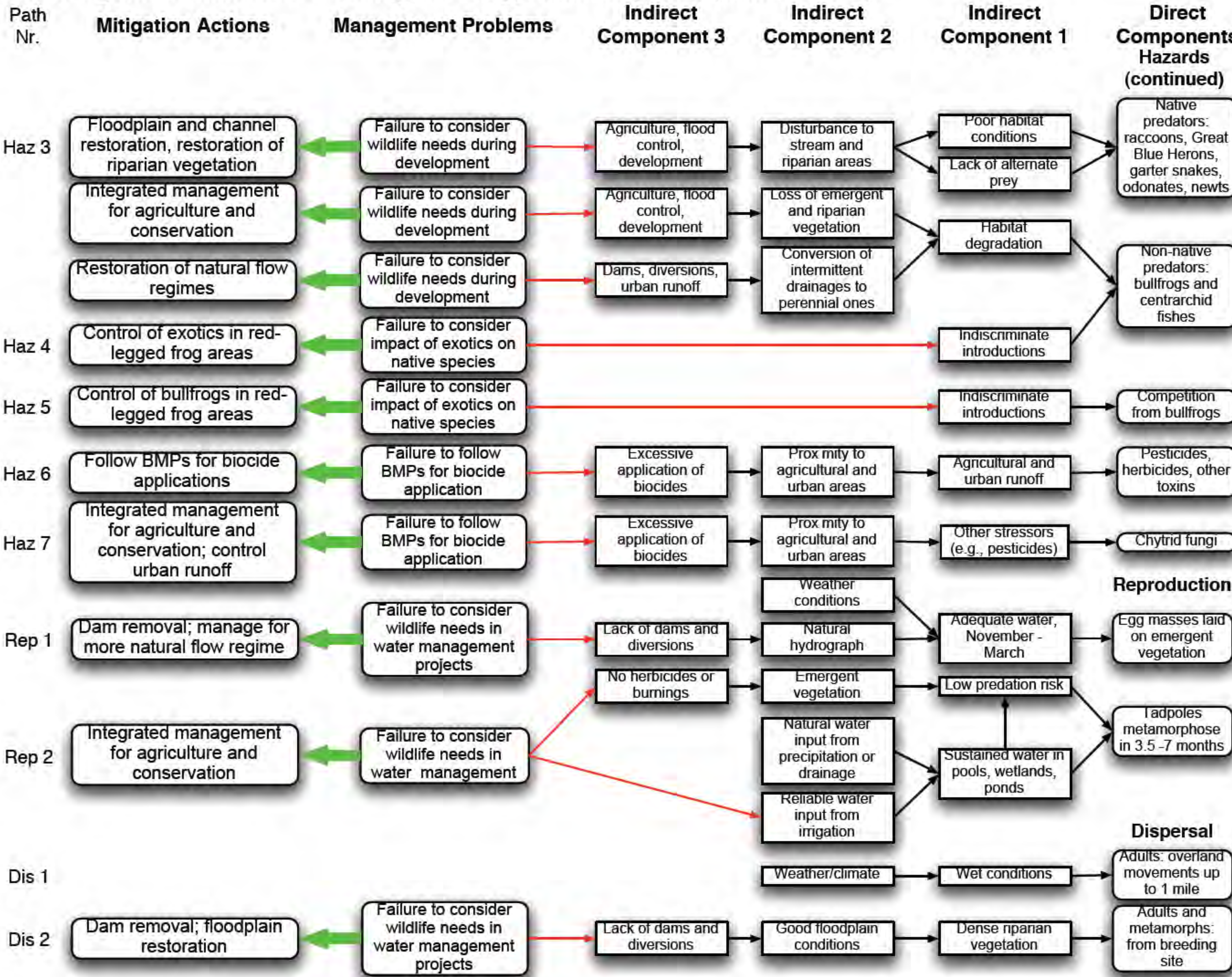
Envirogram 8 California Red-legged Frog, *Rana draytonii*



Envirogram 8 California Red-legged Frog. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram 8 California Red-legged Frog, *Rana draytonii* (continued 2)



Envirogram Narrative

California Red-legged Frog (*Rana draytonii*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to help depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: California red-legged frog requires aquatic habitats with deep (>2.3 feet), slow-moving water and emergent vegetation for breeding. These microhabitats are found in pools in stream channels, fresh emergent wetlands, and artificial ponds. Appropriate stream pools are most likely to be found in non-channelized creeks that flow as a result of normal fall and winter rains. Channelized and impounded streams sometimes can be restored to more natural conditions to improve breeding conditions for the frog. Areas with natural depressions that capture precipitation or runoff often form fresh emergent wetlands with the proper microhabitat for California red-legged frog breeding. Many of these wetlands in Placer County have been filled or drained, but it is possible to restore some of them. Sometimes farm ponds that receive inflow from diversions or wells also provide breeding habitat, provided that the timing of water inputs and releases is compatible with California red-legged frogs' needs. Integrated management for conservation and agricultural production can help insure this.

Res2: Emergent vegetation is important adult habitat for California red-legged frog. Persistent water and a proper substrate and depth profile of the water body are necessary for emergent vegetation. Persistent water can come either from irrigation or natural flow. The timing of water inputs and releases is important to the former, and this depends on how well conservation and agriculture is integrated. Natural flow largely depends on the timing and amount of precipitation. Proper substrate and depth profiles depend on a moderate amount of sedimentation, a function of conditions in the upper watershed. Excessive disturbance may result in too much sediment and loss of aquatic habitat; extensive restoration of both the habitat and upper watershed may be necessary. Finally, the persistence of emergent vegetation also depends on a management strategy that tolerates its presence rather than using burning or herbicides to eliminate it.

Res3: Riparian vegetation regulates water temperature, helps maintain water quality, and provides cover and thus is another important adult habitat component for the California red-legged frog. The development of riparian vegetation depends on a proper water table that in turn depends on unmodified stream flow and a healthy floodplain. The effects of destruction of riparian vegetation, stream channelization, and dams and diversions need to be reversed in California red-legged frog conservation areas.

Res4: California red-legged frog tadpoles eat algae, and adults eat a variety of insects and small vertebrates. The presence of these items depends on the presence of a healthy aquatic ecosystem and good water quality free of pollutants. Excessive application of agrochemicals and urban runoff result inpollution; following best management practices for chemical use and runoff management can help ensure healthy aquatic ecosystems.

Res5: Adult and metamorph California red-legged frogs use uplands as aestivation sites to survive pond/creek drying. (This is probably one reason why it occasionally can persist in the presence of bullfrogs—bullfrogs will not use uplands.) Appropriate sites need a thick duff layer into which the frogs can burrow; these are usually found in oak woodlands. Persistent oak removal has made these sites scarce in western Placer County. Restoration and enhancement of oak woodland would benefit California red-legged frog.

Hazards

Haz1: As with most species in Placer County, one of the biggest hazards for the California red-legged frog is the loss and degradation of its aquatic habitats resulting from the draining and filling of wetlands for agricultural, urban, and commercial development and from changes to the flow regime and morphology of streams through dams and diversions and channelization. Floodplain and channel restoration and water management that considers wildlife's needs can mitigate this hazard to some extent.

Haz2: A second major hazard is the loss and degradation of the California red-legged frog's terrestrial habitat, riparian vegetation, and upland aestivation sites (usually oak woodlands). Clearing for agriculture and development and dropping water tables resulting from dams, diversions, and stream channelization are largely responsible for the loss of this vegetation. Floodplain and channel restoration and the restoration of riparian vegetation may help mitigate these problems.

Haz3: The California red-legged frog is preyed upon by a wide variety of native predators. Raccoons, great blue herons, and garter snakes take post-metamorphic stages, while odonate larvae and newts prey on tadpoles. Poor habitat conditions and the scarcity of alternate prey resulting from various disturbances to aquatic and terrestrial habitats exacerbate predation pressure. Floodplain and channel restoration and the restoration of riparian vegetation should help reduce native predation pressure to tolerable levels.

Haz4: Non-native predators, particularly bullfrogs and centrarchid fishes, are a much bigger problem for the California red-legged frog than are native predators. Non-native predators result from both thoughtless and indiscriminate introductions and habitat modifications that favor exotic species. Loss of emergent and riparian vegetation from agriculture, flood control, and development and the conversion of intermittent streams into perennial ones because of impoundments and irrigation runoff are habitat modifications that make California red-legged frogs more vulnerable to predation by exotic species. Integrated management for agriculture and conservation, restoration of natural flow regimes, and control of exotics in California red-legged frog conservation areas are necessary to reduce these impacts.

Haz5: Bullfrogs also compete directly with adult California red-legged frogs for food and space. Bullfrogs must be controlled in California red-legged frog conservation areas.

Haz6: Pesticides, herbicides, and other toxins in agricultural and urban runoff, largely resulting from over- application of these chemicals, can poison California red-legged frog directly. Following best management practices for the application of pesticides can help mitigate this problem.

Haz7: Chytrid fungi and perhaps other pathogens attack California red-legged frog that have been already weakened by other stressors such as pesticides, herbicides, and other toxins in agricultural and urban runoff. The presence of these pesticides in aquatic habitats usually result from their over-application, and following best management practices for the application of pesticides can help mitigate this problem.

Reproduction

Rep1: California Red-legged frog egg masses are laid on emergent vegetation so that they float on the surface. The presence of appropriate breeding conditions depends on adequate water from November to March and is related to both weather conditions and a natural flow regime. If the flow regime is altered by dams or diversions, a management regime that considers the California red-legged frog should be instituted.

Rep2: Tadpoles metamorphose in 3.5-7 months provided that there is sustained water and low predation risk. Sustained water can come from either natural sources or from irrigation, and emergent vegetation provides cover that lowers predation risk. Managing water and vegetation in ways that benefit the California red-legged frog are necessary in agricultural areas with conservation easements.

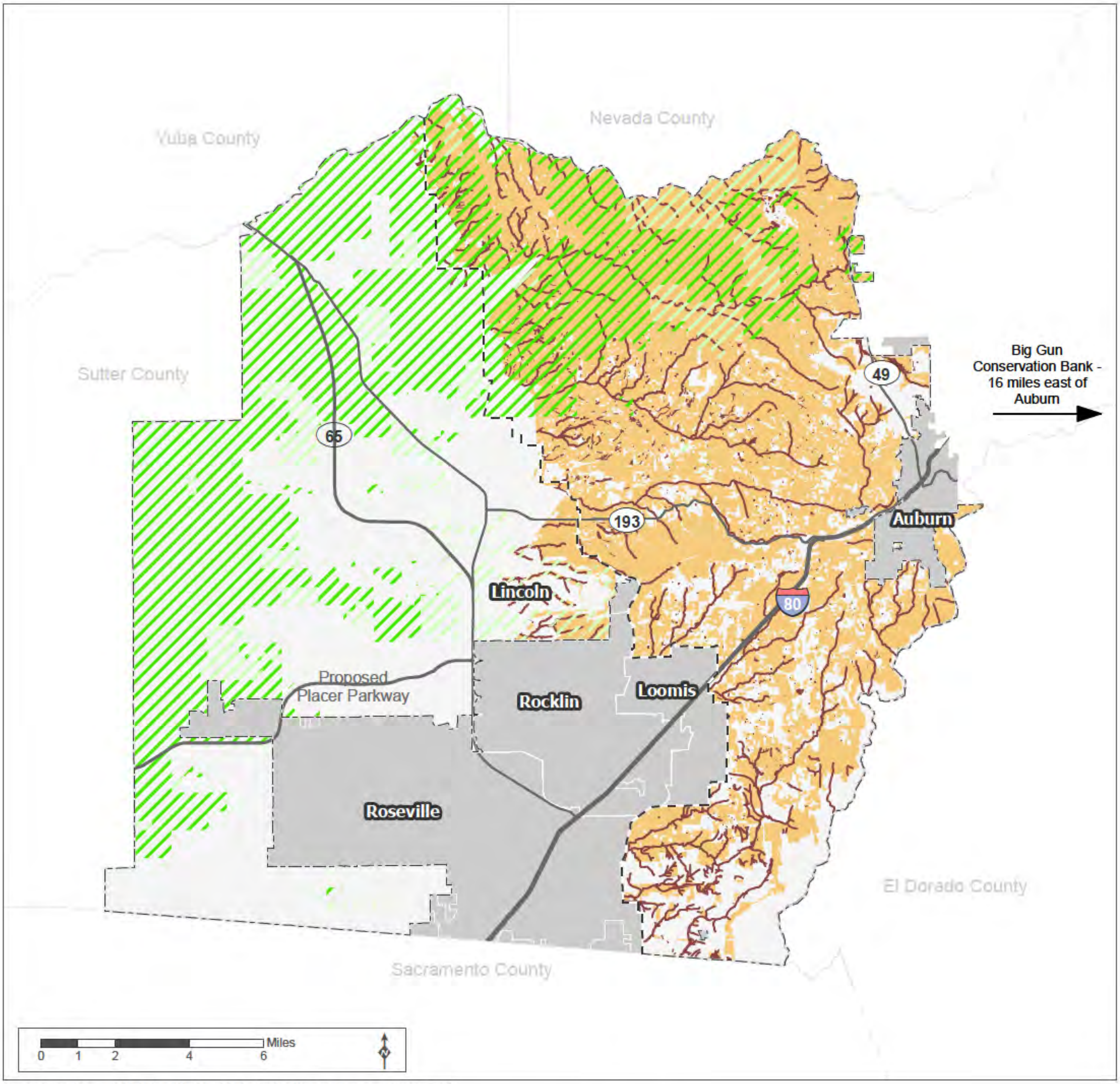
Dispersal

Dis1: In addition to movements to aestivation sites during dry conditions, adult California red-legged frog has been known to disperse overland up to one mile during wet conditions. These movements tend to be in straight lines, point-to-point, rather than along corridors.

Dis2: Adults and metamorphs disperse from breeding sites using dense riparian vegetation for cover. Thus, the presence of dispersal habitat depends on a proper water table that in turn relies on unmodified stream flow and a healthy floodplain. Dam removal and floodplain restoration are necessary in California red-legged frog conservation areas.

Summary

California red-legged frog is a conservation challenge not only because it relies on both aquatic and terrestrial habitats but also because it is so strongly affected by bullfrogs and centrarchid fishes that are now virtually ubiquitous in the western part of Placer County. California red-legged frog is also susceptible to various pesticides and pathogens—also virtually ubiquitous. Restoration and enhancement of habitat (healthy wetlands and creeks with natural flow regimes and channel morphology and functional floodplains), along with diligent control of exotic species, may be an effective conservation strategy for this species.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2015; Westervelt Ecological Services, 2015

Occurrences
General Location

Modeled Habitat
Aquatic Habitat
Upland Habitat
Modeled as Non-habitat

Exis ing Reserve
Reserve Aquisition Area
NPC

Major Road
Valley/Foothill Divide
Area A Boundary

Species Map 8.

California Red-legged Frog Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

Central Valley Steelhead – Distinct Population Segment (*Oncorhynchus mykiss irideus*)

Status

Federal: Threatened (NMFS 1998a; 2006); Magnuson-Stevens Act managed species

State: None

Critical Habitat: Critical habitat designated September 2, 2005 (NMFS 2005)

Recovery Plan: Recovery Plan for Sacramento River winter-run Chinook salmon, Central Valley spring-run Chinook salmon, and Central Valley steelhead (NMFS 2014)

Distribution

North America

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

California

The Central Valley steelhead Distinct Population Segment (DPS) includes the Sacramento and San Joaquin rivers, along with all of their tributaries (NMFS 2009). Existing wild steelhead populations in the Sacramento River basin occur in the upper Sacramento River and its tributaries, including Cottonwood, Antelope, Deer, and Mill creeks and the Yuba River (NMFS 2014). Other Sacramento River basin populations may exist in Big Chico and Butte creeks, and a few wild steelhead are produced in the American and Feather rivers (McEwan 2001). A hatchery supported population of steelhead also occurs in the Mokelumne River, which flows directly into the Delta in between where the Sacramento and San Joaquin rivers enter the Sacramento-San Joaquin Delta (Delta) (NMFS 2014). Central Valley steelhead were thought to be extirpated from the San Joaquin River system, until recent monitoring detected small populations of steelhead in the Stanislaus, Mokelumne, and Calaveras rivers, and other streams previously thought to be devoid of steelhead (McEwan 2001).

Placer County Plan Area

<u>Historical</u>

McEwan (2001) synthesized many historical accounts of steelhead and Chinook distribution throughout the Central Valley and estimated that steelhead were historically well distributed throughout Sacramento and San Joaquin watersheds, including those west-draining tributaries of the Sacramento. One-time sampling events conducted in 1966, 1967, 1972, and 1984 on Secret Ravine indicated the presence of juvenile steelhead (California Department of Fish and Wildlife [CDFW] Region II memos summarized by Bailey 2003). Auburn Ravine sampling events in 1959, 1971, 1979, and 1984, summarized by Bailey (2003), indicate historic presence of steelhead. Although no sampling data were found, Bailey (2003) reported several anecdotal stories as evidence of historic steelhead presence in the Coon Creek watershed.

<u>Current</u>

Central Valley steelhead is known to be present in the Bear River, Coon Creek (including the Doty Ravine tributary), Auburn Ravine, and Dry Creek (including Secret Ravine and Miner's Ravine tributaries) (Bailey 2003; County of Placer 2009; NMFS 2009). Coon Creek and one of its tributaries, Doty Ravine, as well as Dry Creek and two of its tributaries, Secret Ravine and Miners Ravine, are listed as critical habitat for Central Valley steelhead (NMFS 2005). Secret Ravine supports the highest quality habitat for steelhead in the Dry Creek watershed. Coon Creek appears to contain good migration corridors for adult salmonids, patchy spawning habitat and good juvenile rearing habitat in lower reaches, and good spawning habitat and juvenile rearing habitat in the reach from McCourtney Road to the downstream end of the canyon section below Garden Bar Road. Doty Ravine contains good migration corridors and juvenile rearing habitat. Lower reaches contain primarily small-sized sediments (sand and gravel) with occasional small patches of larger material. Spawning gravel is larger and more abundant in upstream reaches.

Population Status & Trends

California

Overall, population trend data is limited for Central Valley steelhead (Williams et al. 2011). Steelhead apparently were common in the Central Valley tributaries, but records for them are few and fragmented (Yoshiyama et al. 1998). One of the primary problems with determining the population status of steelhead is the difficulty in distinguishing between the truly ocean-run fish and resident rainbow trout (Vogel 2011). What is known is that Central Valley steelhead are now restricted to the Sacramento River downstream of Keswick Dam; the lower reaches of the Feather River, American River, and other large tributaries downstream of impassable dams; small, perennial tributaries of the Sacramento River; and the Sacramento–San Joaquin Delta. Lindley et al. (2006) estimated that historically there were at least 81 independent Central Valley steelhead populations primarily distributed throughout the eastern tributaries of the Sacramento and San Joaquin rivers. Hallock (1987) as cited by Vogel (2011) estimated that upper Sacramento River steelhead populations decreased from more than 20,000 in the 1950s to less than 5,000 in the 1980s. In 1996, NMFS estimated the Central Valley steelhead run size based on dam counts, hatchery returns, and past spawning surveys. They found that probably fewer than 10,000 fish were present. Presently, impassable dams block access to 80 percent of historically available habitat, and block access to all historical spawning habitat for about 38 percent of historical populations (Lindley et al. 2006). Good et al. (2005) estimated that an average of 3,628 naturally-spawning females spawned in the entire Central Valley between 1998 and 2000. This estimate was calculated by applying the following two assumptions to juvenile steelhead abundance data collected downstream of the Sacramento-San Joaquin river confluence between 1998 and 2000: each female lays 5,000 eggs and approximately 1% of all eggs survive (Good et al. 2005). The most recent status review for the Central Valley steelhead DPS was completed in June 2005 (Good et al. 2005). The majority opinion of the Biological Review Team (BRT) (66% of the members) was that the Central Valley steelhead DPS is "in danger of extinction." This is in agreement with three previous status reviews (Busby et al. 1996; National Marine Fisheries Service 1997 and 1998a). Overall, the status of Central Valley steelhead appears to have worsened since the Good et al. (2005) status review when the BRT concluded that the

Central Valley steelhead DPS was in danger of extinction (Williams et al. 2011). An idea of the decline of steelhead can be obtained by looking at returns to the upper Sacramento River, which are based on counts from fish ladders and hatchery returns. These estimates went from an average of 6,574 steelhead in 1967-1991 to an average of 1,282 steelhead from 1992 to 2008 (Moyle et al. 2008).

Placer County Plan Area

Status

The CDFW has, throughout its management history of the Dry Creek drainage, regarded Antelope Creek and its tributary, Clover Valley Creek, as salmonid spawning and rearing habitat. In a memorandum dated October 19, 1964, CDFW staff reported the presence of rainbow trout in upper Clover Valley creek (CDFW 2015). Spawned-out salmon carcasses and live salmon were observed in Clover Valley Creek in December 1963 and salmon fry were observed in the creek in April 1964 (CDFW 2015). Currently, there is no reliable data to determine whether steelhead are present in Antelope Creek or Clover Valley Creek.

A 2004 to 2005 fish community survey was performed by the California Department of Fish and Game (2008) throughout the main stems of Auburn Ravine (seven sampling locations) and Coon Creek (seven sampling locations) in western Placer County. Multiple-pass, depletion electrofishing methods were applied in December of 2004 and again in April and June of 2005 (California Department of Fish and Game 2008). Steelhead were found to be, on average, the most abundant fish species during both the winter 2004 and spring 2005 sampling efforts in Auburn Ravine. Enough steelhead data were collected to estimate an average of 2,163 juvenile steelhead present per river mile between the McBean Park and Wise Road sampling locations. Far fewer steelhead were found on Coon Creek. At the Spears Ranch Lower sampling location in Coon Creek, one juvenile steelhead was found in December of 2004 and 12 juveniles were found in April of 2005 (CDFW 2008).

Various small spring and summer survey efforts conducted between 1992 and 2002 in the main stem of Dry Creek yielded no evidence of juvenile steelhead rearing (Bailey 2003). Electrofish and screw trap sampling conducted between the winter of 1998 and the summer of 2000 in Miners and Secret Ravine documented the presence of steelhead in both Dry Creek tributaries. In Miners Creek, juvenile steelhead were found exclusively at the Dick Cook Road crossing site, while steelhead were somewhat common in the central and upper portions of Secret Ravine (CDFG memo summarized by Bailey 2003). In addition, several steelhead smolts were caught in the spring of 1999 and 2000 just downstream of the confluence of Secret and Miners Ravine, suggesting the presence of a naturally-spawning population.

During the fall/winter of 2004 and the spring of 2005, CDFW conducted two-pass electrofishing surveys on a total of seven reaches in Dry Creek, as well as in several reaches in Miners and Secret ravines. During the 2004 fall/winter survey, 41 steelhead/rainbow trout were captured in Secret Ravine and during the 2005 spring survey, 95 steelhead/rainbow trout were captured here (CDFW 2005, unpublished data). During the 2005 spring survey in Secret Ravine, five pit-tagged steelhead/rainbow trout were re-captured from the 2004 fall/winter survey. No steelhead/rainbow trout were identified in Dry Creek or Miners Ravine.

A 2013 study found different life stages of rainbow trout/steelhead in Auburn Ravine. Life stage developmental stages were classified as parr, silvery parr, and smolt. Auburn Ravine is designated critical habitat for Central Valley steelhead; however, opportunities for anadromy have been compromised for several decades and thus the collection of smolt life stages is an important finding (Healey 2014).

Trends

With little historic or current steelhead population data for western Placer County, assessing current trends is difficult. However, several regional trends suggest population declines throughout the area are likely. As mentioned above in the California Population Status and Trends section, the majority opinion of the NMFS biological review team was that Central Valley steelhead are in danger of extinction (NMFS 2005). In addition, 2008 saw the smallest number of returning adult Sacramento River Chinook salmon since records began in 1970 (Pacific Fishery Management Council Salmon Technical Team 2009). The proximate cause of the decline was found to be poor ocean conditions (National Marine Fisheries Service 2009). These poor ocean conditions likely had a similar impact on Central Valley steelhead populations.

Natural History

The habitat requirements, ecological relationships, life history, and threats to Central Valley steelhead described below are summarized in diagram form in the Envirogram 9 Central Valley Steelhead.

Habitat Requirements

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

The quality of spawning habitat is also correlated with intra-gravel flow. Low intra-gravel flow may provide insufficient dissolved oxygen, contribute to growth of fungus and bacteria, and result in high levels of metabolic waste. High percentage of fines in gravel substrates can substantially limit intra-gravel flow, affecting the amount of spawning gravel available in the river (Healey 1991). Raleigh et al. (1986) concluded that optimal gravel conditions would include less than 5–10% fine sediments measuring 0.12 inch or less in diameter. In addition, alevins of Chinook salmon (*Oncorhynchus tshawytscha*), steelhead, and coho salmon (*Oncorhynchus kisutch*) have been observed in laboratory studies to have difficulty emerging when gravels exceeded 30–40% fine sediments (Phillips et al. 1975 as cited in Bjornn and Reiser 1991; Waters 1995).

Water depth criteria vary widely; there is little agreement among studies about the minimum and maximum values for depth (Healey 1991). Salmonids will spawn in water depths that range from a few inches to several feet. A minimum depth of 0.8 foot for steelhead spawning has been widely used in the literature and is within the range observed in some Central Valley rivers (California Department of Fish and Game 1991). Minimum water depth for steelhead spawning has been observed to be at least deep enough to cover the fish (Bjornn and Reiser 1991). Many fish spawn in deeper water.

Preferred water temperature range for steelhead spawning is reported to be approximately 30–52ºFarenheit (ºF) (CDFW 2000 as cited in NMFS 2014). Conditions supporting steelhead spawning and incubation are assumed to deteriorate as temperature warms to 52–59ºF (Myrick and Cech 2001). Steelhead eggs that are subjected to temperatures warmer than 59ºF are prone to increased mortality.

Rearing habitat for salmonids is defined by environmental conditions such as water temperature, dissolved oxygen, turbidity, substrate, area, water velocity, water depth, and cover (Bjornn and Reiser 1991; Healey 1991; Jackson 1992). Environmental conditions and interactions among individuals, predators, competitors, and food sources determine habitat quantity and quality and the productivity of the stream (Bjornn and Reiser 1991). Regardless of life history strategy, for the first year or two, rainbow trout and steelhead are found in cool, clear, fast-flowing permanent streams and rivers where riffles predominate over pools, there is ample cover from riparian vegetation or undercut banks, and invertebrate life is diverse and abundant (Moyle 2002). Everest and Chapman (1972) found juvenile Chinook salmon and steelhead of the same size utilizing similar in-channel rearing areas. Juvenile steelhead are year-round residents; they generally use riffles and runs in the main and secondary channels along with the head and tail of pools. Shallow riffles are the most important channel type for steelhead during their first year (Barnhart 1986). Steelhead also uses seasonal habitats of intermittent streams for rearing (McEwan 2001). Floodplain habitat does not appear to provide significant rearing habitat for steelhead as it does for Chinook salmon.

Water velocity is of particular importance in determining where juvenile salmonids occur because it determines the energetic requirements for maintaining position and the amount of food delivered to a particular location. Juvenile salmonids tend to select positions that maximize energetic gain, but these positions can be altered by interaction with other fish and the presence of cover (Shirvell 1990). The water velocity preferred by Chinook salmon varies with size of the fish; larger fish occupy higher velocity and deeper areas than smaller fish, potentially gaining access to abundant food and avoiding predatory birds (Bjornn and Reiser 1991; Jackson 1992). Griffith (1972) as cited in Raleigh et al. (1984) found water velocities of 0.32–0.72 foot per second to be associated with occurrence of rainbow trout. Sheppard and Johnson (1985) found similar results for juvenile steelhead; they measured velocities of 0.40 to 0.80 foot per second. Bovee (1978) as cited in California Department of Fish and Game (1991) presented water velocities of 0.6 –1.2 feet per second as having a suitability index of 1 for juvenile rainbow trout and steelhead. Moyle (2002) found that water velocities over redds are typically 20 to 155 centimeters per second.

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

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

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

Juvenile steelhead can be found where daytime water temperatures range from 32–81ºF in the summer, although mortality may result at extremely low (<39 ºF) or extremely high (>73 ºF) water temperatures (NMFS 2014). Juvenile rearing success is assumed to deteriorate at water temperatures of

62.6–77ºF (Raleigh et al. 1984; Myrick and Cech 2001). Smolt transformation requires cooler temperatures than rearing; successful transformation occurs at temperatures of 42.8–50ºF. Juvenile steelhead, however, are observed to migrate through the Delta at water temperatures substantially warmer than 55ºF. Juvenile steelhead has been captured at Chipps Island in June and July and at water temperatures exceeding 68ºF (Nobriega and Cadrett 2001). Optimal water temperatures for growth of steelhead have been reported to be 59 ºF to 64.4 ºF (Moyle 2002).

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

Reproduction

Spawning in the Sacramento River basin typically occurs from late December through April, with most adults spawning in January through March. The female steelhead selects a site with good intergravel flow, digs a red with her tail (usually in coarse gravel in or near a riffle), and deposits eggs while an attendant male fertilizes them (NMFS 2014). Water velocities over redds are typically 20 to 155 centimeters per second and the depths are 10 to 150 centimeters (Moyle 2002). The preferred water temperature range for steelhead spawning is approximately 30–52ºF (CDFW 2000 as cited in NMFS 2014). The eggs hatch 19 to 80 days after spawning, depending on water temperature (NMFS 2014). Steelhead eggs can survive at water temperatures of 35.6–59ºF; however, the highest survival rates are observed at water temperatures from 44.6–50ºF (Myrick and Cech 2001 as cited in NMFS 2014). Larvae remain in the gravel for four to six weeks before emerging as young juveniles or fry and begin actively feeding (NMFS 1998b; Moyle 2002). Unlike other pacific salmonids, steelhead are capable of spawning more than once before they die. However, it is rare for a steelhead to spawn more than twice before dying (Moyle 2002).

Dispersal Patterns

Adult Central Valley steelhead migrates upstream from the ocean July through May; most migrate after October and before May. Juvenile migration to the ocean generally occurs from November through May. Based on salvage data at the state and federal export facilities in the Delta, the peak months of juvenile migration appear to be March and April. After two to three years of ocean residence, adult steelhead returns to the natal stream to spawn as four or five-year-olds (National Marine Fisheries Service 1998b). Juvenile steelhead rear a minimum of one and typically two or more years in fresh water before migrating to the ocean following smoltification (e.g., the process of physiological change that allows ocean survival).

Longevity

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

Sources of Mortality

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

Reynolds et al. (1993) reported that 95 percent of salmonid habitat in California's Central Valley has been lost, mainly due to mining and water development activities. They also noted that declines in Central Valley steelhead stocks are due mostly to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions. Entrainment at diversions is a source of mortality; low flows can confuse or detain migrating juveniles, resulting in higher entrainment at diversions.

Behavior

Steelhead and rainbow trout are the same species. In general, steelhead refers to the anadromous form of the species. Normally, adult steelhead reach a larger size than resident rainbow trout (NMFS 2014). Currently, Central Valley steelhead are recognized only as winter-run, although prior to the construction of large dams there may have been summer-run steelhead present (Moyle 2002; Sandtrom et al. 2012).

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

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

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

Movement and Migratory Patterns

Steelhead typically migrate to marine waters after spending two years in freshwater (NMFS 2014). They typically reside in marine waters for two or three years prior to returning to their natal stream to spawn as four or five year olds (NMFS 2014). Central Valley steelhead enter freshwater from August through April (NMFS 2014). Steelhead adults typically spawn from December through April, with peak spawning occurring from January through March in small streams and tributaries where cool, well oxygenated water is available year-round (McEwan 2001). Juvenile steelhead generally migrate to the ocean in spring and early summer at one to three years of age, with migration through the Delta occurring in March and April (NMFS 2014). Steelhead may remain in the ocean from one to four years, growing rapidly as they feed on highly productive currents, before they return to freshwater (NMFS 2014).

Ecological Relationships

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

Population Threats

The widespread degradation, destruction, and blockage of freshwater habitats within the Central Valley, and the continuing impacts to habitat resulting from water management were identified as key reasons why Central Valley steelhead were listed under the Endangered Species Act. Good et al. (2005) described the threats to Central Valley salmon and steelhead as falling into three broad categories: loss of historical spawning habitat, degradation of remaining habitat, and genetic threats from the stocking programs. The decline in steelhead populations is attributable to changes in habitat quality and quantity. The availability of steelhead habitat in the Central Valley has been reduced by as much as 95% or more by barriers to movement (i.e., dams). Other factors contributing to the decline of steelhead in the Central Valley include mining; agriculture; urbanization; logging; harvest; hatchery influences; and flow management, including reservoir operations, hydropower generation, and water diversion and extraction (NMFS 1996).

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

Along with habitat loss and habitat degradation, hatchery management was identified as a key factor in listing the Central Valley steelhead (NMFS 1998a). Over the past several decades, the genetic integrity of the Central Valley steelhead population has been diminished by increases in the proportion of hatchery fish relative to naturally produced fish, the use of out-of-basin stocks for hatchery production, and straying hatchery produced fish (NMFS 2014). Potential threats to steelhead from hatchery programs includes: mortality of natural steelhead in fisheries targeting hatchery origin fish, disease transmission, and genetic introgression by hatchery origin fish that spawn naturally and interbreed with natural populations (NMFS 2014).

Predation on steelhead parr and smolts by both native and non-native predators is highly likely both in their native rivers and during their migration through the lower rivers in the Delta. In Clifton Court

Forebay, tagged hatchery smolts are known to be heavily preyed on by striped bass (*Morone saxatilis*). However, predation on steelhead is difficult to quantify.

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

Context for a Regional Conservation Strategy

As few sampling efforts have been conducted in Placer County, knowledge of Central Valley steelhead distribution in the Plan Area is incomplete. Central Valley steelhead is known to be present in the Plan Area in Bear River, Coon Creek (including the Doty Ravine tributary), Auburn Ravine, and Dry Creek (including Secret Ravine and Miner's Ravine tributaries) (Bailey 2003; County of Placer 2009; NMFS 2009). Coon Creek and one of its tributaries, Doty Ravine, as well as Dry Creek and two of its tributaries, Secret Ravine and Miners Ravine, are listed as critical habitat for Central Valley steelhead (NMFS 2005, p. 52614). In California, the population is restricted to the Sacramento and San Joaquin rivers and their tributaries. Due to the need for unimpeded access from spawning sites to the ocean, maintenance of migration routes and habitat in the Plan Area has great bearing on the California population throughout the downstream Sacramento riverine system. Degradation of habitat within Placer County and the species range overall has led to declines in steelhead populations. Restoration efforts within the Plan Area are most crucial to maintaining and restoring population levels.

The Recovery Plan (NMFS 2014) states that presently, no viable independent steelhead populations have been identified and all are at high risk of extinction. Therefore, the recovery strategy includes securing extant populations in the near-term, and establishing spawning populations in numerous streams and rivers within individual Diversity Groups throughout the Central Valley. The recovery strategy for the Auburn Ravine/Coon Creek watershed includes the maintenance of steelhead spawning populations in the upper reaches of Auburn Ravine and Coon Creek, and in Doty Ravine. The recovery strategy for the Dry Creek Watershed includes the maintenance of steelhead spawning populations in Miner's Ravine and Secret Ravine.

Modeled Species Distribution in the Plan Area

Model Assumptions

<u>Spawning and Rearing Habitat</u>

Modeled spawning and rearing habitat for Central Valley steelhead includes riverine and valley foothill riparian land-cover types in the following river reaches: Coon Creek upstream of Gladding Road; Doty Creek (a tributary to Coon Creek) upstream of Crosby Herold Road; Auburn Ravine east of the Highway 65 bridge in the City of Lincoln; and the entirety of Secret and Miners Ravine in the Dry Creek watershed, as well as Linda, Cirby, Clover Valley, and Antelope creeks.

<u>Migration and Rearing Habitat</u>

Modeled migration and rearing habitat for Central Valley steelhead includes riverine and valley foothill riparian land-cover types in the following river reaches: Bear River between the western border of the County and Camp Far West Reservoir; the main stem of Coon Creek downstream of the Gladding Road Crossing to the western border of the County; Doty Creek downstream of the Crosby Herold Road crossing; Auburn Ravine downstream of the Highway 65 bridge in the City of Lincoln to the western border of the County, and the main stem of Dry Creek downstream of the confluence with Miners and Secret Ravine to the southern border of the County.

Rationale

Central Valley steelhead spawn in stream reaches with flows generally between 7 to 61 in/s (Moyle 2002), depths between 4 and 59 in (Moyle 2002), and temperatures between 30 °F and 55 °F (California Department of Fish and Game 2000, as cited in NMFS 2009). Central Valley steelhead typically spawn in sediments dominated by gravels at the tail of pools or in riffles. Gravel-sized sediment is small enough to be moved by the digging action of an adult female steelhead but coarse enough to provide adequate intergravel flow, and therefore oxygenated water, to incubating eggs and aelvin. Pool tails and riffles are those habitat types that provide the best conditions for intergravel flow. In western Placer County streams, these characteristics are found primarily in the upper reaches of the watersheds.

Because juvenile steelhead migrate downstream during their first year or two of life, rearing occurs throughout the watershed. Juvenile steelhead are primarily found in stream reaches dominated by runs, riffles, and pools and/or characterized by complex habitat such as undercut banks, large woody debris, and an intact riparian canopy. Juvenile steelhead can tolerate a wide range of temperatures, from 32 °F to 81 °F (Moyle 2002), but physiological stress has been documented to begin at 71.6 °F (Nielsen et al. 1994).

Migration and holding habitat are those corridors through which adult and juvenile Central Valley steelhead must travel to get to spawning and rearing grounds. In western Placer County, migration habitat includes the canals, culverts, diversion dams, and hardened stream reaches characteristic of highly urbanized areas.

Rearing habitat in the lower stream reaches of western Placer County is limited to those areas that were able to maintain those remnant habitat characteristics described above. In addition, Central Valley steelhead are also known to take advantage of urbanized stream features such as pools associated with water diversion and flood control structures.

This habitat model uses the spawning, migration, and rearing habitat identified by the National Marine Fisheries Service (NMFS 2009) in the Draft Recovery Plan for Central Valley steelhead. NMFS used the observations and survey data synthesized in the Auburn Ravine and Coon Creek Ecosystem Restoration Plan (County of Placer 2002) and the Dry Creek Coordinated Resource Management Plan (ECORP Consulting 2003) to identify the location of spawning, migration, and rearing habitat.

Model Results

Species Map 9 Central Valley Steelhead *Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for steelhead in the Plan area.

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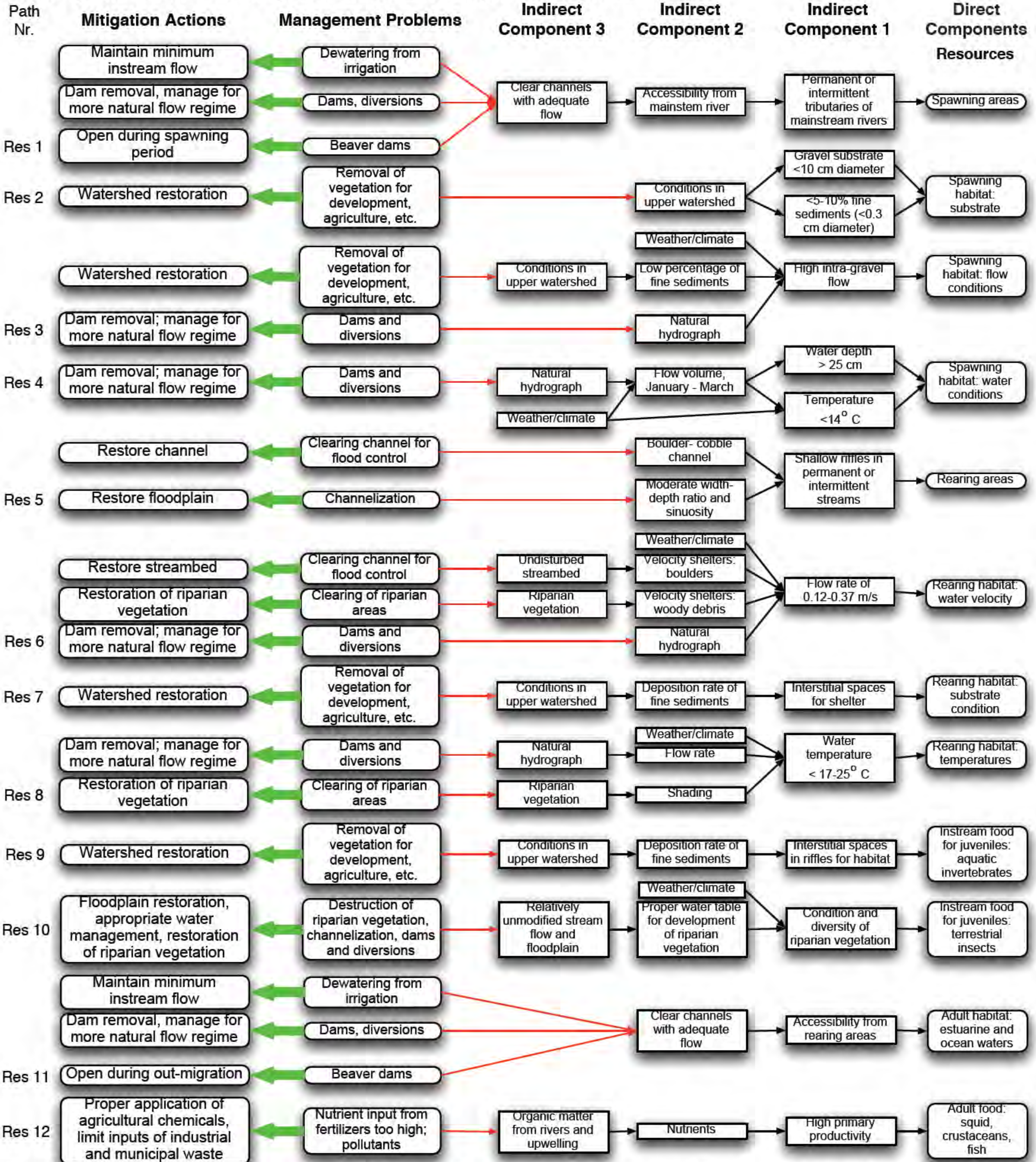
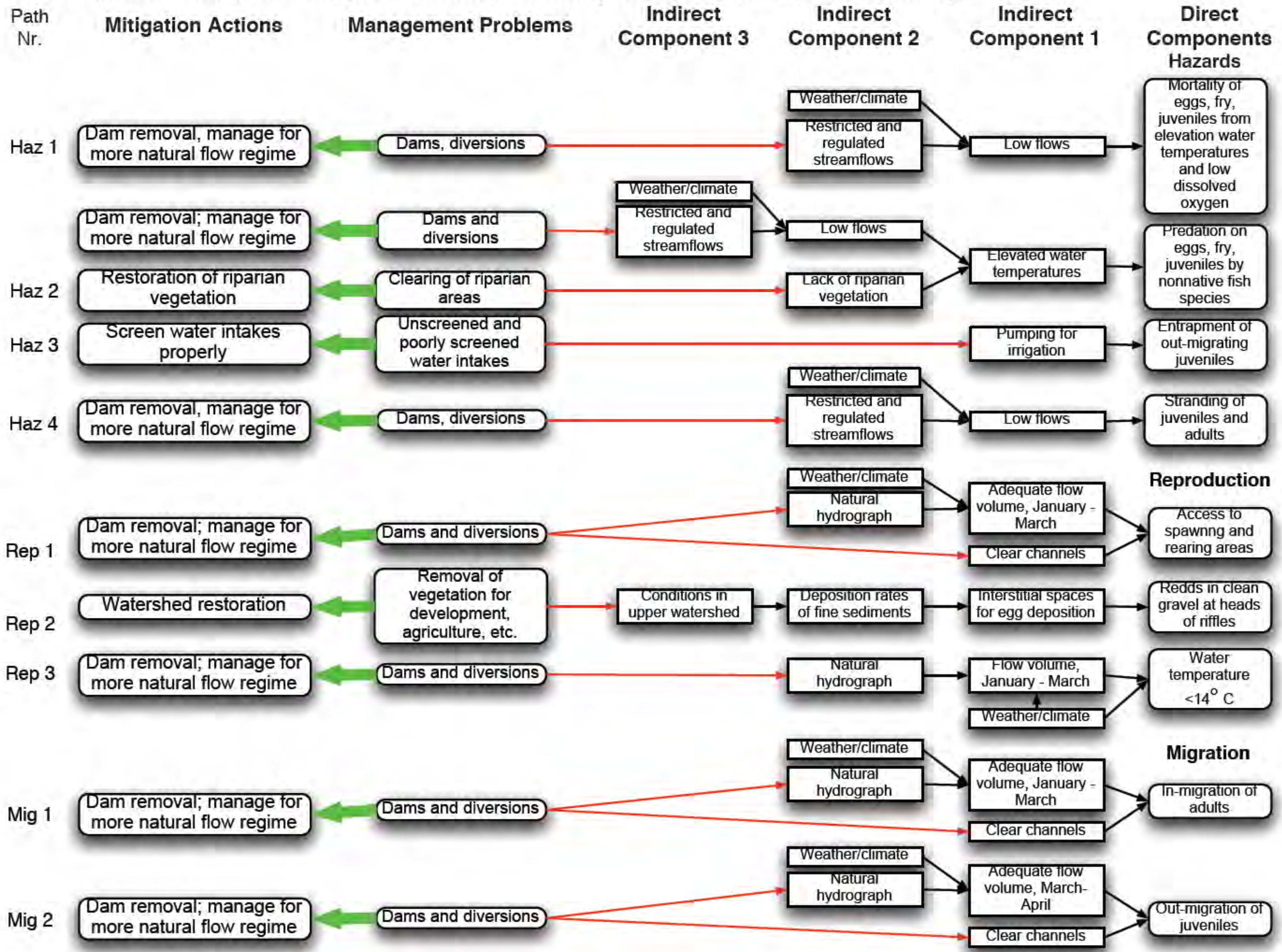
Envirogram 9 Central Valley Steelhead, *Oncorhynchus mykiss*



Envirogram 9 Central Valley Steelhead. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Mig = Migration.

Envirogram 9 Central Valley Steelhead, *Oncorhynchus mykiss* (continued 2)



Envirogram Narrative

Central Valley Steelhead (*Oncorhynchus mykiss irideus*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Spawning areas for the Central Valley steelhead are in permanent or intermittent tributaries of the Sacramento River. The fish need to access these streams from the mainstem river, and to do this they need clear channels with adequate flow. Dams and diversions, dewatering from irrigation, and beaver dams often make this access problematic. Dam removal, maintaining minimum instream flow, managing for a more natural flow regime, and opening up beaver dams in spawning streams improve access. None of these options is without cost; for example, beaver dams are very useful for restoring floodplains and wetlands.

Res2: Central Valley steelhead have very specific requirements for spawning substrate. The gravel needs to be smaller than 10 cm in diameter, and fine sediments need to be less than 5-10% of the substrate. Proper substrate conditions depend to a large extent on conditions in the upper watershed; sedimentation resulting from logging, development, agriculture, or other activities degrades spawning areas. Watershed restoration can help mitigate this problem.

Res3: High intra-gravel flow is important to egg development, and it depends on weather, the amount of fine sediments in the substrate (<5-10%), and on a natural hydrograph (flow regime). These in turn depend on stream characteristics (see path Res1) and the shape of the upper watershed (path Res2).

Res4: Water depth (>9.75 inches) and temperature (<57 °F) are also important to egg development. These factors are related to flow volume during the spawning season (January through March), which is also dependent on a natural hydrograph. Weather conditions are also a major influence. Dams and diversions that impede normal flow should be removed if possible so that a more normal flow pattern and volume can be re-established.

Res5: Shallow riffles in permanent or intermittent streams provide rearing areas for fry and juveniles. The presence of these areas depends on channel composition and stream morphology, both of which can be altered by channelization and channel clearing. Floodplain and channel restoration may be necessary to re-create proper conditions.

Res6: To be good rearing habitat, riffles also need to have moderate flow rates of 0.12 to 0.37 m/s. Fish can use boulders and woody debris as velocity shelters to protect themselves from high flows resulting from heavy rains if these objects are present. Boulders and logs are often removed from streambeds by channel clearing, and woody debris is not replenished if the riparian vegetation is missing. Streambed and riparian restoration may be necessary to re-create velocity shelters.

Res7: Rearing habitat must have interstitial spaces in the substrate to provide shelter for juveniles. If the deposition rates of fine sediments are too high because of surface disturbances in the upper watershed, these spaces disappear. Restoration of the upper watershed is the appropriate mitigation.

Res8: Rearing habitat requires cool water temperatures (estimates of upper limits range from 17 to 25 °C). Weather conditions, flow rate, and shading all affect water temperature. The re-creation of a more natural hydrograph through dam removal and flow management and the restoration of riparian vegetation can help mitigate past management mistakes.

Res9: Rearing habitat also must have interstitial spaces in the substrate to provide habitat for the aquatic invertebrates that are food for juvenile steelhead. If the deposition rates of fine sediments are too high because of surface disturbances in the upper watershed, these spaces disappear. Restoration of the upper watershed is the appropriate mitigation.

Res10: Juveniles also eat terrestrial insects that fall into the water. The health and diversity of the riparian vegetation determines the number of terrestrial insects available, and riparian quality depends to some extent on weather but largely on the water table, flow regime, and floodplain condition. Destruction of riparian vegetation, the creation of dams and diversions, and channelization have modified these things in most Placer County streams, and appropriate restoration actions will be necessary.

Res11: Estuarine and ocean waters provide habitat for adult Central Valley steelhead. To access these areas from rearing habitat requires open channels and adequate flow; the management problems and mitigation actions for these conditions are the same as for path Res1.

Res12: Adult steelhead feed on crustaceans, squid, and fish. These organisms are plentiful in estuarine and oceanic ecosystems because of the high productivity that results from nutrient inputs coming from upwelling and fresh-water flows. However, excess inputs of nutrients and toxins from agricultural, industrial, and municipal runoff have deleterious effects on these ecosystems, which may affect steelhead populations negatively. Proper application of agricultural chemicals and limiting the inputs of industrial and municipal wastes can help restore appropriate nutrient cycles.

Hazards

Haz1: Elevated water temperatures (see path Res8) and low dissolved oxygen resulting from low flows are a hazard for eggs, fry, and juveniles. Low flows can be caused by drought conditions, but they are more likely to result from restricted and regulated streamflows because of dams and diversions. Dam removal and water management for a more natural flow regime can mitigate these problems to some extent.

Haz2: Predation on all life stages by non-native fishes is another hazard. These species thrive in elevated water temperatures resulting from low flows and lack of riparian vegetation. Low flows can be caused by drought conditions, but they are more likely to result from dams and diversions restricting and regulating streamflow. Loss of riparian vegetation results from clearing riparian areas for agriculture or flood control. Dam removal and water management for a more natural flow regime and riparian restoration can help mitigate these problems.

Haz3: Entrapment of out-migrating juvenile steelhead is another hazard. Entrapment results from unscreened or poorly screened water intakes on irrigation pumps or hydroelectric generators, and it can be mitigated by proper screening.

Haz4: Juvenile and adult steelhead can be stranded by low flows resulting from drought or dams that restrict and regulate streamflow. Dam removal and water management for a more natural flow regime can mitigate this problem to some extent.

Reproduction

Rep1: To reproduce, adults require access to spawning and rearing areas. Access depends on adequate flows from January to March; this is related to favorable weather, clear channels, and a natural hydrograph. Dams and diversions block the channel and change the hydrograph, and removing these structures or managing for a more normal flow regime is necessary to ensure access.

Rep2: Redds for egg laying are constructed by female steelhead in clean gravel at the heads of riffles. Interstitial spaces in the gravel are necessary to protect the eggs from predators, and the presence of these spaces depends on the deposition rate of fine sediments. If surface disturbances in the upper watershed result in excessive deposition of fine sediments, they must be mitigated by restoration.

Rep3: Successful reproduction also depends on water temperature (<57 °F) during January through March. Temperature is related to weather conditions and flow volume, and a natural hydrograph is important to maintaining adequate flows. Dams and diversions change the hydrograph, so removing these structures or managing for a more normal flow regime may be necessary to maintain suitable temperatures.

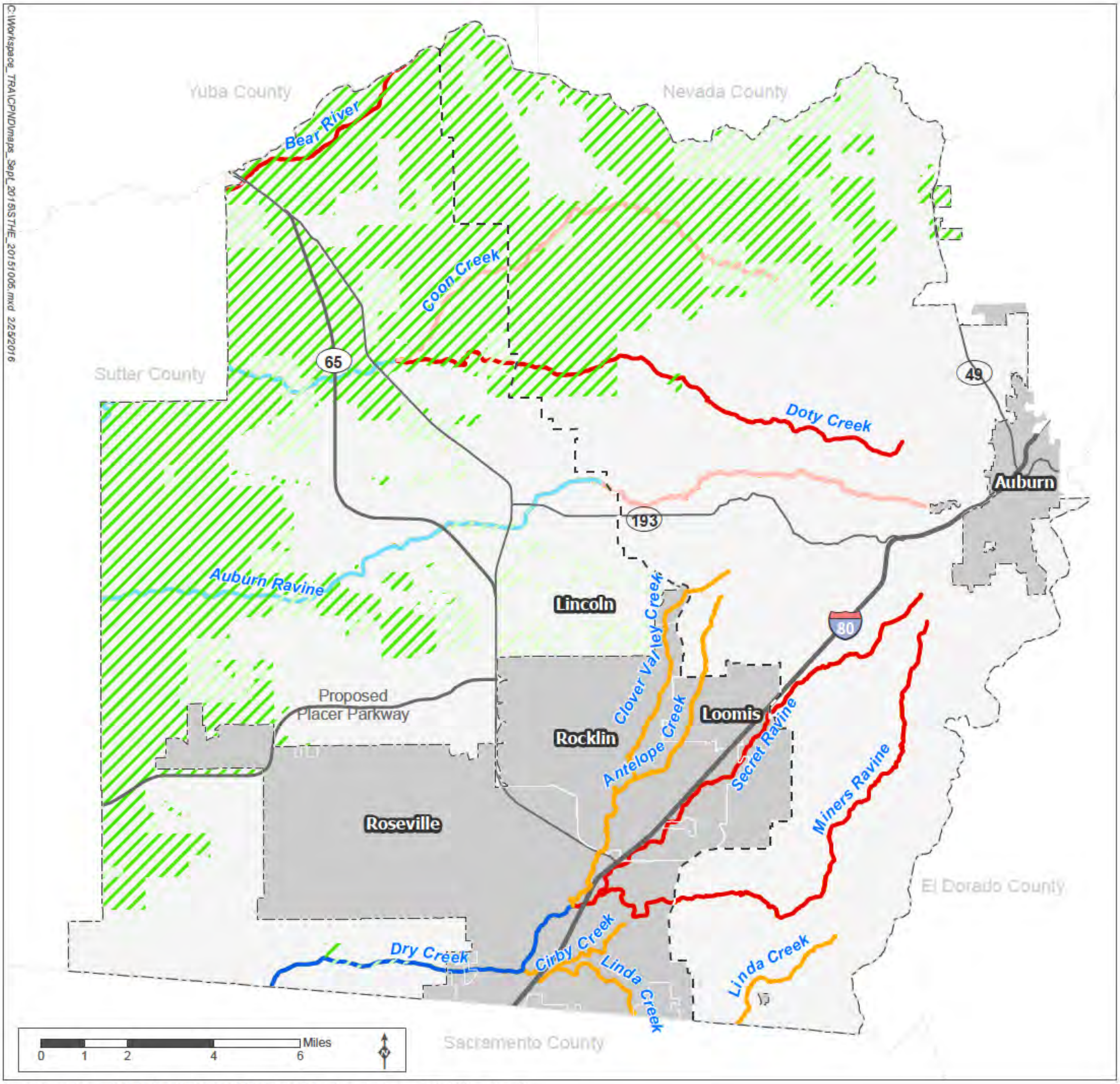
Migration

Mig1: The life history of the Central Valley steelhead depends on two types of migration, the in-migration of adults to the spawning areas and the out-migration of juveniles to estuarine and oceanic waters. The in-migration path is the same as path Res1.

Mig2: The out-migration of juveniles usually occurs during March and April and depends on weather conditions, adequate flow volumes, and clear channels. The latter two can be disrupted by dams and diversions, so removing those structures or managing for a more normal flow regime may be necessary for fish passage.

Summary

The reproductive biology of the Central Valley steelhead depends on a natural flow regime, proper substrate that is not choked with fine sediments, cool temperatures, and clear channels. Thus, for breeding populations of this species to remain in Placer County, a great deal of stream, riparian, and upper watershed restoration will be required. Placer County also may be contributing to the heavy nutrient and pollutant load in the Bay-Delta region, so attention paid to the proper use of agricultural chemicals and eliminating sources of industrial and municipal waste are also important.



Source: Placer County, 2014; MIG | TRA, 2015; NOAA NMFS, 2014; Dry Creek Conservancy 2003-2014; Bates 2015

Habitat Distribution - Steelhead Recovery Plan

- Spawning - Independent Population
- Rearing and/or Migration - Independent Population
- Spawning
- Rearing and/or Migration
- Spawning - Other Sources
- Modeled as Non-habitat
- Exis ing Protected Area
- Reserve Aquisition Area
- Non-participating City
- Major Road
- Valley/Foothill Divide
- Area A Boundary

MIG | TRA

Species Map 9.

Central Valley Steelhead Habitat Distribution

Placer County Conservation Program – Western Placer County HCP/NCCP

Central Valley Fall/Late Fall-Run Chinook Salmon – Evolutionary Significant Unit (*Oncorhynchus tshawytscha*)

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Status

Federal: Species of Concern (NMFS 2004); Magnuson-Stevens Act managed species

State: Species of Special Concern

Critical Habitat: Not Applicable (N/A)

Recovery Plan: N/A; however, recovery actions identified in the Recovery Plan for the Evolutionary Significant Units of Sacramento River Winter-run Chinook Salmon and Central Valley Spring-run Chinook Salmon and the Distinct Population Segment of Central Valley Steelhead (NMFS 2014) would likely also apply to the recovery of Central Valley fall/late fall-run Chinook salmon.

Distribution

North America

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

California

The Central Valley fall/late fall-run evolutionarily significant unit (ESU) includes fall-run and late fall-run Chinook salmon in the Sacramento and San Joaquin rivers and their tributaries east of Carquinez Strait (NMFS 1999). Historically, Chinook salmon were widely distributed throughout all major streams of the Central Valley drainage (Yoshiyama et al. 2001). The runs of Chinook salmon in California are differentiated by the maturity of fish entering fresh water, time of spawning migrations, spawning areas, incubation times, incubation requirements, and migration of juveniles (Moyle et al. 1995). The late fall-run Chinook salmon was identified as separate from the fall-run in the Sacramento River after the Red Bluff Diversion Dam was constructed in 1966 and fish counts could be more accurately made at the fish ladder in this location (Moyle et al. 1995). Central Valley fall-run Chinook salmon currently spawn in suitable habitat downstream of dams on every major tributary in the Sacramento and San Joaquin River system (Moyle et al. 2008). Late fall-run Chinook salmon spawning is limited to the mainstem and tributaries of the Sacramento River, and most spawning occurs in the reach between Red Bluff Diversion Dam and the Keswick Dam in Redding (Moyle et al. 2008).

Placer County Plan Area

Historical

The Bear River watershed comprises a small portion of northeastern Placer County, and is the second largest tributary to the Feather River. The Bear River historically hosted a "substantial" Chinook run (Reynolds et al. 1993 as cited in Yoshiyama et al. 2001). Adult salmon ascended as far as present day Camp Far West Reservoir, where a waterfall in that vicinity probably barred further passage (Yoshiyama et al. 2001). In addition, the American River watershed in Placer County was also known to have historically hosted fall-run Chinook (County of Placer 2013).

In the 1950's, there were up to a thousand Chinook salmon spawning in the Dry Creek system (Miners Ravine, Secret Ravine, Antelope Creek, Clover Valley Creek, Linda/Cirby Creek, and the main stem of Dry Creek), about 10 percent of which used Miners Ravine (Finlayson 1977 as cited in California Department of Water Resources 2002). A 1964 California Department of Fish and Game (CDFG[1]) memo summarized by Bailey (2003) estimated Secret Ravine, a tributary to Dry Creek, to have a run of 600 plus Chinook salmon. The oldest known record from Auburn Ravine was a CDFG report summarized by Bailey (2003), which estimated that the stream had a run of approximately 300 Chinook. Anecdotal observations from local residents, summarized by Bailey (2003), suggest Coon Creek also had a historic Chinook salmon run. Doty Ravine, a major tributary to Coon Creek, was known to have significant runs of Chinook salmon every fall (County of Placer 2013). In 1964, fall-run Chinook salmon were observed spawning and rearing in both Antelope Creek and Clover Valley Creek (CDFW 2015). The estimated run in Antelope Creek was 10 Chinook salmon (CDFW 2015). Pleasant Grove and Curry Creek are not believed to have historically hosted Chinook salmon runs, likely due to their intermittent nature (Bailey 2003).

Current

Central Valley fall/late fall-run Chinook salmon spawn and rear, or have potential to spawn and rear, in western Placer County streams, including Bear River, Coon Creek, Doty Ravine, Auburn Ravine, Dry Creek, Antelope Creek, Clover Valley Creek, Secret Ravine, and Miners Ravine (Jones and Stokes 2005). Bailey (2003), summarizing data from multiple sources including unpublished data held by the Region II Office of the California Department of Fish and Game, found native and hatchery-origin, fall-run Chinook to be present in the Coon Creek, Auburn Ravine, and Dry Creek Watersheds and absent from the Pleasant Grove and Curry Creek watersheds, likely due to their intermittent character. Juvenile, fall-run Chinook originating from the Feather River and Nimbus hatcheries are known to occur in the Coon Creek and Auburn Ravine watersheds (Bailey 2003). Juvenile fall-run Chinook from hatcheries on the Feather River have been stocked in the tributaries of Dry Creek (ECORP 2003). Fall-run Chinook salmon continue to be documented in Antelope Creek during an annual one-day salmon count coordinated by the Dry Creek Conservancy (CDFW 2015). In 2003, 44 live Chinook salmon and 7 carcasses were observed in Antelope Creek (CDFW 2015). The Bear River supports an occasional run of adult fall-run Chinook salmon in years when flows are sufficient to provide passage (Yoshiyama et al. 1996; County of Placer 2013). The American River watershed in Placer County no longer supports salmonids (County of Placer 2013).

[1] As of January 1, 2013, the California Department of Fish and Game (CDFG) was renamed the California Department of Fish and Wildlife. When this document cites reports prepared by the Department prior to 2013, the reference includes the prior department name of CDFG. Both CDFW and CDFG refer to the same agency.

As part of the Placer County Legacy Program, two concrete structures (i.e., Nevada Irrigation District Gaging Station in the City of Lincoln and the Nevada Irrigation District Hemphill Dam in Placer County) impeding salmon movement in the Auburn Ravine watershed have been modified to allow fish passage (County of Placer 2013). With the successful modification of the Nevada Irrigation District Gaging Station, nearly 300 Chinook salmon ascended the structure in November and December 2012 (County of Placer 2013). In response to Chinook salmon observations upstream of the Nevada Irrigation District Gaging Station, a rotary screw trap was deployed in Auburn Ravine at the Aitken Ranch site in January 2013. In April 2013, twenty five juvenile fall-run Chinook salmon were collected in Auburn Ravine at Hemphill Dam approximately 8 miles upstream of the rotary screw trap (CDFW 2014). Additional fall-run size Chinook salmon were also captured at the rotary screw trap location (CDFW 2014).

Chinook salmon were also found at the Hidden Falls Park after new gravel was placed as part of the construction of a new bridge over Coon Creek (County of Placer 2013). Additional fall-run sized Chinook salmon were observed in Coon Creek near McCourtney Road in May 2015 (Haas pers. comm.).

Population Status & Trends

California

The historic abundance of fall-run Chinook is hard to ascertain because they were heavily fished in the 19th century, hydraulic mining debris buried major spawning and rearing areas, and estimates are inaccurate due to poor record keeping (Moyle et al. 2008). The most abundant populations of fall-run Chinook salmon occur in the Sacramento, Feather, Yuba, and American Rivers (Mills and Fisher 1994). The ESU also occurs in smaller tributaries of the Sacramento River and in tributaries of the San Joaquin River. Fall-run Chinook salmon have a relatively large hatchery component, averaging more than 25,000 adults. Natural spawners average about 200,000 adults for the Sacramento and San Joaquin systems (Moyle 2002). In 1992 to 2005, the run averaged about 450,000 fish per year, although it dropped to less than 200,000 fish in 2006 and to about 90,000 spawners in 2007 (Moyle et al. 2008). Yoshiyama et al. (2001) calculated that approximately 72% of the historic spawning and holding habitat in the Central Valley drainage is no longer available.

In 2008, approximately 66,200 Sacramento River fall-run Chinook adults returned to spawn in the Sacramento River Basin. This is the lowest return of Sacramento River fall Chinook on record and is well below the annual conservation objective of 122,000-180,000 adult spawners set by the Pacific Fishery Management Council's Salmon Fishery Management Plan (Pacific Fishery Management Council 2010). A National Marine Fisheries Service (NMFS) working group found poor ocean conditions to be the proximate cause of Chinook population declines (NMFS 2009). This is based on evidence of normal juvenile recruitment rates prior to ocean entry and known poor ocean conditions upon entry such as weak upwelling, warm sea temperatures, and low densities of prey items (NMFS 2009). Although the NMFS group points to ocean conditions as the reason for recent significant declines, it acknowledge that decades of freshwater and estuarine habitat degradation along with hatchery production has created a population that has little fitness or resiliency to withstand natural stochastic events (NMFS 2009).

Historic abundance of late fall-run Chinook salmon is not known because it was recognized as distinct from fall-run Chinook only after Red Bluff Diversion Dam was constructed in 1966 (Moyle et al. 1995). Late fall-run Chinook are one of the least numerous runs in the Sacramento River (Moyle et al. 1995). During 1967 to 1976, the run averaged about 22,009 fish annually. Between 1982 and 1991, the run averaged 9,700 fish annually. During 1992 to 2007 the run averaged 21,000 fish (Moyle et al. 2008). The population today is likely partly sustained by hatchery production at the Coleman National Fish Hatchery on Battle Creek.

Placer County Plan Area

The most current occurrence data are from the Dry Creek watershed. With six years of winter data (2003 – 2008) from one-day surveys performed generally in mid-November, the Dry Creek Conservancy has counted live and moribund adult Chinook salmon, and the presence of redds, in the main stem of Dry Creek as well as all of its major tributaries: Linda Creek, Cirby Creek, Antelope Creek, Miners Ravine, and Secret Ravine (Dry Creek Conservancy 2009; Gregg Bates pers. comm. 2015).

A 2004 – 2005 fish community survey was performed by the California Department of Fish and Game (2008) throughout the main stems of Auburn Ravine (seven sampling locations) and Coon Creek (seven sampling locations) in western Placer County. Multiple-pass, depletion electrofishing methods were applied in November and December of 2004 and again in April of 2005 (CDFG 2008). Because juvenile Chinook are expected only in the spring, just the April 2005 data are presented here. One juvenile Chinook salmon was found in Auburn Ravine (at the Catlett Road crossing site) with a catch per unit effort of 0.09 fish / hour (total effort was 11.03 hours). In Coon Creek, 25 juvenile Chinook were collected exclusively from the Gladding Road and Garden Bar Road crossing sites, with a catch per unit effort of 3.99 fish / hour (total effort was 6.26 hours). Additionally, three adult Chinook salmon were observed spawning at the Gladding Road site in December 2004 (CDFG 2008).

In Bear River, the fall run occurs only occasionally when heavy rains and dam spillage provide adequate flows (Reynolds et al. 1993, as cited in Yoshiyama et al. 2001). At these times, the run may number in the "hundreds" (Reynolds et al. 1993, as cited in Yoshiyama et al. 2001).

The California Department of Fish and Game has conducted periodic adult Chinook salmon surveys in Dry Creek at least as far back as 1963, primarily upstream of the confluences with Secret and Miners ravines (ECORP 2003). The fall-run adult Chinook salmon population in the Dry Creek watershed was estimated to be just over 1,000 in 1964, with the majority of spawning occurring in Secret and Miners ravines. Since the late 1990's, adult Chinook salmon populations in Secret Ravine have averaged about 160 fish per year (ECORP 2003). From 1997 to 2002, outmigrating juvenile accounts from Secret Ravine averaged approximately 15,000 per year (Ayres et al. 2003).

Although there are not enough quantitative data to estimate population sizes, historical evidence summarized by Bailey (2003) provides evidence for the existence of a continued Chinook and steelhead run in Auburn Ravine. Additionally, anecdotal evidence presented in Bailey (2003) suggests the existence of "half-pounders" in Auburn Ravine; smaller, but sexually mature males and females that return from the ocean after just one year. Although there was insufficient data to support conclusions about the status of fish populations in Antelope Creek, fall-run Chinook salmon have been documented spawning in Antelope Creek over the last 40 years; therefore, fall-run Chinook are believed to persist in the creek (Bailey 2003). Even less fisheries data are available for Coon Creek and Miners Ravine, however, both fall-run Chinook and steelhead runs are believed to persist in the watershed (County of Placer 2002; Bailey 2003). Fall-run and spring-run Chinook were stocked in Doty Ravine three times in the mid-1980s and data suggests that fall-run Chinook salmon do use the stream for spawning in certain years (Bailey 2003).

One-day winter counts of live and moribund Chinook adults and redds performed by the Dry Creek Conservancy (2009) indicate a negative trend in all Dry Creek watershed tributaries surveyed (Miners

Ravine, Secret Ravine, Antelope Creek, Linda/Cirby Creek, and the main stem of Dry Creek). Each year between 2003 and 2008, almost without exception, fewer adults and redds are observed in each of the Dry Creek watershed's main reaches (Dry Creek Conservancy 2009). Although there aren't enough data from other watersheds in western Placer County to confirm this trend, it is certainly likely to be the case given the current status of all Sacramento River fall-run Chinook populations (see discussion in California Status and Trends section above). Factors contributing to the decline of Chinook salmon in Secret Ravine are thought to include increased sediment, altered flow regimes, reduced access to habitat, and toxicity (Ayres et al. 2003).

As part of the Placer County Legacy Program, two concrete structures (i.e., Nevada Irrigation District Gaging Station in the City of Lincoln and the Nevada Irrigation District Hemphill Dam in Placer County) impeding salmon movement in the Auburn Ravine watershed have been modified to allow fish passage (County of Placer 2013). With the successful modification of the Nevada Irrigation District Gaging Station, nearly 300 Chinook salmon ascended the structure in November and December 2012 (County of Placer 2013). In response to Chinook salmon observations upstream of the Nevada Irrigation District Gaging Station, a rotary screw trap was deployed in Auburn Ravine at the Aitken Ranch site in January 2013. In April 2013, twenty five juvenile fall-run Chinook salmon were collected in Auburn Ravine at Hemphill Dam approximately 8 miles upstream of the rotary screw trap (CDFW 2014). Additional fall-run size Chinook salmon were also captured at the rotary screw trap location (CDFW 2014).

Chinook salmon were also found at the Hidden Falls Park after new gravel was placed as part of the construction of a new bridge over Coon Creek (County of Placer 2013). Additional fall-run sized Chinook salmon were observed in Coon Creek near McCourtney Road in May 2015 (Haas pers. comm.).

Natural History

The habitat requirements, ecological relationships, life history, and threats to Central Valley fall/late fall-run Chinook salmon described below are summarized in diagram form in the Envirogram 10 Fall-Run Chinook Salmon.

Habitat Requirements

Chinook salmon depends on suitable water temperature and substrate for successful spawning and incubation. Although the suitability of gravel substrates for spawning depends largely on fish size, a number of studies have determined substrate sizes that represent the most suitable conditions. Generally, Chinook salmon require substrates of 0.1–5.9 inches, whereas steelhead prefer substrate no larger than 3.9 inches (Bjornn and Reiser 1991).

The quality of spawning habitat is also correlated with intra-gravel flow. Low intra-gravel flow may provide insufficient dissolved oxygen, contribute to growth of fungus and bacteria, and result in high levels of metabolic waste. High percentage of fines in gravel substrates can substantially limit intra-gravel flow, affecting the amount of spawning gravel available in the river (Healey 1991). Raleigh et al. (1986) concluded that optimal gravel conditions would include less than 5–10% fine sediments measuring 0.12 inch or less in diameter. In addition, alevins of Chinook salmon, steelhead, and coho salmon have been observed in laboratory studies to have difficulty emerging when gravels exceeded 30–40% fine sediments (Phillips et al. 1975 as cited in Bjornn and Reiser 1991; Waters 1995).

Water depth is one factor affecting spawning gravel selection (Raleigh et al. 1986; Bjornn and Reiser 1991). Minimum water depths at redd areas (i.e., gravel nests) vary with fish size and water velocity, because these variables affect the depth necessary for successful digging (Healey 1991). In general, water should be at least deep enough to cover the fish during spawning. Burner (1951, as cited in Healey 1991 and Bjornn and Reiser 1991) observed Chinook salmon spawning in water as shallow as 0.16 foot; Vronski (1972 as cited in Healey 1991) found Chinook salmon spawning in water depths of 23.6 feet. Thompson (1972, as cited in Bjornn and Reiser 1991), who also studied water depth requirements for spawning, found Chinook salmon spawning in depths less than 0.8 foot.

Flow velocity also affects spawning gravel selection; however, the range in water depth and velocity is very broad (Healey 1991). Healey found water velocities of 0.98–6.2 feet/second reported in the literature. Studies in northern California found that Chinook salmon from the Yuba and Sacramento rivers preferred velocities of 1.55–2.95 feet/second and 0.9–2.7 feet/second respectively (CDFG 1991).

Survival of Chinook salmon eggs and larvae during incubation declines as water temperatures increase to 53.6–60.8ºF (Myrick and Cech 2001).

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

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

Instream and overhead cover, in the form of undercut banks, downed trees, and overhanging ranches is important for juvenile rearing. Streamside riparian vegetation is a primary source of cover. The root systems of riparian vegetation and large organic debris (e.g., fallen logs) in the stream channel provide refuge from predators and high flow conditions (Jones and Stokes 2005).

Survival of juvenile Chinook salmon declines as water temperatures increase to 64.4–75.2ºF (Myrick and Cech 2001; Rich 1987). Juveniles require cooler water temperature to complete the parr-smolt transformation and to maximize their saltwater survival. Successful smolt transformation deteriorates at temperatures of 62.6–73.4ºF (Marine 1997 as cited in Myrick and Cech 2001).

Freshwater migration corridors include river channels, channels through the Sacramento-San Joaquin Delta, and the Bay-Delta estuary. Migration corridors should be generally free from obstructions (passage barriers and impediments to migration), have favorable water quality, and contain natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

Reproduction

Central Valley fall-run Chinook salmon spawns from late September to December, with peak spawning taking place during late October and November when water temperatures decrease (Moyle 2002). Fall-run Chinook salmon spawns over gravel (redds) soon after arriving at the spawning grounds. Egg incubation for fall-run Chinook salmon begins in September and can extend to March (Vogel and Marine 1991 as cited in Jones and Stokes 2005). Juvenile fish remain in redds from about 32 days at 61°F to 159 days at 37°F (Healey 1991). Central Valley late fall-run Chinook spawn from December to April, with peak spawning taking place during February and March. Late-fall run Chinook do not feed while migrating and holding in the river, and instead rely on stored body fat reserves for maintenance (Moyle et al. 1995). Egg incubation for late fall-run Chinook salmon occurs from December through June (Vogel and Marine 1991 as cited in Jones and Stokes 2005).

Dispersal Patterns

After emerging from gravel, juvenile Chinook salmon moves downstream, mostly at night. It rears in the mainstem rivers or the Delta before migrating to the ocean.

Longevity

Chinook salmon generally matures at three to four years and can reach five to eight years (Healey 1991). A minority of individuals return to the river as sexually mature two-year-olds (grilse).

Sources of Mortality

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

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

Behavior

Chinook salmon are anadromous (i.e., they migrate from the marine environment into freshwater rivers and stream of their birth) and semelparous (i.e., they spawn and die in the freshwater streams of their birth) (NMFS 1999). While in streams, Chinook salmon is an opportunistic feeder and varies its diet according to seasonal availability. In the summer months, it feeds primarily on drifting aquatic invertebrates, terrestrial insects, and active bottom invertebrates. Individual fish, however, do not usually feed on the full range of food available. Larger fish tend to eat larger prey. Feeding can occur any time of day, but most activity occurs around dusk (Moyle 2002).

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

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

Movement and Migratory Patterns

Fall-run Chinook salmon migrates from the Pacific Ocean to Central Valley rivers from approximately July to December. Within western Placer County stream, migration is dependent on adequate flows and suitable water temperatures, which usually occur following storm events in October or November (Jones and Stokes 2005). Peak spawning for fall-run spawning fish occurs during late October and November, as water cools. Juvenile fall-run Chinook salmon start emigrating towards the Pacific Ocean from January through June, shortly after emerging from the redds. Within western Placer County stream, juvenile Chinook salmon tend to migrate from February through June, with peak migration occurring from March to May (ECORPS 2003). Late fall-run Chinook salmon migrate from the Pacific Ocean to Central Valley rivers from approximately mid-October through mid-April. Peak spawning for late fall-run Chinook salmon occurs in February and March. Juvenile late fall-run Chinook salmon start emigrating toward the Pacific Ocean from April to December, with the primary movement occurring in the winter months. Central Valley Chinook salmon enter the ocean near the Gulf of the Farallones and then distribute north and south along the continental shelf, mostly between Point Conception and Washington (Healey 1991). Chinook salmon migration from freshwater habitats to the ocean may be as long as 373 miles, transiting many different habitats, all with varying natural conditions (Michel et al. 2012).

During a study by Michel et al. (2012) smolt movement rates were found to vary substantially throughout the watershed. The fastest movement rates were seen in the river regions, with the Upper Sacramento River having the fastest rates in the study, potentially due to the faster water velocities which allowed for faster passive transport of actively migrating smolts. The slowest movement rates were seen in the Sacramento-San Joaquin River Delta, which is a highly modified and complex system of sloughs and channels. Smolts entering this region may enter the interior delta, predisposing them to longer routes, higher predation, and risk of entrainment into water pumps, which inevitably leads to higher mortality rates (Perry et al. 2010). Michel et al. (2012) also found that river width-to-depth ratio had a negative relationship with movement rates (i.e., smolts were found to move slower through wider, shallower reaches), flow was positively related to movement rates, and turbidity had a positive relationship with movement rates (perhaps because turbidity dramatically decreases predator efficiency).

Ecological Relationships

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

Threats

Degradation and loss of habitat have contributed substantially to the decline of Chinook salmon. Shasta and other dams blocked access to historic spawning and rearing habitat, as it did in the case of steelhead. Zueg et al. (2010) found that extirpation of fall-run Chinook salmon were best predicted by habitat loss and migration barriers. Other factors affecting abundance include modifications of water temperatures that result from reservoir operations, harvest, entrainment in diversions, contaminants, predation by nonnative species, and interaction with hatchery stock (U.S. Army Corps of Engineers 2000).

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

Smolt mortality is likely a factor affecting fall-/late fall-run Chinook. Small numbers of outmigrants are presumably entrained at every irrigation diversion along the Sacramento River that is operating during the migration period (Moyle et al. 1995). In addition, extensive bank alteration along the migration path reduces the amount of cover available to protect outmigrants from predators (Moyle et al. 1995). Predation on juvenile salmon by nonnative fish has been identified as an important threat to fall- and late fall-run Chinook salmon in areas with high densities of nonnative fish that prey on out-migrating juvenile salmon (Lindley and Mohr 2003).In the Delta, flows drawn through the Delta Cross Channel (DCC) and Georgiana Slough transport a proportion of migrants into the central Delta. The number of juveniles entering the DCC and Georgiana Slough is assumed to be proportional to the volume of flow diverted from the Sacramento River (CDFG 1987). Survival of juvenile Chinook salmon drawn into the central Delta is lower than survival of juvenile Chinook salmon remaining in the Sacramento River channel.

Diversions in the Central Valley associated with the State Water Project and the federal Central Valley Project in the south Delta entrain large numbers of Chinook salmon. The diversions are screened and salmon are “salvaged” from the projects by capturing, trucking, and then releasing them downstream in the Delta; however, both direct (e.g., predation and stress from salvage) and indirect mortality (e.g., changes in hydrology) is likely high due to entrainment associated with these diversions (Moyle et al. 2008).

Artificial propagation programs (i.e., hatchery production) for fall- and late fall-run Chinook salmon in the Central Valley likely present multiple threats to wild Chinook salmon populations, including genetic introgression by hatchery origin fish that spawn naturally and interbreed with local wild populations (NMFS 2014). Interbreeding with hatchery fish may contribute to reduced genetic diversity and introduce maladaptive genetic changes to the wild population.

Context for a Regional Conservation Strategy

Central Valley fall/late fall-run Chinook salmon spawn and rear, or have potential to spawn and rear, in western Placer County, including Bear River, Antelope Creek, Clover Valley Creek, Miners Ravine, Secret Ravine, tributaries to Dry Creek, Coon Creek, Linda Creek, Cirby Creek, Auburn Ravine, and Doty Ravine. The Placer County populations are part of the state’s most abundant fall/late fall-run of Chinook salmon, which occur through the Sacramento, Feather, Yuba, and American rivers. The Plan Area supports habitat for spawning and juvenile salmon. Stressors to Chinook salmon in the Plan Area include passage impediments/barriers affecting adult migration and spawning, low flow conditions, limited instream gravel supply, water temperature and water quality issues from agricultural and urban runoff, loss of riparian habitat and instream cover, and predation (NMFS 2014). Due to the need for unimpeded access from spawning sites to the ocean, maintenance of migration routes and habitat in the Plan Area has great bearing on the California population throughout downstream riverine systems.

Modeled Species Distribution in the Plan Area

Model Assumptions

Spawning and Rearing Habitat

Modeled spawning and rearing habitat for Central Valley fall/late fall-run Chinook salmon includes riverine, urban riparian, and valley foothill riparian land-cover types in the following river reaches: Bear River, Coon Creek upstream of Gladding Road; Doty Creek (a tributary to Coon Creek) upstream of Crosby Herold Road; Auburn Ravine east of the Highway 65 bridge in the City of Lincoln; and the entirety of Secret and Miners Ravine in the Dry Creek watershed, as well as Linda, Cirby, Clover Valley, and Antelope creeks.

Migration and Rearing Habitat

Modeled migration and rearing habitat for Central Valley fall/late fall-run Chinook salmon includes riverine and valley foothill riparian land-cover types in the following river reaches: Bear River between the western border of the County and Camp Far West Reservoir; the main stem of Coon Creek downstream of the Gladding Road Crossing to the western border of the County; Doty Creek downstream of the Crosby Herold Road crossing; Auburn Ravine downstream of the Highway 65 bridge in the City of Lincoln to the western border of the County, and the main stem of Dry Creek downstream of the confluence with Miners and Secret Ravine to the southern border of the County.

Rationale

Central Valley fall/late fall-run Chinook salmon spawn in stream reaches with flows generally between 7 to 61 in/s (Moyle 2002), depths between 4 and 59 in (Moyle 2002), and temperatures between 30 °F and 55 °F (CDFG 2000, as cited in NMFS 2009). Central Valley fall/late fall-run Chinook salmon typically spawn in sediments dominated by gravels at the tail of pools or in riffles. Gravel-sized sediment is small enough to be moved by the digging action of an adult female Chinook salmon but coarse enough to provide adequate intergravel flow, and therefore oxygenated water, to incubating eggs and aelvin. Pool tails and riffles are those habitat types that provide the best conditions for intergravel flow. In western Placer County streams, these characteristics are found primarily in the upper reaches of the watersheds.

Juvenile Chinook salmon migrate downstream during their first year or two of life, and rearing occurs throughout the watershed. Juvenile Chinook salmon are primarily found in stream reaches dominated by runs, riffles, and pools and/or characterized by complex habitat such as undercut banks, large woody debris, and an intact riparian canopy.

Migration and holding habitat are those corridors through which adult and juvenile Central Valley fall/late fall-run Chinook salmon must travel to get to spawning and rearing grounds. In western Placer County, migration habitat includes the canals, culverts, diversion dams, and hardened stream reaches characteristic of highly urbanized areas.

Rearing habitat in the lower stream reaches of western Placer County is limited to those areas that were able to maintain those remnant habitat characteristics described above. In addition, Central Valley fall/late fall-run Chinook salmon are also known to take advantage of urbanized stream features such as pools associated with water diversion and flood control structures.

This habitat model uses the spawning, migration, and rearing habitat defined for Central Valley steelhead in the Recovery Plan for Central Valley steelhead (NMFS 2014) for Central Valley fall/late fall-run Chinook salmon. Life history requirements are similar enough between these two species to generalize the application of modeled habitat for Central Valley Steelhead to Central Valley fall/late fall-run Chinook salmon at the level of scale and precision of this habitat model. Occurrence data for Central Valley fall/late fall-run Chinook salmon used to develop this model were generally consistent with the Central Valley steelhead model. Occurrence data are from Dry Creek Conservancy (2009), California Department of Fish and Game (2008), County of Placer (2002), Bailey (2003), ECORP Consulting (2003), and Reynolds et al. (1993, as cited in Yoshiyama et al. 2001).

Model Results

Species Map 10. Central Valley Fall/Late Fall-run Chinook Salmon *Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for Chinook Salmon in the Plan area.

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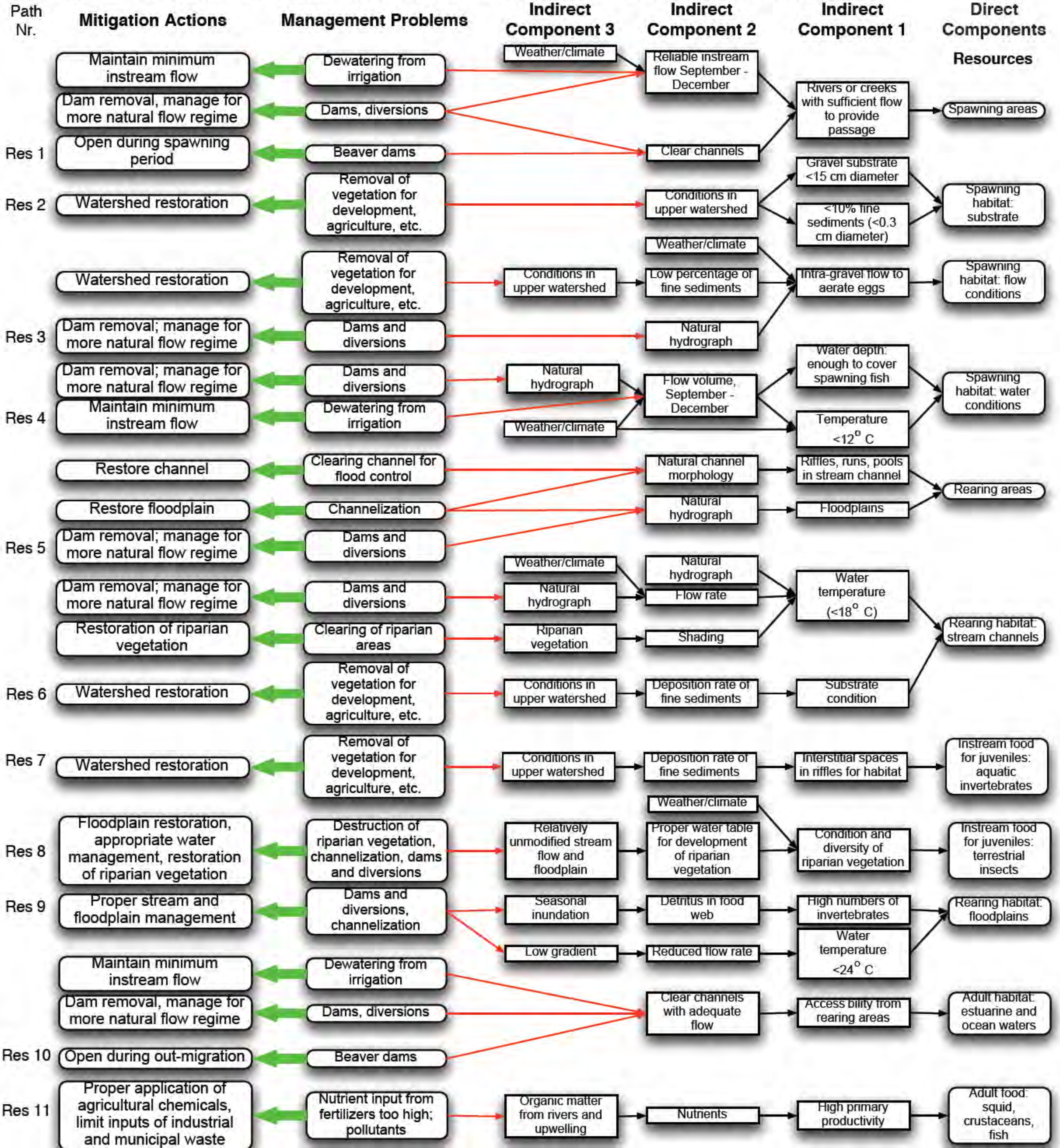
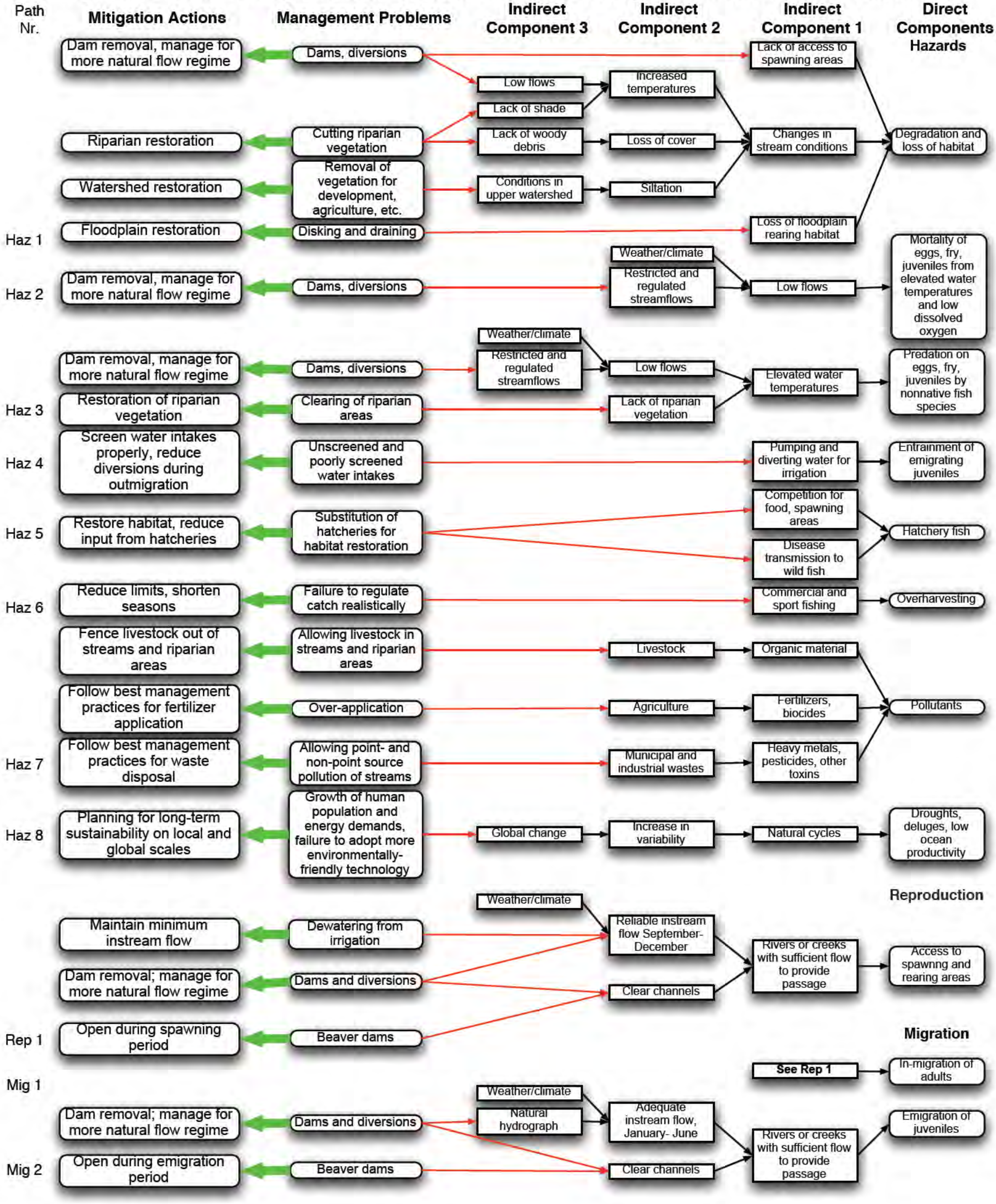
Envirogram 10 Central Valley Fall-Run Chinook Salmon, *Oncorhynchus tshawytscha*



Envirogram 10 Central Valley Fall/Late Fall Run Chinook Salmon. Res = Resources; Haz = Hazards; Rep = Reproduction; Mig = Migration.

Envirogram 10 Central Valley Fall-Run Chinook Salmon, *Oncorhynchus tshawtscha* (continued 2)



Envirogram Narrative

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

The envirogram was created based on the information provided in the species account. The envirogram is a tool to depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Spawning areas for the Central Valley fall/late fall-run Chinook salmon are in tributaries of the Sacramento River. The fish enter these streams from the mainstem river, requiring clear channels with adequate flow for access, particularly from September to December. Dams and diversions, dewatering from irrigation, and beaver dams often make this access problematic. Dam removal, maintaining minimum instream flow, managing for a more natural flow regime, and opening up beaver dams in spawning streams improve access. None of these options is without cost; for example, beaver dams are very useful for restoring floodplains and wetlands.

Res2: Central Valley fall/late fall-run Chinook salmon have specific requirements for spawning substrate. The gravel needs to be smaller than 5.9 inches in diameter, and fine sediments (<0.1 inch in diameter) need to be less than 10% of the substrate. Proper substrate conditions depend to a large extent on conditions in the upper watershed; sedimentation resulting from logging, development, agriculture, or other activities degrades spawning areas. Watershed restoration can help mitigate this problem.

Res3: Water flow through the gravel substrate is important to egg development, and its extent depends on weather, the amount of fine sediments in the substrate (< ca. 10%), and on a natural hydrograph (flow regime). These in turn depend on stream characteristics (see path Res1) and the shape of the upper watershed (path Res2).

Res4: Water depth sufficient to cover spawning fish and temperature (<54 °F) are important to egg deposition and development. These factors are related to flow volume during the spawning season (September through December), which is also dependent on a natural hydrograph. Weather conditions are also a major influence. Dams and diversions that impede normal flow should be removed if possible so that a more normal flow pattern and volume can be re-established.

Res5: Riffles, runs, and pools in stream channels and floodplains provide rearing areas for fry and juveniles. The former depends on natural stream morphology, which often has been altered by channelization and channel clearing. Floodplains also depend upon natural stream morphology and a natural hydrograph unimpeded by dams and diversions. Floodplain and channel restoration may be necessary to re-create proper conditions.

Res6: For stream channels to provide good rearing habitat, water temperatures have to be less than 64 °F and substrates need to have few fine sediments. Water temperature is a function of flow rate and shading. Flow rate depends on weather and a natural hydrograph; shading is provided by riparian vegetation that is often lost during clearing for agriculture or development. High deposition rates of fine sediments result from surface disturbances in the upper watershed. Restoration of the upper watershed, flow regime, and riparian vegetation are necessary to mitigate these problems.

Res7: Aquatic invertebrates are important food for juveniles. These organisms depend on interstitial spaces in riffles for shelter and feeding habitat, and the presence of these spaces depends on a low deposition rate of fine sediments. High deposition rates of fine sediments result from surface disturbances in the upper watershed. Restoration of the upper watershed is necessary to mitigate this problem.

Res8: Terrestrial insects are also an important food for juveniles. The diversity and abundance of these organisms depends on healthy riparian vegetation, which in turn depends on weather and a proper water table—related to the hydrograph and the condition of the floodplain. Dams and diversions, channelization, and cutting of riparian vegetation should be mitigated by better water management and by floodplain and riparian restoration.

Res9: Floodplains provide better rearing habitat for juvenile salmon than stream channels. Water temperatures can be toward the high end for survival (<75 °F), but these temperatures, along with a large number of invertebrates for food, result in higher growth rates. Water temperatures are higher because of lower gradients and flow rates, and detritus deposited during seasonal inundation provides food for the invertebrates. The creation of dams and diversions and channelization result in loss of floodplains. These practices must be replaced by proper stream and floodplain management if this important habitat component is to be restored in Placer County.

Res10: Estuarine and ocean waters provide habitat for adult Central Valley fall/late fall-run Chinook salmon. To access these areas from rearing habitat requires open channels and adequate flow; the management problems and mitigation actions for these conditions are the same as for path Res1.

Res11: Adult Central Valley fall/late fall-run Chinook salmon feed on crustaceans, squid, and fish. These organisms are plentiful in estuarine and oceanic ecosystems because of the high productivity that results from nutrient inputs coming from upwelling and fresh-water flows. However, excess inputs of nutrients and toxins from agricultural, industrial, and municipal runoff have deleterious effects on these ecosystems, which may negatively affect Central Valley fall/late fall-run Chinook salmon populations. Proper application of agricultural chemicals and limiting the inputs of industrial and municipal wastes can help restore appropriate nutrient cycles.

Hazards

Haz1: The degradation and loss of habitat is probably the greatest hazard faced by the Central Valley fall/late fall-run Chinook salmon. Habitat problems include lack of access to spawning areas, changes in stream conditions, and loss of floodplain rearing habitat. Dams and diversions affect access directly and also result in low flows that lead to elevated temperatures. Cutting riparian vegetation results in loss of shade and increased temperatures along with the loss of woody debris that provides cover for the fish. Removal of vegetation in the upper watershed results in siltation, and disking and draining removes floodplain rearing habitat. Dam removal, managing for more natural flows, and riparian, watershed, and floodplain restoration are necessary to mitigate these problems.

Haz2: Elevated water temperatures (see path Res6) and low dissolved oxygen resulting from low flows are a hazard for eggs, fry, and juveniles. Low flows can be caused by drought conditions, but they also result from restricted and regulated streamflows caused by dams and diversions. Dam removal and water management for a more natural flow regime can mitigate these problems to some extent.

Haz3: Predation on all life stages by non-native fishes (primarily centrarchids) is another hazard. These introduced species thrive in elevated water temperatures resulting from low flows and lack of riparian vegetation. Low flows can be caused by drought conditions, but they are more likely to result from dams and diversions restricting and regulating streamflow. Loss of riparian vegetation results from clearing riparian areas for agriculture or flood control. Dam removal and water management for a more natural flow regime and riparian restoration can help mitigate these problems.

Haz4: Entrainment of emigrating juvenile Central Valley fall/late fall-run Chinook salmon is another hazard. Entrainment results from unscreened or poorly screened water intakes on irrigation pumps or hydroelectric generators, and it can be partially mitigated by proper screening.

Haz5: Hatchery-raised fish can compete for food and spawning areas and transmit diseases to native salmon. There has been a general tendency in salmon management to substitute hatcheries for habitat restoration; there needs to be a plan that coordinates the two.

Haz6: Overharvesting by commercial and sport fishing, resulting from unrealistic limits, is another hazard. Reduced limits and shortened seasons based on science instead of politics is the only way to deal with this problem.

Haz7: Pollutants, in the form of organic material from livestock, fertilizers and pesticides from agriculture, and heavy metals, pesticides, and other toxins from municipal and industrial wastes, are yet another hazard faced by Central Valley fall/late fall-run Chinook salmon. Livestock should be kept out of streams and riparian zones, particularly during spawning season and when juveniles are present, and over- application of fertilizers and pesticides and other point-source and non point-source pollution can be mitigated to some extent by following best management practices.

Haz8: Natural events, such as droughts, floods, and periods of low ocean productivity have been hazards to Central Valley fall/late fall-run Chinook salmon for millennia. These natural cycles are evidently increasing in both severity and variability because of global changes that result from the growth of the human population, increasing energy demands, and a failure to adopt less environmentally damaging technology. Planning for sustainability at both local and global scales is necessary to ensure that salmon will continue to ascend Placer County's streams.

Reproduction

Rep1: To reproduce, adults require access to spawning and rearing areas. Access depends on adequate flows and clear channels from September to December. Although weather patterns have some influence on these factors, dams and diversions, pumping for irrigation, and beaver dams block the channel and change the hydrograph. Removing these structures or managing for a more normal flow regime and minimum instream flow are necessary to ensure access. Beaver dams impede access, but they also are very useful for restoring floodplains and wetlands. Thus, a beaver management plan must be developed for salmon spawning streams. Other requirements for reproduction are spelled out in Resources.

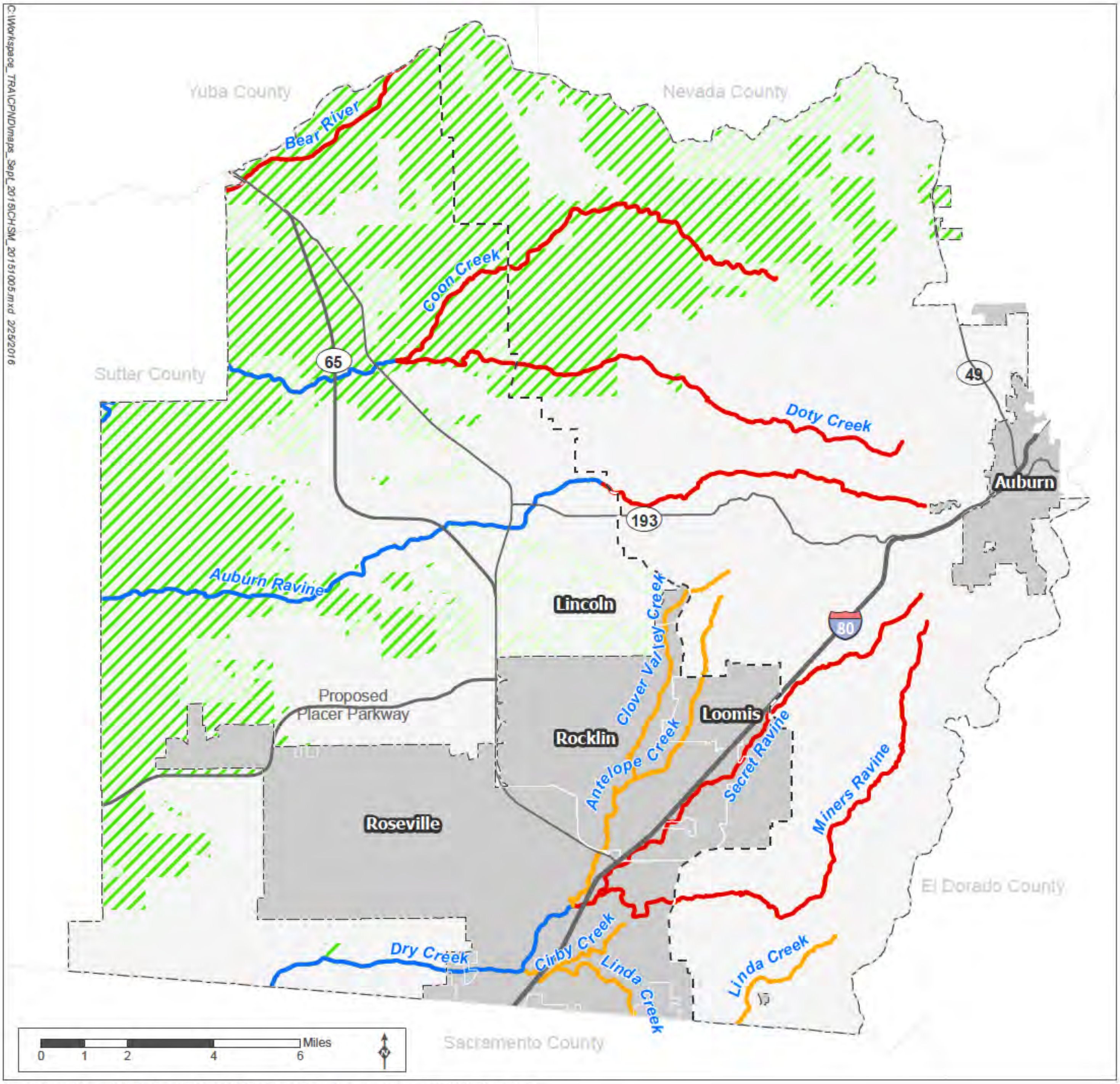
Migration

Mig1: The life history of the Central Valley fall/late fall-run Chinook salmon depends on two types of migration, the immigration of adults to the spawning areas and the emigration of juveniles to estuarine and oceanic waters. The immigration path is the same as path Res1.

Mig2: The emigration of juveniles usually occurs from January to June and depends on weather conditions, adequate flow volumes, and clear channels. The latter two can be disrupted by dams and diversions, so removing those structures or managing for a more normal flow regime may be necessary for fish passage. See path Rep1 for comments on beaver dams.

Summary

The reproductive biology of the Central Valley fall/late fall-run Chinook salmon depends on access to spawning areas, a natural flow regime, proper substrate that is not choked with fine sediments in stream channels, and functional flood plains. Thus, for Placer County to contribute to the recovery of this species, a great deal of stream, riparian, and upper watershed restoration will be required. Success is not guaranteed, however. Events in the Sacramento River, the Delta, and the Pacific Ocean—as well as environmental change on a global scale—also are contributing to the species' decline. Both upstream and downstream conditions must be addressed simultaneously.



Source: Placer County, 2014; MIG | TRA, 2015; NOAA NMFS, 2014; Dry Creek Conservancy 2003-2014; CDFW 2015

Habitat Distribution - Steelhead Recovery Plan

- Spawning
- Rearing and/or Migration
- Spawning - Other Sources
- Non-habitat
- Existing Protected Area
- Reserve Aquisition Area
- Non-participating City
- Major Road
- Valley/Foothill Divide
- Area A Boundary

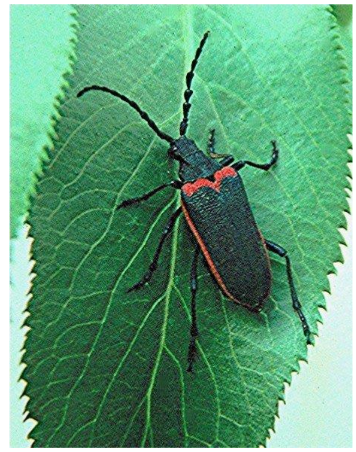
MIG | TRA

Species Map 10.

Central Valley Fall/Late Fall-run Chinook Salmon Habitat Distribution

Placer County Conservation Program – Western Placer County HCP/NCCP

Valley Elderberry Longhorn Beetle
(*Desmocerus californicus dimorphus*)


© USFWS

Status

Federal: Threatened (USFWS 1980). Recommended for delisting in the 5-year review (USFWS 2006). A delisting proposal was released by the USFWS (USFWS 2012); however, the proposed rule was withdrawn (USFWS 2014).

State: None

Recovery Plan: Recovery Plan for the Valley Elderberry Longhorn Beetle (USFWS 1984).

Critical Habitat: Critical habitat was established on August 8, 1980. No Valley elderberry longhorn beetle critical habitat occurs within the Plan Area (USFWS 1980).

Distribution

California

Valley elderberry longhorn beetle (*Desmocerus californicus dimorphus*) is endemic to the upland riparian areas of the Central Valley of California (Linsley and Chemsak 1972). Neither subspecies of *Desmocerus californicus* have been observed outside of California. Three species of the genus *Desmocerus* occur in North America.

At the time of its listing, the valley elderberry longhorn beetle was known in only 10 occurrence records at 3 locations in Merced, Sacramento, and Yolo counties (USFWS 2012). The current range of valley elderberry longhorn beetle extends from Shasta County in the north to Fresno County in the south. It is mostly concentrated at elevations below 3,000 feet in the watersheds of the American, San Joaquin, and Sacramento Rivers. The range of valley elderberry longhorn beetle may overlap with that of *D. c. californicus* along the eastern edge of the Coast Ranges and in the southern San Joaquin Valley (Halstead and Oldham 2000). Delineating the ranges of these two taxa will require focused distribution studies because of the species' reclusive nature, short-lived adult forms, and sexual dimorphism.

Placer County Plan Area

<u>Historical</u>

Upland riparian habitat historically occurred along low-elevation creeks, streams, and rivers throughout western Placer County. Valley elderberry longhorn beetle is likely to have occurred in a patchy distribution along Bear River, Coon Creek, Markham Ravine, Auburn Ravine,

Pleasant Grove Creek, Dry Creek, the American River, and associated tributaries that supported *Sambucus* spp. and associated riparian vegetation.

Current

Valley elderberry longhorn beetle is known to occur in the American River watershed below Auburn in the vicinity of Folsom Lake; in the Dry Creek watershed along Secret Ravine Miners Ravine, and Coon Creek; at the Wildlands Sheridan Mitigation Bank; and in the Bear River watershed near Wheatland in Sutter County. The taxon has not been observed in Placer County higher than 640 feet above sea level (CNDDB 2015; USFWS 2012). To date, bore holes and/or adults have been observed at the following locations within the Plan Area (CNDDB 2015):

- In 1992, near Douglas Boulevard two groups of plants were observed with boreholes.
- In 1991 recent exit holes were observed on red elderberry shrubs at two sites along Miners Ravine.
- In 1992, along Linda Creek, at the Granite Bay Golf Club eight elderberry shrubs with exit holes were recorded.
- In 2002, at a mitigation site at Sterling Point Estates, exit holes were observed from 1993 – 2002 at a 1.84 acre mitigation area.
- In 2003, along the Sutter/Placer County Line just north of Bear River Road, four newly emerged adult beetles were observed.
- In 2005, at Redwings Preserve.
- Barr (1991) observed valley elderberry longhorn beetle at two sites along Folsom Lake in 1991, and these locations were found to be still occupied in 2008 (Holyoak and Koch-Munz 2008) and 2010 (Holyoak and Graves 2010).

Old exit holes were also observed in 1991 in oak woodland along Secret Ravine, at a site which is in western Placer County, but outside of the Plan Area.

In addition, there have been several observations outside of, but near, western Placer County (CNDDB 2015):

- South of western Placer County: along the American River; and
- Northwest of western Placer County: along Bear River, Feather River, and Rest Slough.

Population Status & Trends

California

Valley elderberry longhorn beetle habitat is steadily declining with the elimination of upland riparian habitat throughout its historical range. Less than 1% of the original upland riparian habitat remains, mostly distributed in small, isolated fragments (Collinge et al. 2001). In addition, Vaghti et al. (2009) quantified elderberry stem diameters along the Sacramento River and four adjacent rivers outside of Placer County. Blue elderberry saplings and shrubs with stems <2.0 inches in diameter were rare, which suggests a lack of recruitment (Vaghti et al. 2009).

At the time of its listing, the valley elderberry longhorn beetle was known in only 10 occurrence records at 3 locations, including Merced River, American River, and Putah Creek, in Merced, Sacramento, and Yolo counties (USFWS 2012). Currently, it is known from 201 occurrence records at 26 locations, including much of the San Joaquin and Sacramento Valleys from Shasta County in the northern Sacramento Valley to Kern County in the southern San Joaquin Valley (USFWS 2012).

There are insufficient valley elderberry longhorn beetle records to directly assess changes in distribution of the beetle from historical times to the present, although it is probable that beetle habitat was coarsely related to the extent of riparian forests where elderberry is present (USFWS 2012). However, there is no way of knowing which areas of riparian forest were historically occupied by the beetle (USFWS 2012).

There are no long-term population data available for the valley elderberry longhorn beetle. Studies have attempted to provide information relevant to population trends by surveying and comparing the same sites in the Sacramento Valley. In a statewide distribution study, Barr (1991) found 64 (27.8%) of the 230 sites surveyed to have been recently occupied by valley elderberry longhorn beetle. In the Sacramento Valley region, Barr surveyed 79 sites and observed exit holes at 29 (36.7%) (Collinge et. al. 2001). In 1997, Collinge et al. (2001) repeated Barr's methods at 65 sites in 14 watersheds and found evidence of valley elderberry longhorn beetle occupancy at 30 (46.2%) of the 65 sites. Generally Collinge et al. (2001) found fewer occupied groups of elderberry shrubs at each site (on average) because the average density of elderberry shrubs had decreased. However, although a moderate downward trend was observed, this trend should not necessarily be extrapolated to the long-term, rangewide status of the valley elderberry longhorn beetle due to the uncertainties involved in obtaining the results (e.g., not all beetle habitat surveyed by Barr was surveyed by Collinge). In 2005 and 2006, Holyoak and Koch-Munz (Holyoak and Koch-Munz 2008) surveyed 45 sites and found that 20 (44%) were occupied.

When considering the low estimates of valley elderberry longhorn beetles occupancy (Talley et al. 2007), extinction and colonization patterns (Collinge et al. 2001), and the distribution of the beetle over the last 16 years (since 1997), it is apparent that the valley elderberry longhorn beetle is clustered in regional aggregations and locally uncommon or rare (USFWS 2014).

Placer County Plan Area

Although sample size is small within the Plan Area, a few sites are consistently occupied, while other sites are consistently unoccupied. Holyoak and Koch-Munz (2008) re-surveyed two sites near Folsom Lake that had been found to be occupied by Barr (1991), and found that they remained occupied. Collinge et al. (2001) re-surveyed two sites that had been found to be un-occupied by Barr (1991) and found that they remained unoccupied. Holyoak and Koch-Munz (2008) re-surveyed one of those sites in 2005 and 2006, and it still remained unoccupied. As discussed below, this pattern may be due to the beetle's limited dispersal.

Although previous occupancy of a site can predict current occupancy, within-site populations can vary from year-to-year. Yearly monitoring at the mitigation site at Sterling Pointe Estates – where elderberry and native tree seedlings were planted – revealed that although the site was occupied for several consecutive years, the population varied between years (CNDDB 2015); four exit holes were found in 1992, one hole in 1996, and two holes in 2002.

Natural History

The habitat requirements, ecological relationships, life history, and threats to valley elderberry longhorn beetle described below are summarized in diagram form in the Envirogram 11 Valley Elderberry Longhorn Beetle.

Habitat Requirements

Habitat for valley elderberry longhorn beetle consists of elderberry shrubs (*Sambucus* sp.) occurring in upland riparian forests or elderberry savannas adjacent to riparian vegetation (Barr 1991). Valley elderberry longhorn beetle is found most frequently and most abundantly in areas that support significant riparian zones (Talley et al. 2007). In Collinge et al. (2001) valley elderberry longhorn beetle exit holes were consistently found to occur in clumps of elderberry bushes rather than in isolated bushes, in elderberry branches 2-4 inches in diameter, and in branches less than 3 feet above the ground. Collinge et al. (2001) also found that plants in isolated drainages are less likely to support valley elderberry longhorn beetle populations than plants with connectivity to other habitat. Talley et al. (2007) found that, in general, density of elderberry shrubs, shrub size, number of stems, and range of branch sizes were the most influential predictors of valley elderberry longhorn beetle presence. Increased local population size of beetles was associated with higher elderberry density and the presence of larger, more mature plants (Talley et al. 2007). Valley elderberry longhorn beetle utilizes two species of elderberry plants: blue elderberry (*Sambucus mexicana*) and red elderberry (*Sambucus racemosa* var. *microbotrys*). Valley elderberry longhorn beetle does not seem to select one species over the other (Barr 1991).

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

Holyoak and Koch Munz (2008) surveyed 30 mitigation sites – four that occur within Placer County and three that occur within the Plan Area. They also surveyed 16 nearby natural sites – two of which occur within the Plan Area. When considering the factors that influence whether a site is suitable for the beetle's host plant, Holyoak and Koch-Munz (2008) found that within the mitigation sites, elderberry health and growth were positively correlated with the amount of total nitrogen in soils and less strongly correlated with other soil nutrients and soil moisture. In a related study, they found that elderberry grew more rapidly in sites closer to riparian areas, indicating that such sites should be favored for mitigation sites (Koch-Munz and Holyoak 2008). Fremier and Talley (2009) found that elderberry shrubs were more frequent at intermediate elevations above the floodplain, but also their location was influenced by the width of the floodplain. The wider the floodplain, the higher the elderberry shrubs.

When considering beetle occupancy of host plant habitat, Koch-Munz and Holyoak (2008) found that valley elderberry longhorn beetle populations were denser in sites with moderate levels of dead stems on elderberry shrubs and with moderate damage to elderberry stems and bark. They concluded that this may indicate that the beetle responds to stressed shrubs, which are likely to contain elevated levels of nitrogen. In addition, they found that beetle density increases with the size and age of mitigation sites. They conclude that this is because it takes approximately seven years to develop the basal stem diameters that have been linked to successful beetle colonization.

Talley et al. (2007) found that beetle occupancy was higher in the lower alluvial plain (11.2%) and the mid-elevation riparian corridor (10.5%) than in the upper riparian terrace (8.7%) or the non-riparian scrub (2.9%) of the American River. Talley et al. (2007) also found that the number of exit holes was more than twice as high in the non-riparian scrub than in other habitat types.

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

Reproduction

Valley elderberry longhorn beetle adults are active during the flowering period of the host elderberry plant, usually from March through June (USFWS 2012). The adults feed on the plants' leaves and flowers, and the females lay hundreds of eggs on the plant stems and leaves. Larvae emerge within a few days and burrow into the plant stem that are at least 1 inch in diameter (USFWS 2012). The larva feeds downward through the stem pith, excavating a distinct feeding chamber filled with frass and shredded wood (Barr 1991). After 1–2 years, the larva chews a hole (i.e., exit hole) to the stem surface, but plugs the hole up again from within using wood shavings and returns to the chamber to pupate (Halstead and Oldham 1990). This allows the beetle to eventually exit the stem after it becomes an adult, as adults are not wood borers (USFWS 2012). When the host plant begins to flower, the pupa emerges as an adult and exits the chamber through a characteristic exit hole 0.15–0.4 inch in diameter (Barr 1991).

Dispersal Patterns

Dispersal may be limited by the fact that adults are short-lived and must remain close to elderberry plants for food and to lay eggs (Halstead and Oldham, 1990; Collinge et al. 2001).

Collinge et al. (2001) found that it is rare for valley elderberry longhorn beetle to colonize new sites, even if occupied sites occur within the same drainage, and that they probably never colonize new sites if the nearest occupied sites are in different drainages. This pattern implies that even when an individual valley elderberry longhorn beetle disperses from its host plant to colonize new habitat, it only travels along the riparian corridor within its home drainage. Most remaining elderberry habitat and riparian vegetation exist in small isolated patches; the distance between valley elderberry longhorn beetle populations and unoccupied valley elderberry longhorn beetle habitat limits the species' ability to successfully colonize new sites.

Longevity

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

Sources of Mortality

Any activity that damages the host elderberry plant could result in valley elderberry longhorn beetle mortality. Valley elderberry longhorn beetle larvae are vulnerable to such actions as pesticide application, trimming, dewatering, flooding, and Argentine ant invasion (Huxel 2000; Collinge et al.

2001; Talley pers. comm.). In addition, the beetle is likely prey to insectivorous birds, lizards, and European earwigs (*Forficularia auricularia*) (Klasson et al. 2005, unpublished report cited in USFWS 2006).

Behavior

Valley elderberry longhorn beetle larvae feed on the soft tissues in the center of the elderberry plant. Larvae leave shredded wood and grass behind as they create feeding chambers in the stem. Pupae do not feed. The pupae undergo metamorphosis within an enlarged pupal chamber. Adult beetles feed on the nectar, flowers, and leaves of the host plant or those of another elderberry plant close to the host plant. The emergence of the adult beetle from the elderberry stem creates a characteristic round to oval exit hole 0.15 to 0.4 inch in diameter (Barr 1991; Colinge et al. 2001; Talley pers. comm.).

Movement and Migratory Patterns

The majority of a valley elderberry longhorn beetle's life span is spent within the stem of the host plant (Collinge et al. 2001). Hanks (1999) found that valley elderberry longhorn beetle can complete its entire lifecycle on one individual host plant, even if the host plant is damaged or weakened.

As discussed above, dispersal may be limited by the fact that adults are short-lived and must remain close to elderberry plants for food and to lay eggs.

Ecological Relationships

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

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

Threats

The greatest threats to the persistence of valley elderberry longhorn beetle are habitat loss and fragmentation, flood management, pesticide and herbicide use, and exotic species invasion (USFWS 1984; Huxel 2000; Collinge et al. 2001). Urban and agricultural development, aggregate mining, and flood control practices (e.g., damming and channel maintenance) have damaged or eliminated a large percentage of the upland riparian forests that once occurred in California, reducing and fragmenting the available habitat for valley elderberry longhorn beetle (Barr 1991).

The beetle likely is the prey of insectivorous birds, lizards, and European earwigs (Klasson et al. 2005, unpublished report cited in USFWS 2006). These three common predators move freely up and down elderberry stems searching for food, and earwigs may be common in riparian areas and lay eggs in dead elderberry shrubs.

Invasion of the exotic Argentine ant (*Linepithema humile*) into riparian habitats may present a threat to the distribution and survival of valley elderberry longhorn beetle (USFWS 2012). Although Argentine ants can invade new sites through colonization by queens and/or workers, they can also invade new sites through the soil of potted plants that have been grown or stored at sites with Argentine ant invasions (Holway et al 2003). Huxel (2000) surveyed 15 sites in the Putah Creek watershed and 15 sites in the American River watershed for presence of *L. humile*, native ant species, and valley elderberry longhorn beetle. Results of the Putah Creek survey showed the presence of Argentine ant to have a negative relationship with valley elderberry longhorn beetle presence and showed native ant species to have a positive relationship with valley elderberry longhorn beetle presence. Although results of the American River survey showed no significant relationships, Huxel et al. (2003) observed that the invasion of Argentine ant into the American River watershed was relatively recent (<5 years). Holyoak and Koch-Munz (2008) found that the frequency of Argentine ants was not related to the frequency of valley elderberry longhorn beetle per shrub. However, they recommend caution when interpreting their results because they did not use bait traps to detect ants. In addition, there was a good deal of flooding of sites prior to sampling in 2006, which might have disrupted ant populations. The average number of recent beetle exit holes per elderberry shrub was found to be lower for shrubs with Argentine ants (Holyoak and Graves 2010 as cited in USFWS 2014). The Argentine ant may interfere with adult mating and breeding behavior or prey on valley elderberry longhorn beetle larvae (Huxel et al. 2003; USFWS 2014).

The magnitude and population-level importance of pesticide effects on the beetle remains uncertain, and merits empirical study (USFWS 2006). However, broad-spectrum insecticides are likely toxic to the beetle. In addition, many herbicides may harm or kill its host elderberry plants, and many other broad-spectrum pesticides may be toxic to the beetle and/ or its host plant (USFWS 2006).

Invasive plants pose a particular threat to the valley elderberry longhorn beetle because of the elderberry's intolerance of competition for light, water and nitrogen (Vaghti et al. 2009). Based on vegetation associations, Vaghti et al. (2009) found that non-native fig (*Ficus carica*), Himalayan blackberry (*Rubus armeniacus*), brome (*Bromus* spp.), and giant reed (*Arundo donax*) are of particular concern. In addition, black walnut (*Juglans hindsii*) may compete with elderberry plants for light, and Bermuda grass (*Cynodon dactylon*) may compete with elderberry plants for water and nutrients (Vaghti et al 2009).

Dust is listed in the valley elderberry longhorn beetle recovery plan as a threat to the valley elderberry longhorn beetle. However, Talley et al (2006) found that neither elderberry density nor valley elderberry longhorn beetle density differed with distance from dirt surfaces.

Context for a Regional Conservation Strategy

Valley elderberry longhorn beetle is known from three watersheds and one mitigation bank within western Placer County. Populations in the state are scattered throughout the Central Valley, with Placer County located in the middle to upper distribution of the species' north-south range. Gains in elevation within the County prohibit colonization further east than the western portion of the County. In the region, valley elderberry longhorn beetle has been recorded in counties north and south of western Placer County, such as Yuba and Sutter Counties to the north, Sacramento, El Dorado, and Amador Counties to the south, as well as Yolo County to the east. The Placer County populations are not significant in terms of the range of the species within California. However, due to the severe reduction in suitable riparian habitat for valley elderberry longhorn beetle, protection of remaining habitat, including that in Placer County, is important for the species conservation and restoration. As valley elderberry longhorn beetle will often spend their entire life on the same plant, or disperse to near-by elderberries in the same drainage, protection of occupied plants and connectivity of occupied drainages is of highest priority. Landscape-scale studies of the valley elderberry longhorn beetle have indicated that large patches of habitat, even when unoccupied, are likely important to maintain the possible metapopulation structure of the beetle (Talley 2007).

Modeled Species Distribution in the Plan Area

Model Assumptions

Year-round Habitat

Modeled habitat for valley elderberry longhorn beetle is defined as valley oak woodland and riverine/riparian below 650 feet elevation.

Rationale

Habitat for valley elderberry longhorn beetle consists of elderberry shrubs (*Sambucus* spp.) occurring in upland riparian forests or elderberry savannas adjacent to riparian vegetation. In Placer County, valley elderberry longhorn beetle has not been observed higher than 640 feet above sea level. The presence of host elderberry plants could not be determined from the land-cover data; therefore, modeled habitat for valley elderberry longhorn beetle is likely an overestimate of occupied habitat. Habitat restoration and enhancement actions will include planting (and transplanting) elderberry to suitable sites thereby increasing the extent of occupied and suitable habitat over the term of the PCCP permit.

Model Results

Species Map 11. *Valley Elderberry Longhorn Beetle Modeled Habitat Distribution and Occurrence* shows the modeled potential habitat for valley elderberry longhorn beetle within the Plan Area. The documented occurrences of valley elderberry longhorn beetle generally correspond to modeled year-round habitat. In some cases, locations of documented occurrences did not occur on modeled habitat, possible because habitat features (i.e., elderberry shrubs) did not always correspond with the mapped land-cover type (i.e., riverine/riparian and valley oak woodland).

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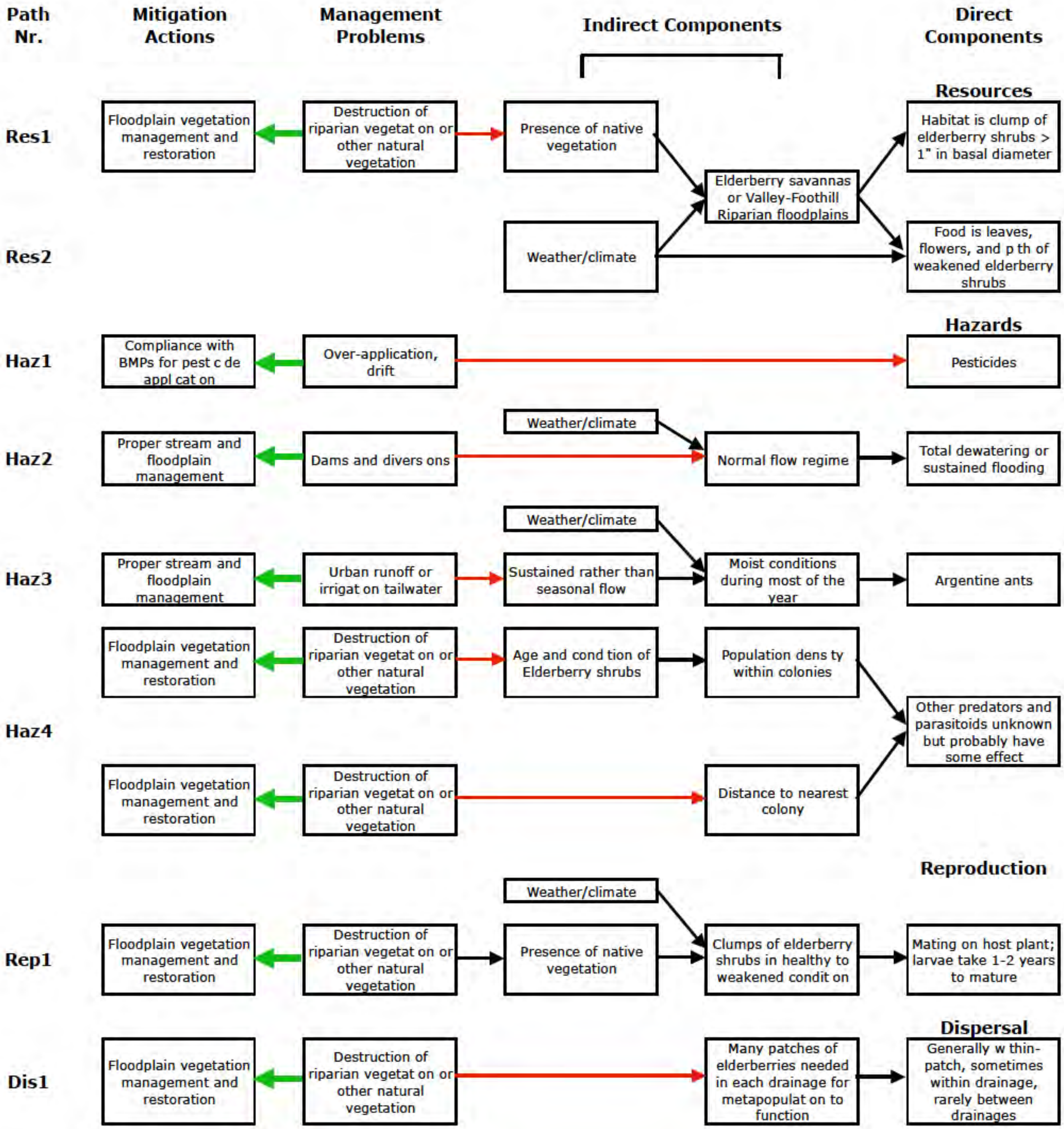
Envirogram 11 Valley Elderberry Longhorn Beetle, *Desmocerus californicus dimorphus*



Envirogram 11 Valley Elderberry Longhorn Beetle. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram Narrative

Valley Elderberry Longhorn Beetle (*Desmocerus californicus dimorphus*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1**:** The habitat and resource needs of valley elderberry longhorn beetle are quite simple; clumps of elderberry shrubs (two species—blue and red elderberry—no evidence of preference) with a basal diameter >1 inch. Two plant communities support elderberries—valley foothill riparian and elderberry savanna. The condition of the shrubs depends on weather conditions and climate trends along with the extent of destruction to riparian vegetation. The loss of elderberry plants can be mitigated by an active floodplain vegetation management and restoration plan.

Res2**:** Valley elderberry longhorn beetle feeds as adults on the leaves and flowers, and the larvae mine and pupate in the pith. This path is the same as Res1.

Hazards

Haz1**:** Drift from improper pesticide application in adjacent agricultural areas is a potential hazard to valley elderberry longhorn beetle. Compliance with best management practices regarding pesticide use and application can reduce this threat.

Haz2**:** Dewatering and flooding resulting from a change in the normal flow regime injures or kills elderberry shrubs. While these events can result from unusual weather conditions, dams and diversions are responsible for most of these problems in Placer County. Proper stream and floodplain management should reduce this problem to some extent.

Haz3**:** Argentine ants evidently prey on one or more life stages of valley elderberry longhorn beetle. These invasive exotics require high soil moisture during most of the year, and an unusually wet year or persistent urban runoff or irrigation tailwater can create appropriate conditions. Allowing the surface soil to dry out during the summer in valley foothill riparian with elderberry and elderberry savanna probably mimics pre-settlement conditions and may help limit colonization by Argentine ants.

Haz4**:** Adult valley elderberry longhorn beetles, with their warning coloration, are probably not preyed on extensively by vertebrates. However, the larvae and pupae may have a complex of fly and wasp parasitoids that can build up during high population densities. The best way to deal with this is to maintain a large number of suitable patches of elderberry shrubs within each drainage. The patches should be far enough apart so that colonization by the beetle is possible but not so close that the parasitoids can find new colonies immediately. Spacing will have to be determined by experimental methods incorporated into an adaptive management framework.

Reproduction

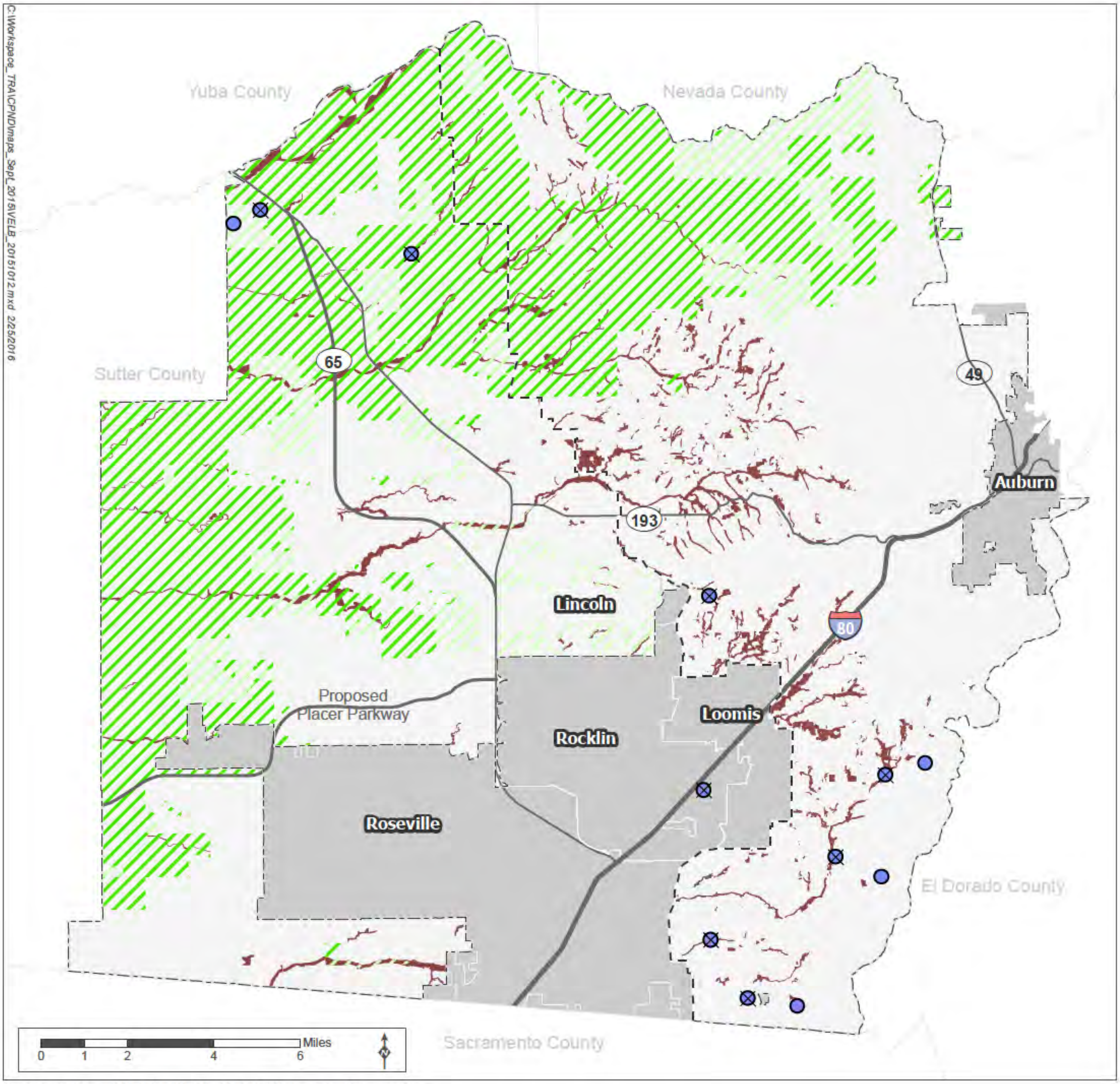
Rep1**:** Valley elderberry longhorn beetle mates on the host plant and the females oviposit on the same or nearby plants. Plants in somewhat weakened condition may be preferred, and plant condition is determined by weather patterns and climate trends and by the overall state of riparian or savanna vegetation. Finding mates should be no problem unless the colony is very sparse. Maintaining or restoring clumps of elderberry shrubs of the proper configuration and spacing should be a component of the floodplain vegetation management plan in the PCCP.

Dispersal

Dis1**:** Valley elderberry longhorn beetle is a very poor disperser, usually moving only within the same clump of elderberry shrubs. Occasional inter-patch dispersal takes place among adjacent patches, but movement between drainages never has been observed. Again, a floodplain vegetation management and restoration plan under the PCCP needs to restore elderberry shrubs in the appropriate patch configuration and structure to ensure the persistence of a metapopulation in each drainage.

Summary

Restoring elderberry savanna and riparian vegetation with elderberry shrubs is the key to recovery of the valley elderberry longhorn beetle in Placer County. However, valley elderberry longhorn beetle is a poor disperser and may need help to colonize restored areas. Maintaining the historic flow regime that allows soil to dry out during the summer may help protect this species from an invasive predator, the Argentine ant.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB, 2015; Restoration Resources, 2005

Species Map 11.

Valley Elderberry Longhorn Beetle

Modeled Habitat Distribution and Occurrence

MIG | TRA

Placer County Conservation Program – Western Placer County HCP/NCCP

Vernal Pool Fairy Shrimp
(*Branchinecta lynchi*)

© Dwight Harvey.
http://www.fws.gov/sacramento/es/recovery_plans/vp_recovery_plan_links.htm

Status

Federal: Threatened (USFWS 1994)

State: None

Critical Habitat: Critical habitat has been designated for vernal pool fairy shrimp (USFWS 2003; USFWS 2005a). Critical habitat for vernal pool fairy shrimp is not present in the Plan Area.

Recovery Plan: Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005). The Plan Area is within the Western Placer County Core Recovery Area (Zone 2) (USFWS 2005b).

Distribution

California

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

Vernal pool fairy shrimp have been reported from the following California Vernal Pool Regions: Northwest Sacramento Valley, Northeast Sacramento Valley, Southeast Sacramento Valley, Solano-Colusa, Livermore, Central Coast, Carrizo, San Joaquin Valley, South Sierra Foothills, and Western Riverside County (California Department of Fish and Game 1998). The California Natural Diversity Database (CNDDB) (2015) lists 738 occurrences of vernal pool fairy shrimp statewide.

Placer County Plan Area

<u>Historical</u>

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

<u>Current</u>

Numerous populations of vernal pool fairy shrimp occur in the Plan Area, which is within the Southeastern Sacramento Valley Vernal Pool Region (USFWS 2007; CNDDB 2015). The majority of extant populations in the Plan Area occur in vernal pools of the northern hardpan and north volcanic mudflow types. These vernal pool types are common to the areas surrounding the Placer County cities of Roseville, Lincoln, and Rocklin within or in close proximity to the Western Placer County core recovery area (USFWS 2007). The most westerly edge of Placer County is primarily converted to rice production and does not contain much vernal pool habitat. The vernal pool fairy shrimp has been recorded from approximately 10 privately or publicly-owned vernal pool, wetland mitigation, or open-space preserves within western Placer County (USFWS 2007). Four of the CNDDB occurrences include multiple records at the Wildlands Inc., Orchard Creek Conservation Bank (CNDDB 2015), where this species has been recorded from 2 of 170 vernal pools surveyed within the conservation bank (USFWS 2007). The vernal pool fairy shrimp has also been observed within the Plan Area at the Redwing Preserve east of Sheridan off of Rioso Road (Restoration Resources 2011) and the Silvergate Mitigation Bank (formerly known as Wildlands Mitigation Bank) south of Wheatland along Riosa Road (Restoration Resources 2010).

Population Status & Trends

California

As of November 2015, the CNDDB listed 738 occurrences of vernal pool fairy shrimp in California (CNDDB 2015). Although vernal pool fairy shrimp are widely distributed, they are locally uncommon throughout their historical range (Eng et al. 1990). In general, the vernal pool fairy shrimp has a sporadic distribution within the vernal pool complexes, with most pools being uninhabited by the species (USFWS 2007). Helm (1998) found vernal pool fairy shrimp in 16.3 percent of pools sampled across 27 counties. Where vernal pool fairy shrimp co-occur with other shrimp species, they are always outnumbered by the other species (Eriksen and Belk 1999).

Placer County Plan Area

Numerous populations of vernal pool fairy shrimp occur in the Plan Area (CNDDB 2015). Several nature preserves and mitigation banks have been established in the Plan Area with the partial goal of preserving habitat for vernal pool fairy shrimp. These preservation areas include Wildlands, Inc.'s, Aitken Ranch Mitigation Bank, Wildlands Mitigation Bank, and Orchard Creek Preservation Area; Eastridge Southern Wetland Preserve; Sterling Pacific Assets' Lincoln Crossing Mitigation Site; Mariner Vernal Pool Conservation Bank, managed by Westervelt Ecological Services; and the City of Roseville's Woodcreek Compensation Area (Jones & Stokes 2004, CNDDB 2015).

Natural History

The habitat requirements, ecological relationships, life history, and threats to vernal pool fairy shrimp described below are summarized in diagram form in Envirogram 12 Vernal Pool Fairy Shrimp.

Habitat Requirements

Vernal pool fairy shrimp inhabit rain-filled ephemeral pools (i.e., vernal pools) that form in depressions, usually in grassland habitats (Eng et al. 1990). Vernal pool fairy shrimp can also inhabit a variety of seasonal wetland habitats (Eng et al. 1990; Helm 1998). Vernal pool fairy shrimp inhabit alkaline pools, ephemeral drainages, pools on rock outcrops, ditches, stream oxbows, stockponds, vernal pools, vernal swales, and other seasonal wetlands. Pools must fill frequently and persist long enough for the species to complete its lifecycle, which takes place entirely within vernal pools. Pools occupied by vernal pool fairy shrimp often have grass or mud bottoms and clear to tea-colored water; they are often in basalt flow depression pools in unplowed grasslands. Water chemistry is key in determining fairy shrimp occurrence; alkalinity, total dissolved solids (TDS), and pH are some of the most important factors (Eriksen and Belk 1999). The species is typically associated with smaller and shallower vernal pools (typically about 6 inches deep) that have relatively short periods of inundation (Helm 1998) and relatively low to moderate TDS and alkalinity (Eriksen and Belk 1999). Occupied habitats range in size from rock outcrop pools as small as 1 square yard to large vernal pools up to 11 acres. The maximum potential water depth of occupied habitat ranges from 1.2 to 48 inches (Helm 1998; Eriksen and Belk 1999).

Vernal pools are characterized by a specific flora endemic to the hydrology and soil composition of the habitat. Vernal pool fairy shrimp and other fairy shrimp species have been observed in depressions filled with water that do not meet the definition of vernal pools (Helm 1998; Stone pers. comm.). Examples of non–vernal pool habitats are roadside ditches, wheel-ruts left by off-highway vehicles or other heavy equipment, and railroad toe-drains (Helm 1998). Vernal pool fairy shrimp are not found in riverine, estuarine, or other permanent waters that support fish (USFWS 1994; Eriksen and Belk 1999).

Reproduction

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

Following fertilization, embryonic and cyst development begins. Embryonic development ceases when the late gastrula stage is reached. At that point, metabolism slows and a halted embryo is isolated from the environment by development of a many-layered membranous shell. The embryo and the shell comprise the cyst, or resting egg. Females carry cysts in a brood sac. Cysts are dropped to the pool bottom or remain in the female's brood sac until the female dies. Cysts are capable of withstanding heat, cold, and prolonged desiccation. When occupied pools fill with water in the same or subsequent seasons, some, but not all, of the deposited cysts may hatch. When temporary pools dry, offspring persist in suspended development as cysts in the pool substrate until the return of winter rains and appropriate temperatures allow some of the cysts to hatch (Eriksen and Belk 1999). The egg bank in the soil may comprise cysts from several years of breeding. When the vernal pools fill with rainwater and the water temperature drops below 50ºF, the resting eggs hatch into small nauplii. The early stages of vernal pool fairy shrimp develop rapidly into adults, reaching maturity in as little as 18 days (Eriksen and Belk 1999). However, the time to maturity and reproduction is temperature-dependent, varying between 18 and 147 days (Helm 1998). Immature and adult shrimp are known to die off when water temperatures rise to approximately 75ºF (Helm 1998).

Dispersal Patterns

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

Longevity

Vernal pool fairy shrimp can achieve reach maturity as few as 18 days after hatching. However, the time to maturity and reproduction is temperature-dependent, varying between 18 and 147 days (Helm 1998). In colder water temperatures (less than 57ºF), individuals have been observed to require 41 days to mature (Helm 1998). Based on laboratory observations, Helm (1998) determined the mean longevity to be 90 days. Field observations indicate that vernal pool fairy shrimp typically persist only 10–12 weeks (Eriksen and Belk 1999; Stone pers. comm.).

Sources of Mortality

The primary threats to vernal pool fairy shrimp are destruction, modification, or curtailment of habitat or range due to urban development; water supply/flood control projects; landfill projects; road development; and agricultural land conversion (USFWS 2007).

Another source of mortality to vernal pool fairy shrimp is predation. The final rule noted that predation of vernal pool crustaceans by nonnative bullfrogs (*Rana catesbeiana*) potentially increased the threat of predation beyond that found naturally (USFWS 2007). Vernal pool crustaceans lack predator-avoidance mechanisms, so they may be particularly susceptible to predation by visual predators (USFWS 2007). Bullfrogs, fish, and crayfish have been noted as potential threats to the species (USFWS 2007). Mosquitofish (*Gambusia affinis*) are also known to occur in significant numbers on vernal pools where the aquatic community or the habitat has been disturbed or degraded (USFWS 1994). Introduced mosquitofish have been shown to significantly reduce fairy shrimp abundance when introduced into pools with active shrimp (Leyse et al. 2004 as cited in USFWS 2007). In addition, both adult fairy shrimp and diapausing cysts can be crushed by foot traffic and off-highway vehicles (Hathaway et al. 1996).

Behavior

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

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

Movement and Migratory Patterns

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

Ecological Relationships

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

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

Threats

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

Within the range of vernal pool fairy shrimp, cities that are rapidly expanding into vernal pool habitat where the shrimp are found include, but are not limited to, White City/Medford in Oregon, and Redding, Chico, Yuba City/Marysville, Roseville, Lincoln, Sacramento, Vacaville, Livermore, Los Banos, Paso Robles, and Hemet in California (USFWS 2007). Growth in Placer County around the City of Roseville and Lincoln is resulting in the loss and fragmentation of an important region of high density vernal pool habitat (USFWS 2007).

Conversion of vernal pool habitat to intensive agriculture continues to contribute to the decline in vernal pools (USFWS 2007). Agricultural conversion primarily threatens vernal pool fairy shrimp in the Northwestern Sacramento Valley, Southeastern Sacramento Valley, San Joaquin Valley, Solano-Colusa, Southern Sierra Foothills, and Carrizo Vernal Pool Regions (USFWS 2007).

Vernal fairy shrimp are also threatened by the encroachment of non-native annual grasses and altered hydrology (USFWS 2007). Non-native grasses maintain dominance at pool edges, sequestering light and soil moisture, promoting thatch build-up, and shortening inundation periods (USFWS 2007). Although the mechanism responsible for the change in inundation is not documented, reduction in inundation period is thought to be due to increased evapo-transpiration at the vernal pools (Marty 2005).

Both lack of grazing and excessive grazing can cause an increase in organic matter in vernal pool habitat that can eliminate the natural vernal pool invertebrate community and promote opportunistic non-native, invasive annual grass species that out compete the obligate vernal pool species (USFWS 2007). In addition, cattle increase water turbidity, deplete water levels in the vernal pools, and can directly damage vernal pool tadpole shrimp cysts with their hooves (USFWS 2007). Conversely, some vernal pools need a certain amount of grazing in order to keep them from being overgrown with non-native plants that generate deep thatch layers on the pool substrate (USFWS 2007). Cessation of cattle grazing has been found to exacerbate the negative effects of invasive nonnative plants on vernal pool inundation period, presumable due to the positive effects of grazing on evapo-transpiration rates (USFWS 2007). Vernal pool inundation has been reduced by 50 to 80 percent in the southeastern Sacramento Valley when grazing is discontinued (Marty 2005).

Context for a Regional Conservation Strategy

Vernal pool fairy shrimp are known from 18 populations in the Plan Area, and may also exist in additional locations that have not been surveyed. The majority of extant populations in the Plan Area occur in vernal pools of the northern hardpan and north volcanic mudflow types. These vernal pool types are common to the areas surrounding the Placer County cities of Roseville, Lincoln, and Rocklin within or in close proximity to the Western Placer County core recovery area (USFWS 2007). The most westerly edge of Placer County is primarily converted to rice production and does not contain much vernal pool habitat. The vernal pool fairy shrimp has been recorded from approximately 10 privately or publicly-owned vernal pool, wetland mitigation, or open-space preserves within western Placer County (USFWS 2007). Four of the CNDDB occurrences include multiple records at the Wildlands Inc. Orchard Creek Conservation Bank (CNDDB 2015), where this species has been recorded from 2 of 170 vernal pools surveyed within the conservation bank (USFWS 2007). The vernal pool fairy shrimp has also been observed within the Plan Area at the Redwing Preserve east of Sheridan off of Riosa Road (Restoration Resources 2011) and the Silvergate Mitigation Bank (formerly known as Wildlands Mitigation Bank) south of Wheatland along Riosa Road (Restoration Resources 2010).

In the region, vernal pool fairy shrimp is found in vernal pool complexes north and south of the Placer County populations, including Yuba, Butte, Sutter, Sacramento, Yolo and Solano counties, among others. There is an absence of suitable habitat to the east, and thus the western Placer County populations probably represent the furthest eastward range of the species for the area. Within California, the greatest concentration of known populations occurs within the vernal pool complexes of western Placer County and Sacramento County. For conservation of vernal pool fairy shrimp within the Plan area, acquisition and conservation of vernal pool habitat and associated uplands and supporting hydrological systems is of highest priority.

The Plan Area is within the Western Placer County Core Recovery Area (Zone 2) identified in the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005b; USFWS 2007). There are multiple sites within this core area that are protected for the benefit of vernal pool species, including the Orchard Creek Vernal Pool Conservation Bank, Twelve Bridges Preserve, Sheridan Conservation Bank, and Yankee Slough Conservation Bank. The U.S. Air Force's Lincoln Communication Facility, which is part of the McClellan Air Force Base, is now part of the 220-acre Western Placer Schools Conservation Bank (USFWS 2007).

Modeled Species Distribution in the Plan Area

Model Assumptions

Year-round Habitat

Modeled year-round habitat for vernal pool fairy shrimp is defined by all densities of vernal pool grassland complex.

Rationale

Vernal pool fairy shrimp inhabits vernal pools that form in depressions, usually in grassland habitats. Pools must fill frequently and persist long enough for this species to complete its lifecycle, which takes place entirely within vernal pools. Not all mapped vernal pools and vernal pool grassland complexes have pools that provide suitable habitat features for vernal pool fairy shrimp; the level of detail necessary to identify microhabitat features (e.g., size and depth of pools, water chemistry) suitable for vernal pool fairy shrimp are not captured in the GIS land-cover data. Therefore, modeled habitat may overestimate suitable habitat available for vernal pool fairy shrimp.

Model Results

Species Map 12. *Vernal Pool Fairy Shrimp Modeled Habitat Distribution and Occurrence* shows the modeled habitat for vernal pool fairy shrimp in the Plan Area. Modeled habitat occurs in the western, Valley portion of the Plan Area, largely below 200 feet elevation. The documented occurrences of vernal pool fairy shrimp corresponds well with modeled habitat.

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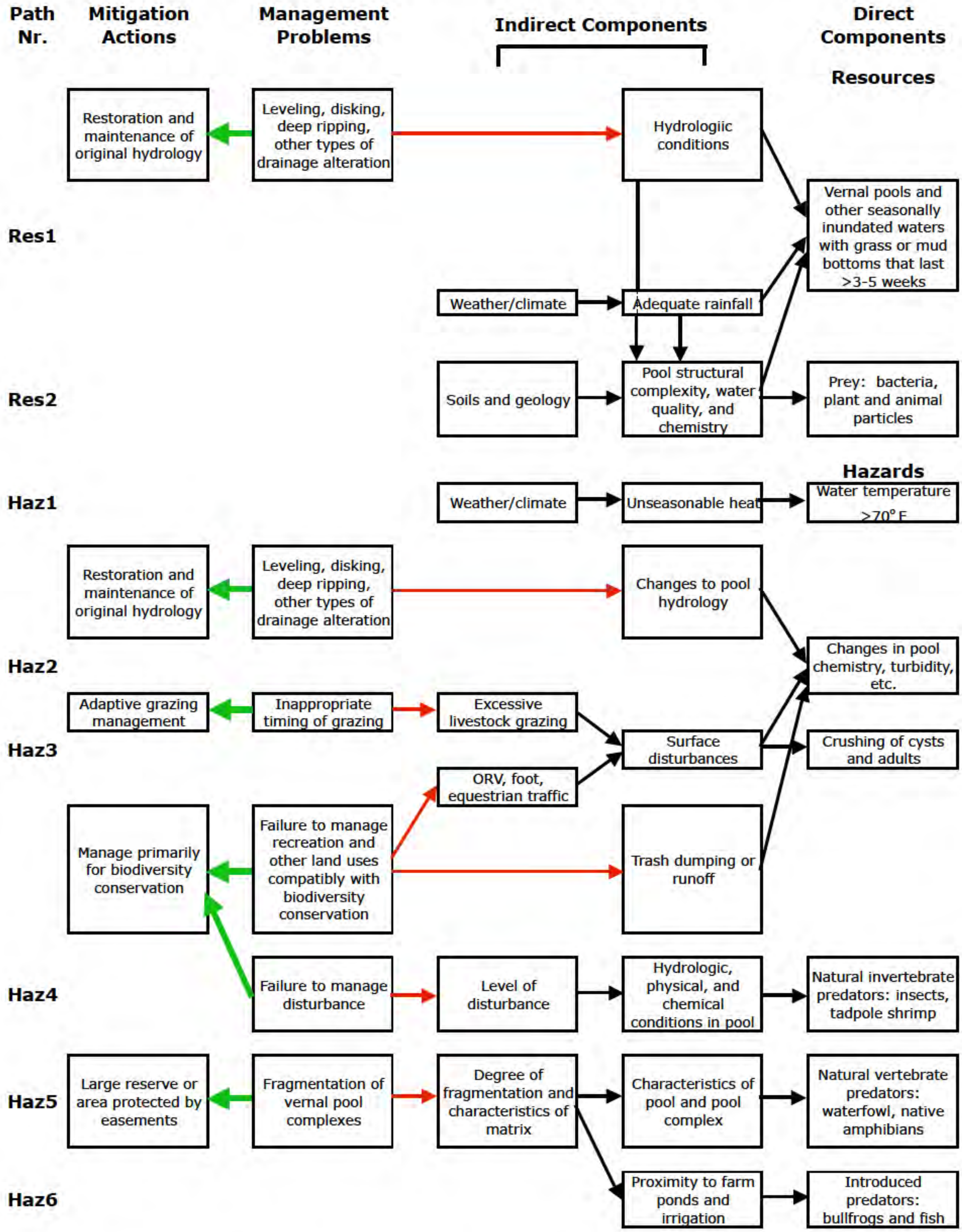
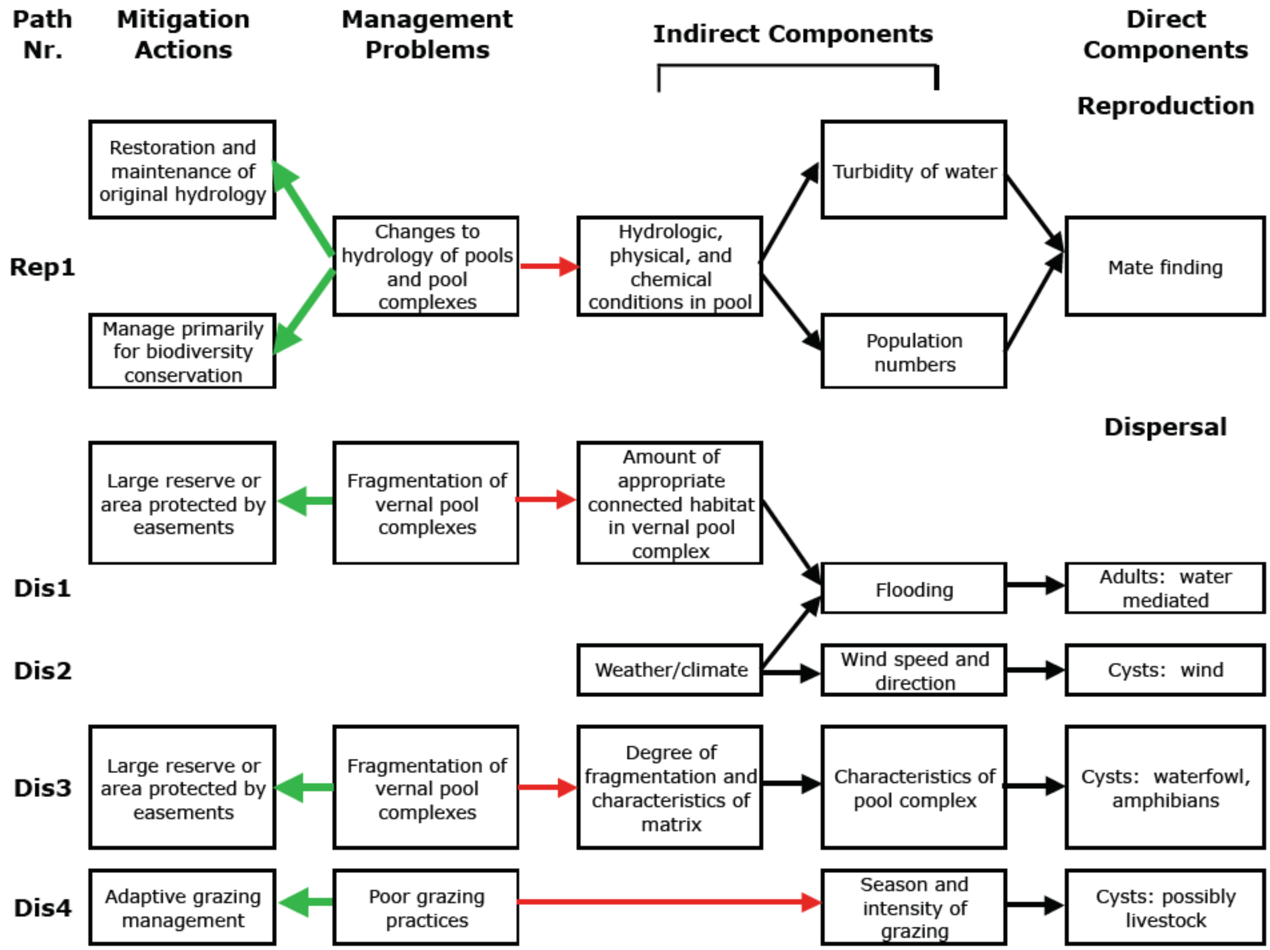
Envirogram 12 Vernal Pool Fairy Shrimp, *Branchinecta lynchi* (page 1)



Envirogram 12 Vernal Pool Fairy Shrimp. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram 12 Vernal Pool Fairy Shrimp, *Branchinecta lynchi* (page 2)



Envirogram Narrative

Vernal Pool Fairy Shrimp (*Branchinecta lynchi*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Vernal pool fairy shrimp are most commonly found in vernal pools and other seasonally inundated waters with grass or mud bottoms that last long enough for them to complete their life cycle (>3-5 weeks). Such waters are usually associated with natural hydrologic conditions; waters that have been modified by leveling, disking, deep ripping, and other types of drainage alterations are generally not suitable, and such water bodies must be restored to their natural hydrologic conditions. Adequate rainfall, a function of weather and climate, is necessary to fill the pools to the appropriate depth, and the structural complexity of the pool and its water quality and chemistry also influence its suitability for vernal pool fairy shrimp.

Res2: Vernal pool fairy shrimp feed on bacteria and plant and animal particles. Abundant and diverse prey species depend on the structural complexity of the pool and its water quality and chemistry, which in turn are influenced by the soils and geological formations in which the pool occurs as well as by hydrologic conditions and the amount and timing of rainfall.

Hazards

Haz1: Vernal pool fairy shrimp are killed by water temperatures >70 °F, which can occur during periods of unseasonable heat. (A warming climate with an increasing frequency of extreme weather events could result in increasing problems of this kind in the future).

Haz2: Changes in pool chemistry and turbidity can be detrimental to vernal pool fairy shrimp. These changes can result from modifications to pool hydrology as a result of drainage alteration and from surface disturbances caused by excessive livestock grazing or recreational use. They also can result from trash dumping or runoff from various sources. Restoration of the original hydrologic conditions and close management of grazing, recreation, runoff, and dumping are necessary to preserve appropriate conditions for this species. This is best achieved by managing vernal pool complexes primarily for biodiversity conservation.

Haz3: Crushing of cysts in dry pools result from surface disturbances such as livestock grazing and ORV, foot, or equestrian traffic. Adults can be crushed by cattle in shallow pools that are drying out. Management primarily for biodiversity conservation is the best mitigation for these hazards.

Haz4: The abundance of natural invertebrate predators such as insects and tadpole shrimp depends on the hydrologic, physical, and chemical conditions in a pool. Excessive disturbance can create conditions that create unnaturally high densities of these predators.

Haz5: Natural vertebrate predators of vernal pool fairy shrimp include waterfowl and native amphibians. The presence of these species depends upon the characteristics of the individual pool and the pool complex, which in turn are determined by the degree of fragmentation of the complex and the characteristics of the surrounding area. Fragmentation and location of the pool complex may result in abnormally high or low densities of these predators in certain pools, which could be a benefit or a disaster to a vernal pool fairy shrimp population.

Haz6: If pool complexes are located near farms and irrigation structures, introduced predators such as bullfrogs and fish could be introduced into a pool, which would inevitably result in local extirpations. Large, unfragmented pool complexes, located well away from farm ponds and irrigation ditches and managed primarily for biodiversity conservation are the best management option to control these hazards.

Reproduction

Rep1: Successful reproduction by vernal pool fairy shrimp depends on finding mates, which is largely dependent upon the turbidity of the water and the numbers of individuals in a pool. Both of these factors are related to the hydrologic, physical, and chemical conditions in the pool. Any alteration to the hydrology of the pool or pool complex can make conditions unsuitable for reproduction. Restoring the original hydrology and managing a pool complex primarily for biodiversity conservation is the best way to preserve the conditions needed for reproduction.

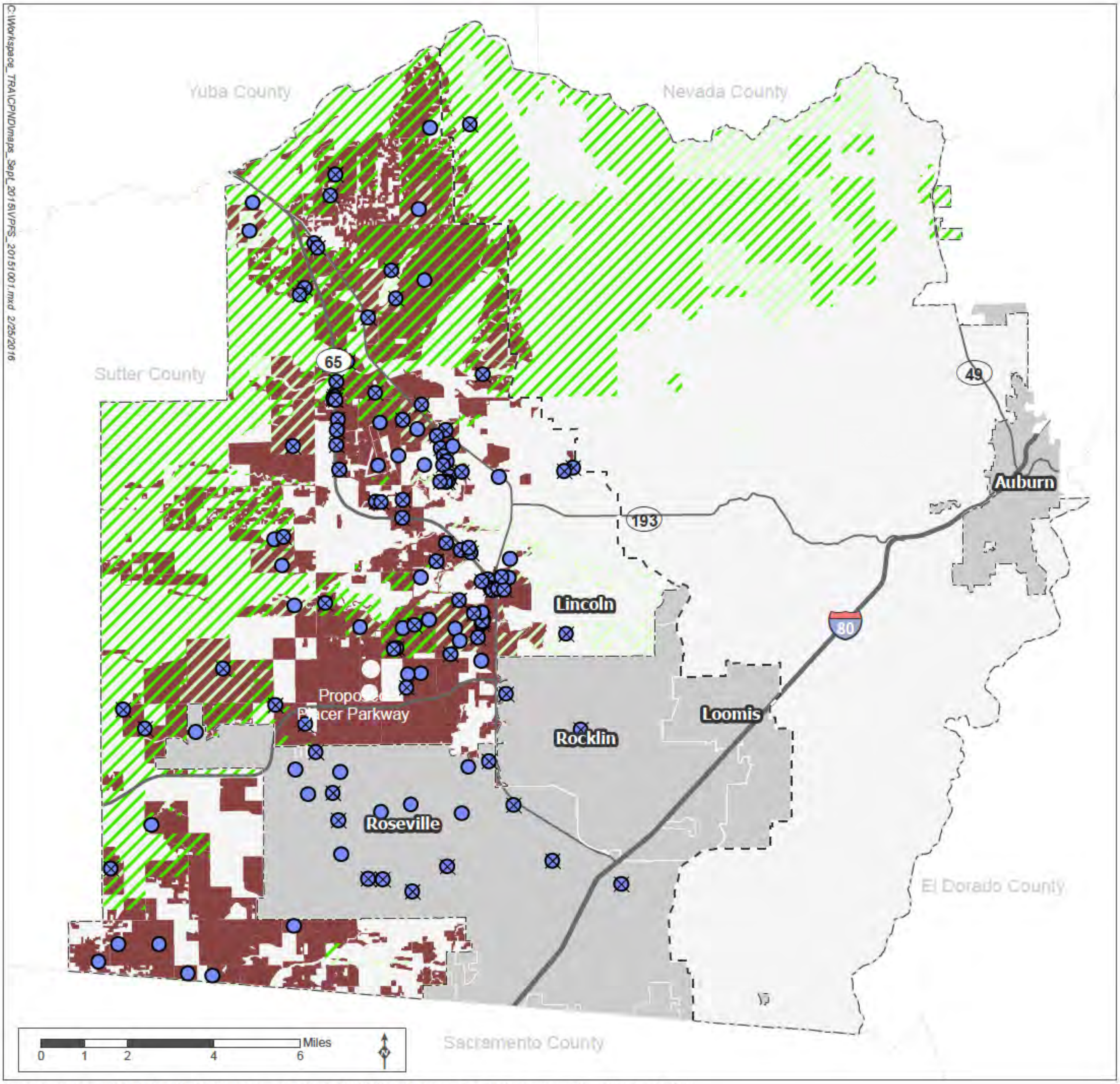
Dispersal

Dis1: Adults can disperse from pool to pool during periods of flooding caused by abundant rainfall, provided that there are appropriate pools to disperse to. Such dispersal is not very likely in small or highly fragmented vernal pool complexes.

Dis2: Cysts can be transported by the wind from dry pools; successful dispersal depends on wind speed and direction.

Dis3: Cysts also can be transported in the guts of waterfowl or amphibians. Success in this mode of dispersal depends on where the cysts are deposited. The chances of a cyst arriving in a suitable location are enhanced considerably in a large, unfragmented pool complex. Dis1 and 3 are facilitated by establishing large reserve areas and managing them primarily for biodiversity conservation.

Dis4: Cysts possibly may be transported by livestock, attached to mud on their hooves. This event would depend on livestock being in the right place at the right time and in densities that are not likely to result in excessive surface disturbance. Adaptive grazing management within a reserve must consider all these factors.



Source: Placer County, 2014; MIG | TRA, 2015; Helm Biological, 2010; CNDDB 2015; LSA Associates, Inc., 1999; Placer Land Trust; Robert Schell, 2013

Occurrences
- Precise Location
- General Location

Modeled Habitat
- Vernal Pool Complex
- Non-habitat

- Existing Protected Area
- Reserve Aquisition Area
- Non-participating City

- Major Road
- Valley/Foo hill Divide
- Area A Boundary

MIG TRA

Species Map 12.

Vernal Pool Fairy Shrimp Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

Vernal Pool Tadpole Shrimp
(*Lepidurus packardi*)

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Status

Federal: Endangered (USFWS 1994)

State: None

Critical Habitat: Critical habitat has been designated for vernal pool tadpole shrimp (USFWS 2003; USFWS 2005a). Critical habitat for vernal pool tadpole shrimp is not present in the Plan Area.

Recovery Plan: Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005b). The Plan Area is within the Western Placer County Core Recovery Area (Zone 2) (USFWS 2005b).

Distribution

California

Vernal pool tadpole shrimp (*Lepidurus packardi* Simon, 1886) is endemic to the Central Valley of California (USFWS 1994; Helm 1998; Rogers 2001; USFWS 2005b). Rogers (2001) determined that specimens from southern Oregon and the California Central Valley that were originally described as *L. packardi* were, in fact, *Lepidurus cryptus*, a recently described species of tadpole shrimp.

The historical range of vernal pool tadpole shrimp in California includes annual grasslands of the Great Central Valley. Today the species has a patchy distribution from Shasta County in the north to Tulare County in the south, with disjunct populations occurring in Alameda and Contra Costa Counties (USFWS 2007; California Natural Diversity Database 2015).

Vernal pool tadpole shrimp have been reported from the following California Vernal Pool Regions: Northwest Sacramento, Northeast Sacramento, Southeast Sacramento, Solano-Colusa, San Joaquin Valley, South Sierra Foothill, and Central Coast (USFWS 2007). The California Natural Diversity Database (2015) lists 309 occurrences of vernal pool tadpole fairy shrimp in California. These occurrences have been documented in 20 counties, including Alameda, Butte, Colusa, Contra Costa, Fresno, Glenn, Kings, Merced, Placer, Sacramento, San Benito, San Joaquin, Shasta, Solano, Stanislaus, Sutter, Tehama, Tulare, Yolo, and Yuba counties. Sacramento County contains the greatest amount of the known occurrences (USFWS 2007).

Placer County Plan Area

<u>Historical</u>

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

<u>Current</u>

There are four recent occurrences of vernal pool tadpole shrimp in the Plan Area. In 1996, vernal pool tadpole shrimp, was found at the U.S. Air Force Lincoln Communications Facility where at least four vernal pools of a 236-pool complex supported vernal pool tadpole shrimp (36 pools were surveyed). Nearby, a population was found on the West Placer School District property, between Markham Ravine and Auburn Ravine. In 2006, twenty shrimp were observed on the site's 9.38 acres of naturally occurring wetlands and swales (California Natural Diversity Database 2015). Vernal pool tadpole shrimp has been found at Woodcreek Oaks Mitigation Site between Kasenburg Creek and the south branch of Pleasant Grove Creek; adults were observed in one pool on this site in 1995; however, this occurrence may be extirpated (USFWS 2007). In 2003, Helm Biological Consulting found vernal pool tadpole shrimp near the intersection of Watt Avenue and Baseline Road (Helm 2012).

Population Status & Trends

California

Vernal pool tadpole shrimp distribution has been greatly reduced from historical times as a result of widespread destruction and degradation of its vernal pool habitat (USFWS 2005b). Vernal pool habitats in the Central Valley are reduced from their former area and the remaining habitats are more fragmented and isolated than during historical times (Holland 1998). As of October 2015, the California Natural Diversity Database (CNDDB) listed 309 extant occurrences of vernal pool tadpole shrimp in California. Although vernal pool tadpole shrimp is widely distributed in California, it is now locally uncommon throughout the historical range (Helm 1998; Eriksen and Belk 1999).

Placer County Plan Area

The CNDDB lists three occurrences of vernal pool tadpole shrimp (CNDDB 2015) and other surveys have found vernal pool tadpole shrimp within the Plan Area (Helm 2012). Several nature preserves and mitigation banks have been established in the Plan area with the partial goal of preserving habitat for vernal pool tadpole shrimp. These preservation areas include Wildlands, Inc.'s, Aitken Ranch Mitigation Bank, Wildlands Mitigation Bank, and Orchard Creek Preservation Area; Eastridge Southern Wetland Preserve; Sterling Pacific Assets' Lincoln Crossing Mitigation Site; and the City of Roseville's Woodcreek Compensation Area (Jones & Stokes 2004, CNDDB 2015).

Natural History

The habitat requirements, ecological relationships, life history, and threats to vernal pool tadpole shrimp described below are summarized in diagram form in the Envirogram 13 Vernal Pool Tadpole Shrimp.

Habitat Requirements

Vernal pool tadpole shrimp occur in a variety of natural and artificial seasonally inundated habitats (Helm 1998). They require seasonally aquatic habitats that are wet for at least seven weeks and dry in summer (Gallagher 1996). Helm (1998) observed vernal pool tadpole shrimp occurring in vernal pools (natural, artificial, and constructed), seasonal wetlands (natural and artificial), alkaline pools, clay flats, vernal swales, stockponds, railroad right-of-way pools, roadside ditches, and road rut pools resulting from vehicular activity. Occupied pools and wetlands typically have highly turbid waters or aquatic vegetation that may provide shelter from predators (USFWS 1994; USFWS 2007; Stone pers. comm.). Although vernal pool tadpole shrimp have been reported to occur in turbid water (USFWS 2007), it is possible that the vernal pool tadpole shrimp actually causes the turbidity since it has been found to be a bioturbator (Croel and Kneitel 2011). Vernal pool tadpole shrimp have been collected in vernal pools ranging in size from 6.5 square feet to 88 acres (Helm 1998).

Reproduction

Vernal pool tadpole shrimp may be hermaphroditic (i.e., individuals have both male and female reproductive organs) (Rogers 2001). Diapausing cysts (eggs) occurring in the dry pool bottom hatch within 3 weeks of inundation (Ahl 1991). The hatched neonate is a metanauplius that undergoes several molts, each gaining additional phyllopod appendages until reaching sexual maturity. This process takes approximately 6–7 weeks depending on temperature and food availability (Ahl 1991; Gallagher 1996; Helm 1998). Reproduction occurs throughout the ponding season, when females average 0.39–0.47 inch in carapace length (Ahl 1991). Vernal pool tadpole shrimp have relatively high reproductive rates (USFWS 2005b). Ahl (1991) found that fecundity increases with body size; large females (greater than 0.8 inch carapace length) could deposit as many as 6 clutches ranging from 32 to 61 eggs per clutch in a single wet season. Laboratory studies conducted by Ahl (1991) revealed that eggs can hatch during the same ponding event in which they were laid without intervening dehydration. The remaining unhatched cysts settle to the pool substrate and contribute to the cyst bank for subsequent wet seasons. Optimal hatching temperature occurs between 50 and 59 degrees Fahrenheit (ºF) with hatching rates becoming significantly lower at temperatures above 68 ºF (Ahl 1991).

Dispersal Patterns

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

Longevity

Vernal pool tadpole shrimp is considered a long-lived species (USFWS 2005b). Adults are often present and reproductive until the pools dry up in the spring (USFWS 1994). Vernal pool tadpole shrimp continue to grow throughout their lives, periodically molting their shells (USFWS 2005b). Helm (1998) found that vernal pool tadpole shrimp took a minimum of 25 days to mature and the mean age at first reproduction was 54 days. Other researchers have observed that vernal pool tadpole shrimp generally take between 3 and 4 weeks to mature (Ahl 1991).

Sources of Mortality

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

Behavior

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

Ecological Relationships

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

Vernal pool tadpole shrimp commonly co-occur with vernal pool fairy shrimp (*Branchinecta lynchi*), Conservancy fairy shrimp (*Branchinecta conservatio*), and California linderiella (*Linderiella occidentalis*) (Helm 1998; Stone pers. comm.). Vernal pool tadpole shrimp are bioturbators and may affect other plant and animal communities in the vernal pool ecosystem by creating turbid water (Croel and Kneitel 2011).

Threats

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

Vernal pool tadpole shrimp are also threatened by the encroachment of non-native annual grasses and altered hydrology (USFWS 2005b; USFWS 2007). Timing, frequency, and length of inundation of vernal pools are critical to vernal pool species. Modification of the watershed surrounding the pools can allow non-native plants and/or opportunistic invertebrates to become established or eliminate the vernal pool habitat altogether (Roger 1998 as cited in USFWS 2007). Hydrology can be altered through direct means (e.g., construction of roads) or indirect means (e.g., diversions of overland flow), both of which result in decreased runoff to the vernal pool complexes and cause the pools to either not fill or to dry prematurely (USFWS 2007). Changes in upland hydrology that results in shorter inundation periods is of particular concern in vernal pool tadpole shrimp due to the species requirement for nearly two months to reach maturity (Helm 1998).

Both lack of grazing and excessive grazing can cause an increase in organic matter in vernal pool habitat that can eliminate the natural vernal pool invertebrate community and promote opportunistic non-native, invasive annual grass species that out compete the obligate vernal pool species (Roger 1998 as cited in USFWS 2007). In addition, cattle increase water turbidity, deplete water levels in the vernal pools, and can directly damage vernal pool tadpole shrimp cysts with their hooves (USFWS 2007). Conversely, some vernal pools need a certain amount of grazing in order to keep them from being overgrown with non-native plants that generate deep thatch layers on the pool substrate (USFWS 2007).

In addition, parasitic castration by flukes (*Trematoda*) has been identified as a major limiting factor for some vernal pool tadpole shrimp populations, such as at the Vina Plains in Tehama County (Ahl 1991).

Context for a Regional Conservation Strategy

Vernal pool tadpole shrimp are known from four populations in the Plan Area, and may also exist in additional locations that have not been surveyed. In the region, vernal pool tadpole shrimp is found in vernal pool complexes north and south of the Placer County populations, including Yuba, Butte, Sutter, Sacramento, Yolo and Solano counties, among others. There is an absence of suitable habitat to the east, and thus the western Placer County populations probably represent the furthest eastward range of the species for the area. Within California, the greatest concentration of known populations occurs within the vernal pool complexes of Sacramento County. The Plan Area is within the Western Placer County Core Recovery Area (Zone 2) identified in the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005b; USFWS 2007). There are multiple sites within this core area that are protected for the benefit of vernal pool species, including the Orchard Creek Vernal Pool Conservation Bank, Twelve Bridges Preserve, Sheridan Conservation Bank, and Yankee Slough Conservation Bank. The U.S. Air Force's Lincoln Communication Facility, which is part of the McClellan Air Force Base, is now part of the 220-acre Western Placer Schools Conservation Bank (USFWS 2007). For conservation of vernal pool tadpole shrimp within the Plan Area, acquisition and conservation of vernal pool habitat and associated uplands and supporting hydrological systems is of highest priority.

Modeled Species Distribution in the Plan Area

Model Assumptions

<u>Year-round Habitat</u>

Modeled year-round habitat for vernal pool tadpole shrimp is defined by all densities of vernal pool grassland complex.

Rationale

Vernal pool tadpole shrimp inhabits vernal pools that form in depressions, usually in grassland habitats. Pools must fill frequently and persist long enough for this species to complete its lifecycle, which takes place entirely within vernal pools. Not all mapped vernal pools and vernal pool grassland complexes have pools that provide suitable habitat features for vernal pool tadpole shrimp; the level of detail necessary to identify microhabitat features (e.g., size and depth of pools, water chemistry) suitable for vernal pool tadpole shrimp are not captured in the GIS land-cover data. Therefore, modeled habitat may overestimate suitable habitat available for vernal pool tadpole shrimp.

Model Results

Species Map 13. *Vernal Pool Tadpole Shrimp Modeled Habitat Distribution and Occurrence* shows the modeled habitat for vernal pool tadpole shrimp in the Plan Area. Modeled habitat occurs in the western, Valley portion of the Plan Area, generally below 200 feet elevation. The documented occurrences of vernal pool tadpole shrimp falls within the modeled habitat.

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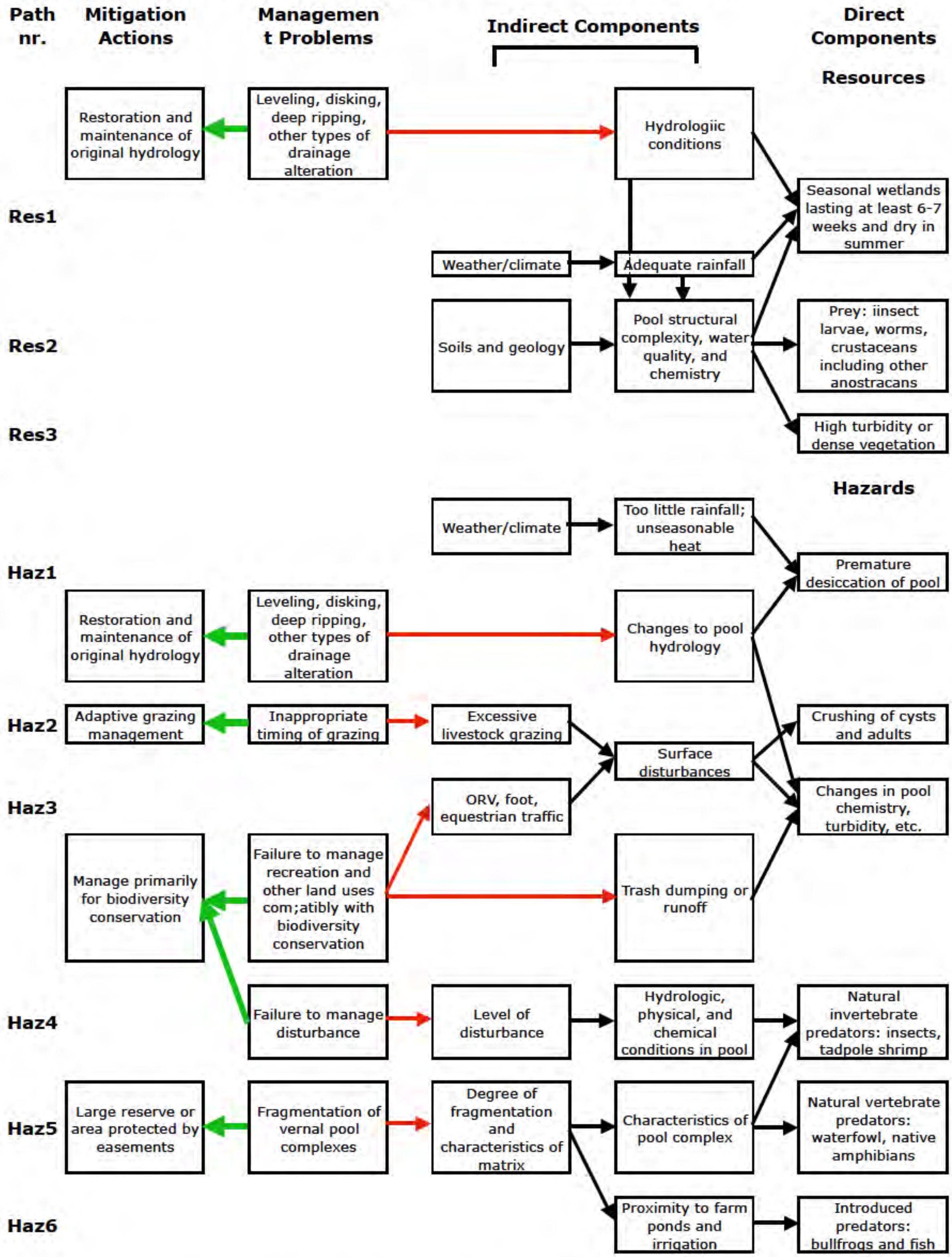
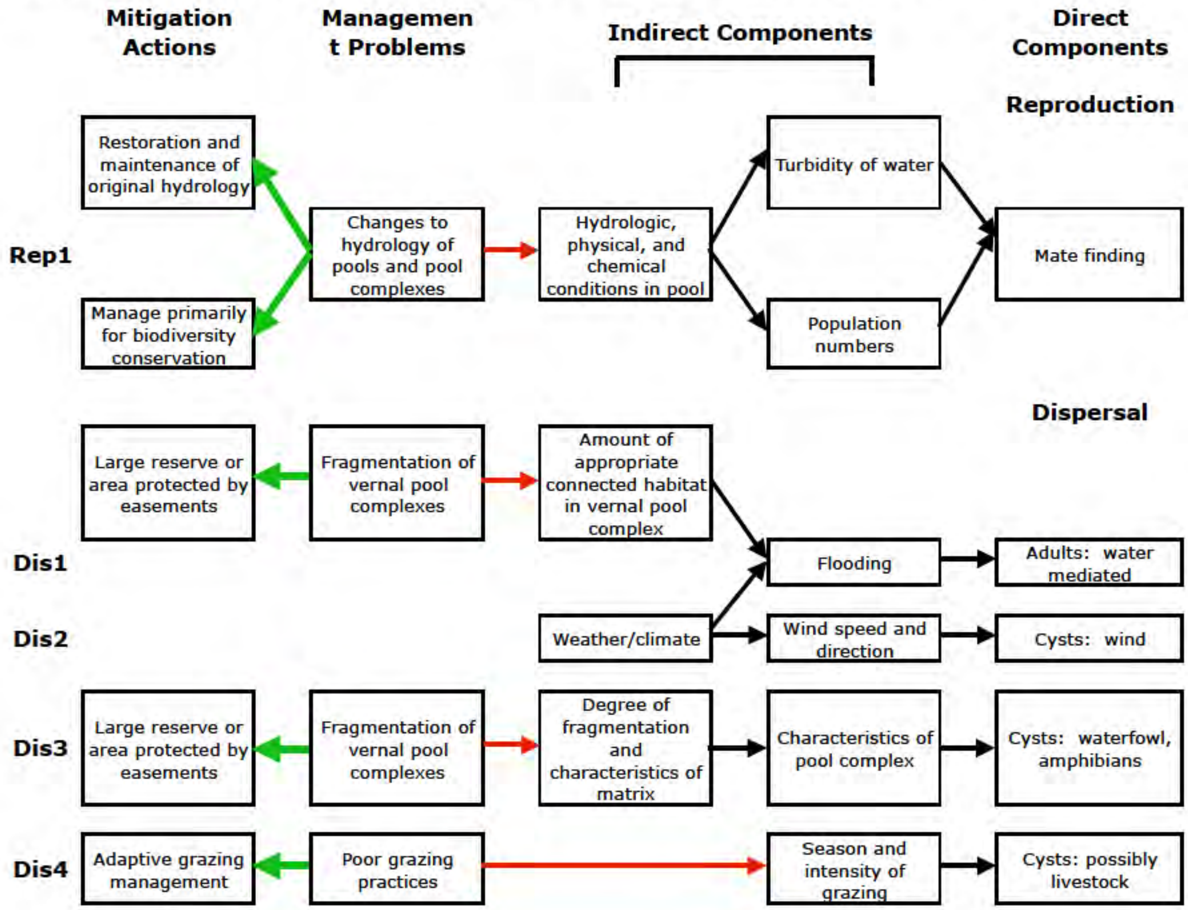
Envirogram 13 Vernal Pool Tadpole Shrimp, *Lepidurus packardii* (page1)



Envirogram 13 Vernal Pool Tadpole Shrimp. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram 13 Vernal Pool Tadpole Shrimp, *Lepidurus packardii* (page 2)



Envirogram Narrative

Vernal Pool Tadpole Shrimp (*Lepidurus packardii*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Vernal pool tadpole shrimp is found in vernal pools and seasonal wetlands that last long enough for it to complete its life cycle (at least 6-7 weeks) and dry up in summer. Such waters are often but not necessarily associated with natural hydrologic conditions; waters that have been modified by leveling, disking, deep ripping, and other types of drainage alterations often are not suitable. Altered water bodies should be restored to their natural hydrologic conditions if they are to support this species. Adequate rainfall, a function of weather and climate, is necessary to fill the pools to the appropriate depth, and the structural complexity of the pool and its water quality and chemistry also influence its suitability for the vernal pool tadpole shrimp.

Res2**:** Vernal pool tadpole shrimp feeds on small invertebrates such as insect larvae, worms and other crustaceans. Abundant and diverse prey species depend on the structural complexity of the pool and its water quality and chemistry, which in turn are influenced by the soils and geological formations in which the pool occurs as well as by hydrologic conditions and the amount and timing of rainfall.

Res3**:** Vernal pool tadpole shrimp requires pools with high turbidity or dense vegetation, probably to protect them from vertebrate predators.

Hazards

Haz1: Vernal pool tadpole shrimp are killed by premature desiccation of the pool, which can result from too little rainfall or unseasonable heat. (A warming climate with an increasing frequency of extreme weather events could create more problems in this regard in the future). Premature desiccation also can result from changes to pool hydrology, discussed in Res1 above.

Haz2: Crushing of cysts in dry pools result from surface disturbances such as livestock grazing and ORV, foot, or equestrian traffic. Adults also may be crushed by livestock in shallow pools that are drying out. Management primarily for biodiversity conservation is the best mitigation for these hazards.

Haz3: Changes in pool chemistry and turbidity can be detrimental to vernal pool tadpole shrimp. These changes can result from modifications to pool hydrology as a result of drainage alteration and from surface disturbances caused by excessive livestock grazing or recreational use. They also can result from trash dumping or runoff from various sources. Restoration of the original hydrologic conditions and close management of grazing, recreation, runoff, and dumping are necessary to preserve appropriate conditions for this species. This is best achieved by managing vernal pool complexes primarily for biodiversity conservation.

Haz4: The abundance of natural invertebrate predators such as insects and other tadpole shrimp depends on the hydrologic, physical, and chemical conditions in a pool. Excessive disturbance can create conditions that result in unnaturally low or high densities of these predators, either to the benefit or detriment of the vernal pool tadpole shrimp population.

Haz5: Natural vertebrate predators of vernal pool tadpole shrimp include waterfowl and native amphibians. The presence of these species depends upon the characteristics of the individual pool and the pool complex, which in turn are determined by the degree of fragmentation of the complex and the characteristics of the surrounding area. Fragmentation and location of the pool complex may result in abnormally low or high densities of these predators in certain pools, which could be an advantage or a disaster to a vernal pool tadpole shrimp population.

Haz6: If pool complexes are located near farms and irrigation structures, introduced predators such as bullfrogs and fish could be introduced into a pool, which would inevitably result in local extirpations. Large, unfragmented pool complexes, located well away from farm ponds or irrigation ditches and managed primarily for biodiversity conservation are the best management option to control these hazards.

Reproduction

Rep1: Successful reproduction of vernal pool tadpole shrimp depends on finding mates, which is largely dependent upon the turbidity of the water and the numbers of individuals in a pool. Both of these factors are related to the hydrologic, physical, and chemical conditions in the pool. Any alteration to the hydrology of the pool or pool complex can make conditions unsuitable for reproduction. Restoring the original hydrology and managing a pool complex primarily for biodiversity conservation is the best way to preserve the conditions needed for reproduction.

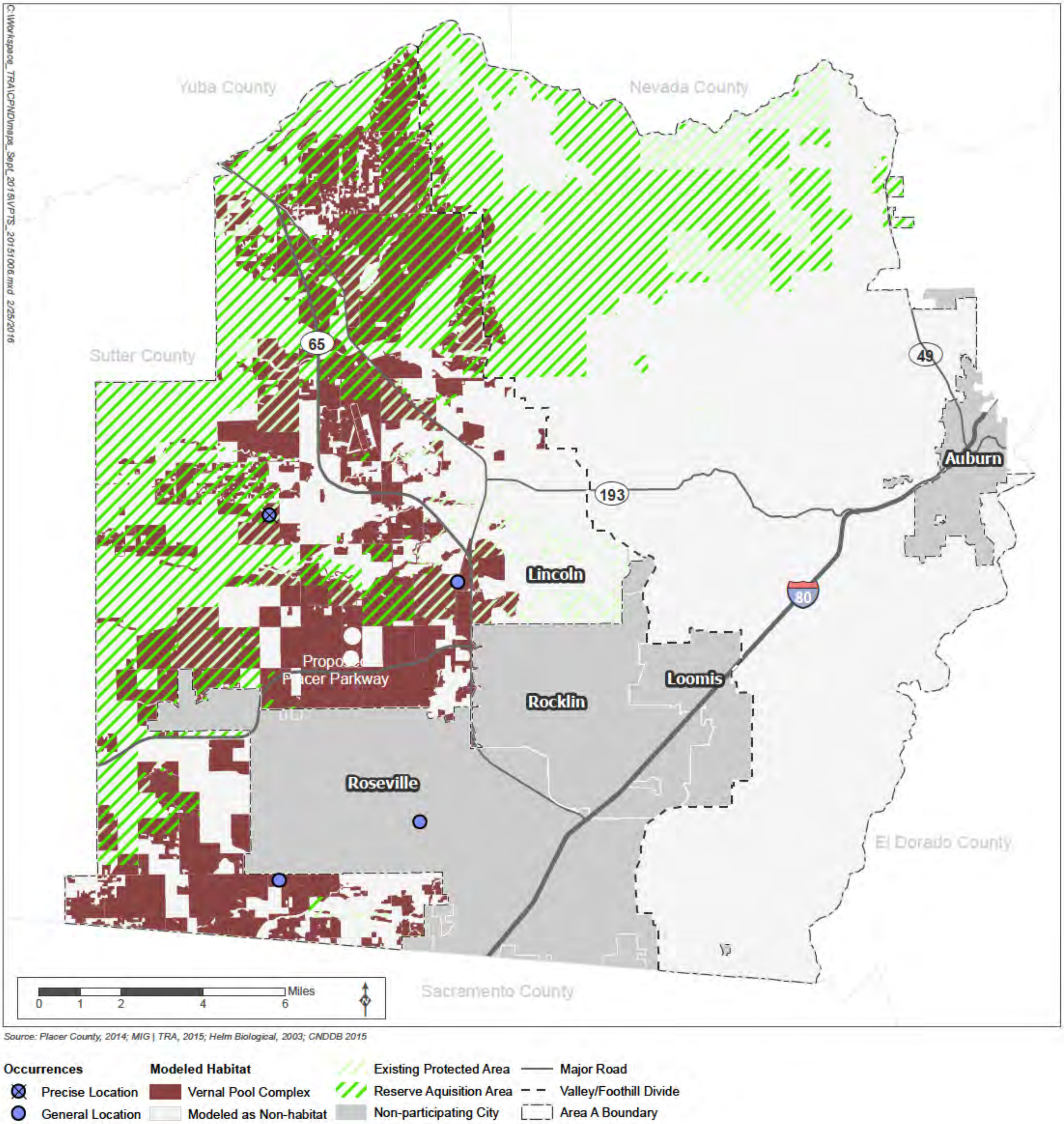
Dispersal

Dis1: Adults can disperse from pool to pool during periods of flooding caused by abundant rainfall, provided that there are appropriate pools to disperse to. Such dispersal is not very likely in small or highly fragmented vernal pool complexes.

Dis2: Cysts can be transported by the wind from dry pools; successful dispersal depends on wind speed and direction.

Dis3: Cysts also can be transported in the guts of waterfowl or amphibians. Success in this mode of dispersal depends on where the cysts are deposited. The chances of a cyst arriving in a suitable location are enhanced considerably in a large, unfragmented pool complex. Dis1 and Dis3 are facilitated by establishing large reserve areas and managing them primarily for biodiversity conservation.

Dis4: Cysts may be transported by livestock, attached to mud on their hooves. Successful dispersal in this manner would depend on livestock moving from pool to pool just as they were drying out. Because high densities of livestock are likely to result in excessive surface disturbance, adaptive grazing management within a reserve must consider all the costs and benefits of using livestock as dispersal agents.



Source: Placer County, 2014; MIG | TRA, 2015; Helm Biological, 2003; CNDDB 2015

MIG TRA

Species Map 13.

Vernal Pool Tadpole Shrimp Modeled Habitat Distribution and Occurrence

Placer County Conservation Program – Western Placer County HCP/NCCP

Conservancy Fairy Shrimp (*Branchinecta conservatio*)

© Larry Serpa

Status

Federal: Endangered (USFWS 1994)

State: None

Critical Habitat: Critical habitat has been designated for Conservancy fairy shrimp (USFWS 2003; USFWS 2005a). Critical habitat for Conservancy fairy shrimp is not present in the Plan Area.

Recovery Plan: Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005b). The Plan Area is within the Western Placer County Core Recovery Area (Zone 2) (USFWS 2005b).

Distribution

California

Conservancy fairy shrimp is endemic to California (Eng et al. 1990). Its historical range is the annual grasslands of the Central Valley. Currently, the species ranges from the Vina plains of Butte and Tehama counties south to the Grasslands Ecological Area in Merced County. A disjunct population occurs in the Los Padres National Forest in Ventura County (USFWS 2007; USFWS 2012).

Conservancy fairy shrimp has been reported from the following California vernal pool regions: northeast Sacramento Valley, Solano-Colusa, Livermore, San Joaquin Valley, South Sierra Foothills, and Santa Barbara (USFWS 2005). Currently, 10 populations of Conservancy fairy shrimp are known to be present in California, including Vina Plains in Butte and Tehama counties; Sacramento National Wildlife Refuge in Glenn County; Mariner Ranch in Placer County; Yolo Bypass Wildlife Area in Yolo County; Jepson Prairie in Solano County; Mapes Ranch in Stanislaus County; University of California Merced area in Merced County; Highway 165 in Merced County; Sandy Mush Road in Merced County; and Los Padres National Forest in Ventura County (USFWS 2012).

Placer County Plan Area

Historical

The historical distribution of Conservancy fairy shrimp is not known and can only be inferred from the historical distribution of its habitat (USFWS 2007). Annual grasslands of western Placer County, particularly within the Central Valley ecoregion, probably supported a patchy distribution of Conservancy fairy shrimp.

Current

There is a single occupied vernal pool with Conservancy fairy shrimp in the Plan Area, which is at the Mariner Conservation Bank within the Southeastern Sacramento Valley vernal pool region (USFWS 2007; Hemmen pers. comm.).

Population Status & Trends

California

Conservancy fairy shrimp are rare, and at the time of listing, six widely separated populations of this species were known (USFWS 1994). Due to its rarity and a lack of monitoring, little is known about current population status and trends. Extensive surveys for fairy shrimp throughout the range of Conservancy fairy shrimp have located five additional populations since the species was listed in 1994 (USFWS 2012). Currently, 10 populations of Conservancy fairy shrimp are known to be present in California.

Placer County Plan Area

There is one known occurrence of Conservancy fairy shrimp in the Plan Area. One male was observed in the spring of 2007 at the Mariner Conservation Bank, located west of the City of Lincoln on North Dowd Road (USFWS 2007; Hemmen pers. comm.). Additional surveys in 2008 and 2011 detected this species in higher numbers within the same vernal pool (Helm Biological Consulting 2011 as cited in USFWS 2012). However, to date, the species is still only present in a single vernal pool at the Mariner Conservation Bank. This locality is within the Western Placer County Core Recovery Area (Zone 2) identified in the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005b; USFWS 2012). There are multiple sites within this core area that are protected for the benefit of vernal pool species, including the Orchard Creek Vernal Pool Conservation Bank, Twelve Bridges Preserve, Sheridan Conservation Bank, and Yankee Slough Conservation Bank. Conservancy fairy shrimp have not been detected during fairy shrimp surveys at any of the other sites (USFWS 2012).

Natural History

The habitat requirements, ecological relationships, life history, and threats to Conservancy fairy shrimp described below are summarized in diagram form in the Envirogram 14 Conservancy Fairy Shrimp.

Habitat Requirements

Conservancy fairy shrimp inhabits rain-filled ephemeral pools (i.e., vernal pools) that form in depressions, usually in grassland habitats (Eng et al. 1990). Pools must fill frequently and persist long enough for the species to complete its lifecycle, which takes place entirely within vernal pools. Conservancy fairy shrimp inhabit alkaline pools, vernal pools, vernal swales, and other seasonal wetlands. The pools inhabited by Conservancy fairy shrimp, often referred to as playa pools, are usually large and often have turbid water (Vollmar 2002). Playa pools often remain inundated much longer than typical vernal pools (in some cases well into the summer) and can be identified by their large size (typically greater than 60 meters in diameter) (Vollmar 2002). These pools are found on different soil and geologic formations, including Peter's clay on the volcanic Tuscan formation in Tehama County and alluvial Pescadero clay Loam of the basin rim landform of Jepson Prairie. Occupied habitats range in size from claypan vernal pools as small as 36 square yards to large vernal pools up to 89 acres. The maximum potential water depth of occupied habitat ranges from 5 to 19 inches (Helm 1998; Eriksen and Belk 1999; USFWS 2007; California Natural Diversity Database 2009). Conservancy fairy shrimp are not found in riverine, estuarine, or other permanent waters that support fish or temporary non-vernal pool habitats such as roadside ditches or railroad toe-drains.

Reproduction

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

Following fertilization, embryonic and cyst development begin. Embryonic development ceases when the late gastrula stage is reached. At that point, metabolism slows and a halted embryo is isolated from the environment by development of a many-layered membranous shell. The embryo and the shell comprise the cyst, or resting egg. Females carry cysts in a brood sac. Cysts are dropped to the pool bottom or remain in the female's brood sac until the female dies. Cysts are capable of withstanding heat, cold, and prolonged desiccation. When occupied pools fill with water in the same or subsequent seasons, some, but not all, of the deposited cysts may hatch. The egg bank in the soil may comprise cysts from several years of breeding. When the vernal pools fill with rainwater and the water temperature drops below 50ºF, the resting eggs hatch into small nauplii. The early stages of Conservancy fairy shrimp develop rapidly into adults, reaching maturity in as little as 19 days (Eriksen and Belk 1999; Helm 1998).

Dispersal Patterns

Conservancy fairy shrimp disperse locally during extremely wet years, when individual pools in a complex spill into or are connected with adjacent pools. Long-distance dispersal can result from cysts being carried on the wind and on the bodies or in the intestines of larger animals. Cysts, including those still in brood sacs, can pass undamaged and undigested through the digestive tracts of birds (Proctor et al. 1967 cited in Eriksen and Belk 1999); subsequent deposition of fecal matter can result in the inoculation of a new site. Cysts trapped in mud can adhere to the feet and feathers of waterfowl and the hooves and fur of grazing mammals and be transported to the dried mud of different vernal pool complexes (Eriksen and Belk 1999). Cysts may also be transported between pools in the digestive tracts of amphibian predators such as frogs and salamanders (Rogers pers. comm.). However, due to the size and isolated nature of the existing populations in California, opportunities for recolonization are low (USFWS 2012).

Longevity

Conservancy fairy shrimp can achieve maturity in as few as 19 days after hatching. Based on laboratory observations, Helm (1998) determined that Conservancy fairy shrimp has a mean longevity of 114 days and that it takes an average of 36 days for Conservancy fairy shrimp to reach maturity. Field observations indicate that Conservancy pool fairy shrimp typically persist 10–12 weeks (Eriksen and Belk 1999; Vollmar pers. comm; Helm pers. comm.).

Sources of Mortality

The greatest sources of mortality to Conservancy fairy shrimp are predation and heatstroke. In general, Conservancy fairy shrimp are unable to filter oxygen from their aquatic habitat when water temperatures remain above 70ºF (Eriksen and Belk 1999). In addition, both adult Conservancy fairy shrimp and diapausing cysts can be crushed by foot traffic and off-highway vehicles (Hathaway et al. 1996).

Behavior

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

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

Movement and Migratory Patterns

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

Ecological Relationships

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

Conservancy fairy shrimp co-occur with vernal pool fairy shrimp (*Branchinecta lynchi)*, California fairy shrimp *(Linderiella occidentalis*), and the vernal pool tadpole shrimp (*Lepidurus packardi*) (King et al. 1996, Helm 1998, Eriksen and Belk 1999). In general, Conservancy fairy shrimp have a large population within a given pool, and is usually the most abundant fairy shrimp when more than one species is present (Helm 1998, Eriksen and Belk 1999). The Conservancy fairy shrimp also co-occurs with several plants found in large vernal pools including Colusa grass (*Neostapfia colusana)* and various Orcutt grass species.

Threats

The greatest threats to the persistence of Conservancy fairy shrimp are habitat loss and degradation resulting from urban development and agriculture. Vernal pools occur in large, flat, open grasslands that are ideal for a number of economic uses, including airports, military bases, rice and grain fields, cattle grazing, aggregate mining, and urban development. Habitat loss is generally the result of agricultural conversion from rangelands to intensive farming, urbanization, aggregate mining, infrastructure (e.g., road and utility) projects, and recreational activities (USFWS 2007). Habitat fragmentation also limits habitat when vernal pools are broken into smaller groups or individual vernal pools and become isolated from each other as a result of human activities (e.g., road development) (USFWS 2005b, 2007). Invasive species, such as perennial pepperweed (*Lepidium latifolium*), also result in loss of vernal pool habitat. Climate change is expected to have an effect on vernal pool hydrology through changes in the amount and timing of precipitation inputs and the rate of loss through evaporation (USFWS 2012). These changes in hydrology will likely affect fairy shrimp species because they are obligate aquatic organisms with life histories dependent on certain hydrologic conditions (Pyke 2005). The suitability of vernal pools for fairy shrimp depends in large part on the timing and duration of wetland inundation since these species are dependent on vernal pools that have sufficient water to remain wet throughout the reproductive phase of the species (USFWS 2012).

Context for a Regional Conservation Strategy

Conservancy fairy shrimp is known from one occurrence in the Plan Area and may exist in additional locations that have not been surveyed. One male was observed in the spring of 2007 at the Mariner Conservation Bank, located west of the City of Lincoln on North Dowd Road (USFWS 2007; Hemmen pers. comm.). Additional surveys in 2008 and 2011 detected this species in higher numbers within the same vernal pool (Helm Biological Consulting 2011 as cited in USFWS 2012). However, to date, the Mariner Conservation Bank is still only present in a single vernal pool. This locality is within the Western Placer County Core Recovery Area (Zone 2) identified in the Recovery Plan for Vernal Pool Ecosystems of California and Southern Oregon (USFWS 2005b; USFWS 2012). There are multiple sites within this core area that are protected for the benefit of vernal pool species, including the Orchard Creek Vernal Pool Conservation Bank, Twelve Bridges Preserve, Sheridan Conservation Bank, and Yankee Slough Conservation Bank. Conservancy fairy shrimp have not been detected during fairy shrimp surveys at any of the other sites (USFWS 2012).

Conservancy fairy shrimp is sparsely distributed in playa vernal pool complexes north and south of Placer County. Currently, 10 populations of Conservancy fairy shrimp are known to be present in California, including Vina Plains in Butte and Tehama counties; Sacramento National Wildlife Refuge in Glenn County; Mariner Ranch in Placer County; Yolo Bypass Wildlife Area in Yolo County; Jepson Prairie in Solano County; Mapes Rach in Stanislaus County; University of California Merced area in Merced County; Highway 165 in Merced County; Sandy Mush Road in Merced County; and Los Padres National Forest in Ventura County (USFWS 2012).

Modeled Species Distribution in the Plan Area

Species Map 14. *Conservancy Fairy Shrimp Occurrence and Vernal Pool Complex* does not model habitat for Conservancy fairy shrimp because its known distribution is highly restricted in the Plan Area to a single vernal pool and because the type of vernal pool this species typically occurs in (e.g., generally large and turbid pools; Helm 1998; USFWS 2007) is not found in the Plan Area.

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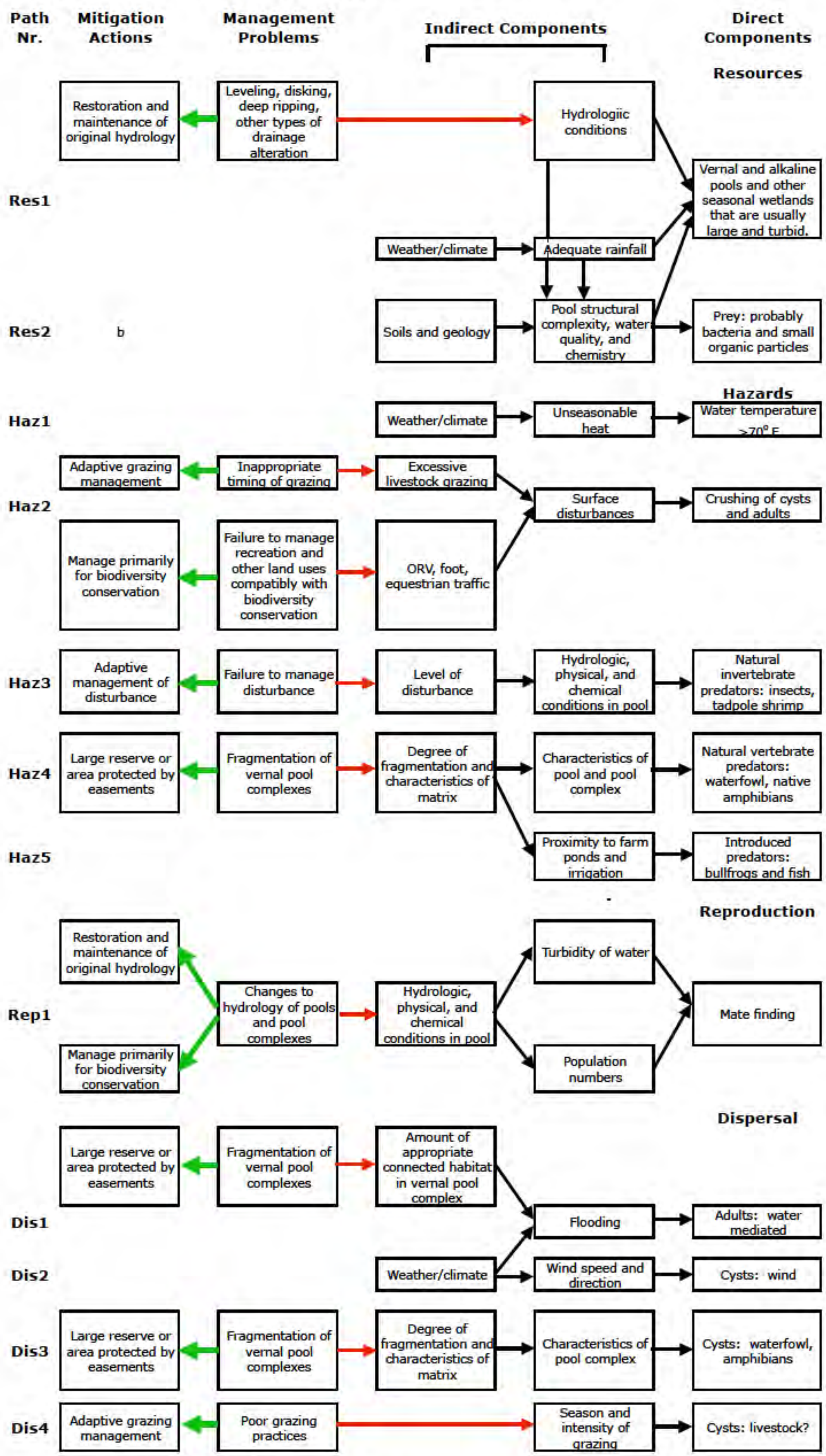
Envirogram 14 Conservancy Fairy Shrimp, *Branchinecta conservatio*



Envirogram 14 Conservancy Fairy Shrimp. Key to abbreviations: Res = Resources; Haz = Hazards; Rep = Reproduction; Dis = Dispersal.

Envirogram Narrative

Conservancy Fairy Shrimp (*Branchinecta conservatio*)

The envirogram was created based on the information provided in the species account. The envirogram is a tool to depict and organize the most important ecological factors that affect a population or group of populations of a particular species. The envirogram consists of Direct Components – components of the environment that directly affect a species' chances to survive and reproduce, and several webs comprised of distal factors (i.e., Indirect Components, Management Problems, and Mitigation Actions) that act in sequence to affect the Direct Components. The Direct Components consist of four major categories: resources, hazards, reproduction, and dispersal. Each of these is subdivided as necessary.

The webs identify the underlying ecological processes or human actions that influence each Direct Component. Distal factors in the web activate proximate components. Each of these pathways in the web are constructed from right to left, with Indirect Components immediately to the left of Direct Components directly affecting the Direct Component, and secondary Indirect Components affecting primary Indirect Components. Management Problems can directly affect the Indirect Components, and Mitigation Actions provide solutions to remedy the Management Problems.

Resources

Res1: Conservancy fairy shrimp are most commonly found in vernal pools, alkaline pools, and other seasonal wetlands that are large and turbid. Such waters are usually associated with natural hydrologic conditions; waters that have been modified by leveling, disking, deep ripping, and other types of drainage alterations are generally not suitable, and such water bodies must be restored to their natural hydrologic conditions to create habitat for this species. Adequate rainfall, a function of weather and climate, is necessary to fill the pools to the necessary depth.

Res2: Conservancy fairy shrimp feed on bacteria and small organic particles. The abundance and diversity of prey items depend on the structural complexity of the pool and its water quality and chemistry, which in turn are influenced by the soils and geological formations in which the pool occurs as well as by hydrologic conditions and the amount and timing of rainfall.

Hazards

Haz1: Conservancy fairy shrimp are killed by water temperatures >70^{o}F, which can occur during periods of unseasonable heat. (A warming climate with an increasing frequency of extreme weather events could result in increasing problems of this kind in the future).

Haz2: Crushing of cysts in dry pools result from surface disturbances such as livestock grazing and ORV, foot, or equestrian traffic; adults also may be crushed by livestock while the pools are still partially filled. Management primarily for biodiversity conservation and managing grazing within this context are the best mitigation strategies for these hazards.

Haz3: The abundance of natural invertebrate predators such as insects and tadpole shrimp depends on the hydrologic, physical, and chemical conditions in a pool. Excessive disturbance can create conditions that create unnaturally high densities of these predators.

Haz4: Natural vertebrate predators of Conservancy fairy shrimp probably include waterfowl and native amphibians. The presence of these species depends upon the characteristics of the individual pool and the pool complex, which in turn are determined by the degree of fragmentation of the complex and the characteristics of the surrounding area. Fragmentation and location of the pool complex may result in abnormally high or low densities of these predators in certain pools, which could be a benefit or a disaster to a Conservancy fairy shrimp population.

Haz5: If pool complexes are located near farms or irrigation structures, introduced predators such as bullfrogs and fish could be introduced into a pool, which inevitably would result in local extirpations. Large, unfragmented pool complexes, located well away from farm ponds or irrigation ditches and managed primarily for biodiversity conservation are the best management option to control these hazards.

Reproduction

Rep1: Successful reproduction in Conservancy fairy shrimp probably depends on finding mates, which in largely dependent upon the turbidity of the water and the numbers of individuals in a pool. Both of these factors are related to the hydrologic, physical, and chemical conditions in the pool. Any alteration to the hydrology of the pool or pool complex can make conditions unsuitable for reproduction. Restoring the original hydrology and managing a pool complex primarily for biodiversity conservation is the best way to preserve the conditions needed for reproduction.

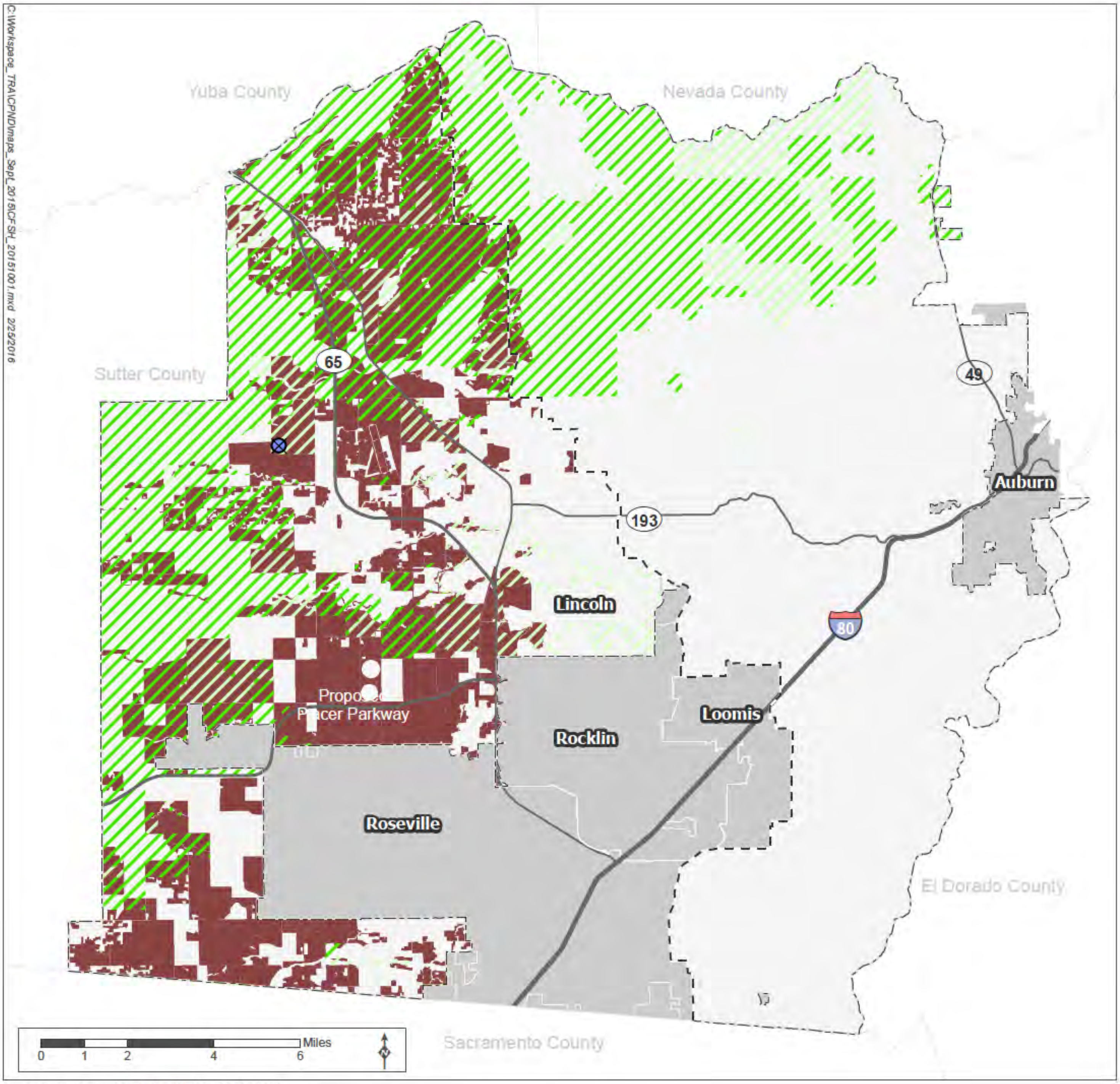
Dispersal

Dis1: Adults can disperse from pool to pool during periods of flooding caused by abundant rainfall, provided that there are appropriate pools to disperse to. Such dispersal is not very likely in small or highly fragmented vernal pool complexes.

Dis2: Cysts can be transported by the wind from dry pools; successful dispersal depends on wind speed and direction.

Dis3: Cysts also can be transported in the guts of waterfowl or amphibians. Success in this mode of dispersal depend on where the cysts are deposited. The chances of a cyst arriving in a suitable location are enhanced considerably in a large, unfragmented pool complex. Dis1 and 3 are facilitated by establishing large reserve areas and managing them primarily for biodiversity conservation.

Dis4: Cysts are possibly transported by livestock, attached to mud on their hooves. This event would depend on livestock being in the right place at the right time and in densities that are not likely to result in excessive surface disturbance. Adaptive grazing management within a reserve must consider all these factors.



Source: Placer County, 2014; MIG | TRA, 2015; CNDDB 2015

Occurrences			
Precise Location	Vernal Pool Complex	Existing Protected Area	Major Road
	Non-habitat	Reserve Aquisition Area	Valley/Foothill Divide
		Non-participating City	Area A Boundary

MIG TRA

Species Map 14.

Conservancy Fairy Shrimp Occurrence and Vernal Pool Complex

Placer County Conservation Program – Western Placer County HCP/NCCP