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STATE OF CALIFORNIA

STATE ENERGY RESOURCES CONSERVATION AND DEVELOPMENT COMMISSION

IN THE MATTER OF: WILLOW ROCK ENERGY STORAGE CENTER	Docket No. 21-AFC-02

COMMENTS OF CALIFORNIA UNIONS FOR RELIABLE ENERGY ON THE PRELIMINARY STAFF ASSESSMENT

ATTACHMENT D (7 OF 7)

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OREGON SPOTTED FROG

Rana pretiosa Baird and Girard 1853b

Status Summary

Rana pretiosa is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 0.82 (82/100). During the previous evaluation, it was also designated as a species of special concern (Jennings and Hayes 1994a) and it was listed as federally Threatened in 2014 (USFWS 2014). We are aware of only two unverified site records for this species in California in the last 25 years.

Identification

Dorsally, *Rana pretiosa* is a dark-brown, reddish, or greenish frog with black spots or blotches (McAllister and Leonard 1997). The dorsal blotching is usually irregular around the edges, rather than sharply demarcated, and has a small light spot in the center of the larger spots. The venter is usually mottled and has a base color that changes from cream white at the chin to orange more ventrally (Dunlap 1955, Stebbins 2003). The ventral coloration often appears to be superficial or "painted on" (Dunlap 1955, Nussbaum et al. 1983). Like many California ranids, this species has a prominent light stripe below the eye (particularly so in juveniles) and thin dorsolateral ridges that

Oregon Spotted Frog: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	0
v. Ecological tolerance (10)	7
vi. Population trend (25)	25
vii. Vulnerability to climate change (10)	Data deficient
viii. Projected impacts (10)	10
Total Score	82
Total Possible	100
Total Score/Total Possible	0.82

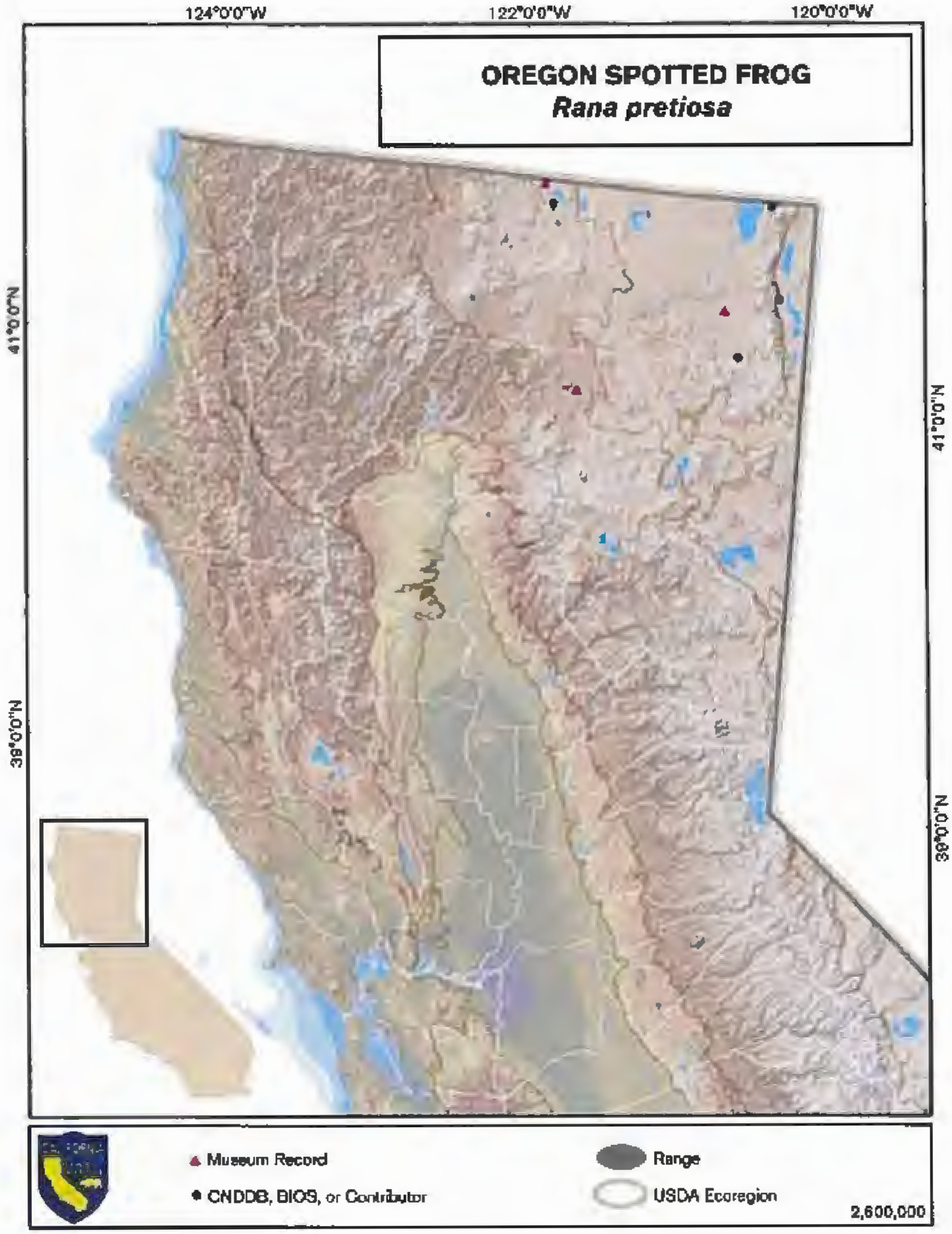


PHOTO ON PREVIOUS PAGE: Oregon spotted frog, Lane County, Oregon. Courtesy of Troy Hibbitts.

dissolve into a series of raised dots two-thirds to three-quarters of the way down the back. The call consists of a series of faint clicks, repeated roughly seven times in rapid succession (Briggs 1987, Stebbins 2003, Elliott et al. 2009).

Within its California range, this species is most likely to be confused with the Cascades frog (*R. cascadae*). Although similar, *R. cascadae* spots tend to have sharply defined edges, no light centers, and appear to be on the surface of the skin, reminiscent of black ink being splattered on the frog (Stebbins 2003). In addition, the underside of the legs are yellow tan in *R. cascadae* (reddish in *R. pretiosa*), the eyes are oriented dorsally when viewed from above in *R. presiosa* (oriented outwardly in *R. cascadae*), and *R. pretiosa* has full, rather than partial webbing between the toes of the rear legs. The Columbia spotted frog (*R. luteiventris*) may also occur in California, and it could also be confused with *R. pretiosa* (see the "Distribution" section).

Taxonomic Relationships

Green et al. (1996, 1997) divided *Rana pretiosa* into two species, *R. pretiosa* and *R. luteiventris*, based on morphology and allozyme variation. The two taxa are morphologically similar (usually distinguishable in the field based on the ventral mottling in *R. pretiosa*; M. Hayes, pers. comm.), but preserved specimens can usually be differentiated with a series of head measurements (Green et al. 1997). The two species are also diagnosable using allozymes (Green et al. 1996) and mitochondrial DNA cytochrome-*b* sequence (Funk et al. 2008).

Life History

No data on life history of California populations exist and much of the data from elsewhere in the range occurred before the partitioning of *Rana pretiosa* and *R. luteiventris*. As California populations of *R. pretiosa* are at the extreme southern edge of the species' range, the timing of life history events may occur earlier relative to those reported from more northerly sites, although the high elevation of California sites may compensate for any potential latitudinal gradient. California populations were geographically closest to Oregon frogs from the Klamath basin, and those populations may serve as the best models for California.

Frogs emerge from hibernation as soon as the winter thaw permits (Stebbins 2003) and water temperatures rise to about 6°C (C. Pearl, pers. comm.). *Rana pretiosa* breeds explosively soon after emergence, usually over a 1- or 2-week period. Males often congregate in shallow water and begin to call (Licht 1969, Nussbaum et al. 1983). Egg masses are deposited together in large groups in vegetated margins of large permanent aquatic habitats, usually at the high-water mark. The species can experience high egg mass mortality when waters recede rapidly, leading to stranding, desiccation, and/or freezing (Licht 1971, Briggs 1987). However, eggs from multiple sites in Oregon were found to resist near-freezing temperatures as long as they remained beneath the water surface (Bowerman and Pearl 2010). Artificially incubated egg masses hatch in as few as 72 hours to as many as 400 hours, depending on temperature (25°C and 10°C, respectively), followed by metamorphosis in approximately 4 months (Licht 1971).

Males appear to have lower survivorship than females, presumably due to the longer periods of time that they spend in breeding congregations and the resulting exposure to predation (Licht 1974, Chelgren et al. 2008). Post-metamorphic frogs consume a wide variety of invertebrate prey including insects, occasional mollusks, and crustaceans, as well as small vertebrates including anurans (Nussbaum et al. 1983, Licht 1986b, Pearl and Hayes 2002, Pearl et al. 2005b).

Habitat Requirements

Information on habitat utilization in California is very limited, although habitat requirements are better studied elsewhere in the range. The species appears to seasonally use different habitat types (Watson et al. 2003, Chelgren et al. 2008). *Rana pretiosa* is highly aquatic and rarely found away from the water (Licht 1986a). It frequently uses temporary pools, ditches, and other shallow water sources, but nearby deep permanent water is always required and serves as a refuge for adult frogs during dry parts of the year and during drought (McAllister and Leonard 1997, Watson et al. 2003). Breeding occurs in shallow water with aquatic vegetation (Licht 1971, Watson et al. 2003). In Oregon, oviposition sites occurred, on average, 14.1 m (range 0.08–35.0 m) from the shore in water that was 18.5 cm deep (range 1–57 cm) (Pearl et al. 2009). At one site in Washington, the species overwintered in shallow water, where it buried itself at the base of emergent plants (Watson et al. 2003). Overwintering in flowing springs has also been documented (Chelgren et al. 2008). Overland dispersal appears to be quite limited, and the species may require habitat where the shallow-water breeding and overwintering habitats are connected to deep-water refuge habitat by intervening water during early spring and late fall to allow inter-habitat migrations (Watson et al. 2003).

The habitat requirements for *R. pretiosa* have likely contributed to its declines. The diversity of habitat types that are used, coupled with the requirement that they are connected by intervening stretches of water, is fairly specific and is probably only common in large, relatively intact wetland complexes. These complexes are becoming increasingly rare throughout the species' range as landscapes are drained and converted to agriculture and grazing.

Data are limited on effects of grazing on this species. At one site in western Washington where reed canarygrass (*Phalaris arundinacea*) forms dense stands, Watson et al. (2003) suggested that grazing could help open patches and make them suitable for *R. pretiosa*. However, grazing also has the potential to reduce water quality and cover from predators. Additional work is needed on how the timing and intensity of grazing affect frog behavior and habitat use.

Distribution (Past and Present)

Few localities for *Rana pretiosa* have been documented in California, and all known localities appear to be extirpated. Historically, *R. pretiosa* occurred in the northeastern corner of California, ranging south to Plnmas and Tehama Counties and west to the eastern portions of Sikiyou, Shasta, and Tehama Counties (Slevin 1928). Within this range, the species has been found in scattered localities in Modoc, Shasta, and Siskiyou Counties (Stebbins 1972, Jennings and Hayes 1994a), with the last documented record occurring in a woodpile in Cedarville, Modoc County, in 1989 (Jennings and Hayes 1994a). This last record is somewhat anomalous, since the frog was found in a heavily modified area near the town center of Cedarville, in habitat that seems to be unsuitable for the frog. Given the very specific habitat requirements of *R. pretiosa*, the fact that no specimen from the site was ever examined by a herpetologist and no vouchers exist, it is possible that this is a misidentified or human-introduced specimen (L. Groff, pers. comm.; M. Hayes, pers. comm.). It remains possible that isolated populations still persist, particularly in remote portions of the Warner Mountains and on private land in Surprise Valley, Modoc County. Fairly recent surveys in the Warner Mountains, Modoc Plateau, and Pitt River drainage failed to locate any individuals (Jennings and Hayes 1994a, Groff 2011). There is an unverified sighting of a "spotted frog" in Surprise Valley from November 2008 (L. Gray, pers. comm.), but a follow-up survey at this locality revealed only *Psuedacris regilla*. A more recent survey comprising 18 localities selected using a species distribution model for this species did not detect *R. pretiosa* in California (Groff 2011), although the southernmost extant locality in Oregon is only about 10 km from the state border. Between 2012 and 2013, USFWS biologists conducted additional surveys at 12 sites within the Pit River watershed and Warner Mountains. Again, no evidence of *R. pretiosa* was found (USFWS-Klamath Falls Field Office, unpublished data, 2013).

Outside of California, *R. pretiosa* is patchily distributed from extreme southwestern British Columbia, south through Washington and

Oregon (Green et al. 1997). This distribution is fragmented, and the species has undergone severe declines through most of its range (McAllister et al. 1993, Green et al. 1997). Declines are thought to have occurred disproportionately in lowland areas, and over two-thirds of the remaining populations occur along the crest and eastern slopes of the Cascade Range (Pearl et al. 2009).

It is possible that some *R. pretiosa* in California, particularly those east of the Warner Mountains in Modoc County, could actually be *R. luteiventris*. There are known *R. luteiventris* populations approximately 16 km north of the California border on the eastern slopes of the Warner Mountains, making the presence of *R. luteiventris* in California plausible (Funk et al. 2008; M. Hayes, pers. comm.). However, the species has not been documented in California.

Trends in Abundance

No abundance data for California populations exist. Reports from parts of the Willamette Valley, Oregon, and Puget Lowlands, Washington, suggest that *Rana pretiosa* was common in those areas around the 1930s. Declines are thought to have been occurring for a large part of the twentieth century (Dumas 1966, McAllister et al. 1993, Pearl and Hayes 2005). At one time, the species was apparently common in Warner Valley, Oregon, immediately north of Surprise Valley in California (Cope 1883). Any remaining populations in California are likely to be isolated and on private land that has not been surveyed. A recent species distribution model generated a set of potential sites, some of which were surveyed, but no California populations were found (Groff 2011).

Nature and Degree of Threat

At least four major factors have likely contributed to the decline of *Rana pretiosa* in California. First, the species has been strongly impacted by the loss of the extensive wetland complexes that were once common in northern California. As land has been drained and modified for livestock grazing and agriculture, the overall amount of available acreage that provides the precise suite of habitat types used by this species has declined. This loss of wetland habitat is further exacerbated by climate projections for northeastern California, which predict increasing temperatures, strongly decreasing precipitation, and reduced snowpack (PRBO 2011); all of these changes will reduce permanent wetlands and place increasing demands on the remaining aquatic habitat. Second, *R. pretiosa* appears to be sensitive to relatively low levels of nitrates and nitrites resulting from agricultural runoff (i.e., those meeting EPA allowances for drinking water; Marco et al. 1999). This observation is consistent with the precipitous declines observed in lowland Oregon and Washington populations, which have been more heavily impacted by agriculture than higher-elevation populations. Application of the pesticide DDT was also correlated with die-offs in the closely related *R. luteiventris* in northern Oregon (reported as *R. pretiosa*; Kirk 1988). Third, the species appears to be sensitive to introduced exotic predators, particularly bullfrogs and exotic fishes. Some data indicate that it is likely more sensitive to the presence of bullfrogs than other native ranid frogs. In areas where *R. aurora* and *R. pretiosa* are sympatric, stronger declines were observed in *R. pretiosa* than *R. aurora* in areas where bullfrogs have invaded (Pearl et al. 2004). Laboratory experiments also demonstrate a differential impact of bullfrogs on *R. pretiosa* relative to *R. aurora*, likely due to *R. pretiosa*'s more strongly aquatic life history (Pearl et al. 2004). Bullfrogs have also been hypothesized to negatively impact small *R. pretiosa* populations via reproductive interference (Pearl et al. 2005c). In combination with the well-documented effects of nonnative fishes on western ranid frogs (Adams 1999, Lawler et al. 1999, Adams 2000, Joseph et al. 2011), this suite of nonnative predators is likely to have a strong negative effect on *R. pretiosa* populations. Finally, *Bd* has been found to be present in remaining populations of *R. pretiosa* (Pearl et al. 2007, Hayes et al. 2009), although experimental work suggests that the species may be resistant (Padgett-Flohr and Hayes 2011). However, given the importance of *Bd* in some anuran declines, further work on its impact on *R. pretiosa* is warranted.

Given the rarity of *R. pretiosa* records from California and our lack of historical population parameters, it is impossible to differentiate between these causes. However, it is reasonable to assume that several or all of these factors were involved in the decline of the species in California.

Status Determination

The limited California range of *Rana pretiosa* and its apparent extirpation from the few known historic localities are the main drivers for its high score. The paucity of historical records in California suggests that this taxon may have historically been rare in the state, and its specialized ecological requirements (large permanent wetlands, specialized sub-habitats for breeding, hibernation, and growth) make it inherently sensitive to declines. Together, these factors justify a Priority 1 designation for this species.

Management Recommendations

Ongoing management efforts for this species should be coordinated through the range-wide conservation strategy that the Washington Department of Fish and Wildlife is leading and the California Department of Fish and Wildlife is participating in (B. Bolster, pers. comm.). Cushman and Pearl (2007) recently assessed *Rana pretiosa* conservation needs and provided a detailed roadmap for management of this species. Our recommendations largely follow theirs. If the surveys outlined below identify any remaining populations of this species in the state, the wetland habitat supporting the population should be protected from fragmentation and modification, including the introduction of exotic fishes and amphibians. Captive populations of this species should also be established to serve as assurance colonies, should the last wild populations go extinct. If continued surveys suggest that the species is extirpated from California, captive breeding and reintroduction programs could be initiated with Oregon animals if appropriate habitat can be identified and protected. Given the very high levels of genetic differentiation and population structure found among extant Oregon and Washington populations (Blouin et al. 2010), populations from the southern Klamath Basin genetic unit are probably the best candidates for such a reintroduction in California. Beyond these two steps, effective management of this taxon in California will require additional research into the causes of decline.

Monitoring, Research, and Survey Needs

Comprehensive surveys throughout *Rana pretiosa*'s known historic range should be conducted to determine if any populations persist in the state. Surveys of remaining large wetland complexes are particularly important, as are surveys of potential habitat on private property. A recent species distribution model (Groff 2011) identified and surveyed some, but not all, of the predicted localities that may support this species in California, and this study provides an excellent starting point for additional surveys. Significant habitat that has not yet been surveyed remains on private property, particularly east of the Warner Mountains (although *R. luteiventris* may replace *R. pretiosa* in this area). The aforementioned recent surveys made a particular effort to gain access to private land, but permission was only granted in approximately 15% of cases (Groff 2011). Future surveys should continue to build partnerships with private stakeholders and survey large wetland complexes on private lands. If any populations are found, nonlethal tissue samples should be collected so that species identification can be verified with molecular data.

Should any populations be located, a monitoring program in conjunction with life history research should immediately be initiated with the goal of quantifying population sizes and connectivity (if multiple adjacent populations are found) and to allow for a better understanding of habitat requirements and causes of decline in this species. Molecular genetic studies using microsatellite and/or single nucleotide polymorphism data from multiple nuclear markers can provide valuable insights into historical population declines/expansions and should be conducted if any native populations are discovered. In addition, given the very high levels of population structure found among extant Oregon and Washington populations, any California populations should be surveyed for genetic variation and integrated into the existing species-wide genetic dataset (Blouin et al. 2010).

LOWLAND LEOPARD FROG

Rana yavapaiensis Platz and Frost 1984

Status Summary

Rana yavapaiensis is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 74% (63/85). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a). *Rana yavapaiensis* has not been confirmed to occur in California since 1965 (Jennings and Hays 1994a).

Identification

Rana yavapaiensis is a medium-sized ranid frog (4.6–8.7 cm SVL) with prominent dorsolateral folds that are discontinuous and angle inward posteriorly (Platz and Frost 1984). The coloration is variable, but is generally gray green, gray brown, or tan with irregular blotches above and cream or white on the venter. The ventral pelvic region is yellow, and this sometimes extends onto the legs. In older individuals, there is also dark mottling on the chin (Jennings and Hayes 1994a; Stebbins 2003). A cream-colored supralabial stripe is present that fades anteriorly in front of the eye (Platz and Frost 1984).

In California, this frog is most likely to be confused with the closely related, nonnative Rio Grande leopard frog (*R. berlandieri*). The distinguishing characters for the two species widely overlap, and positive identification is therefore

Lowland Leopard Frog: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	0
v. Ecological tolerance (10)	3
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	10
Total Score	63
Total Possible	85
Total Score/Total Possible	0.74

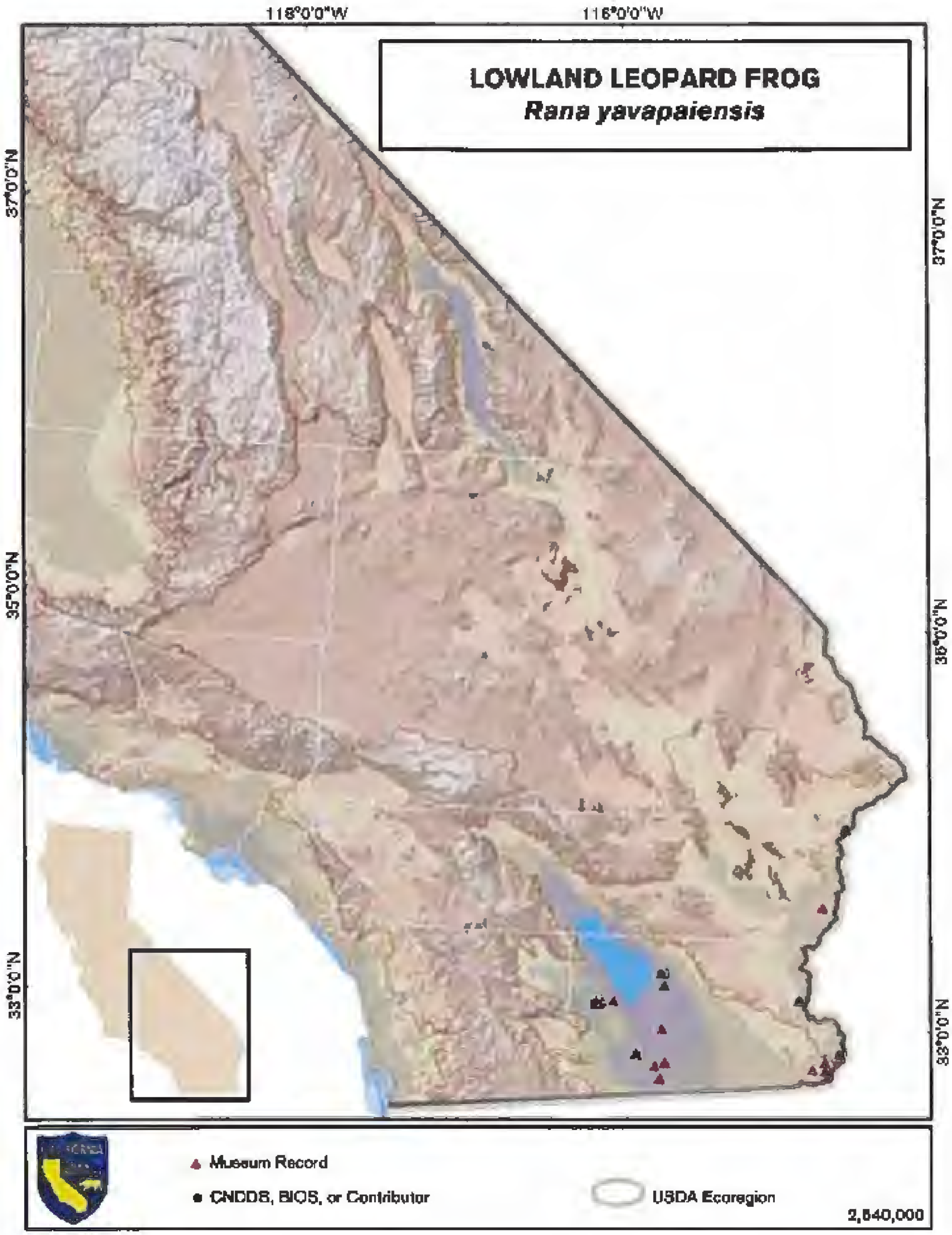


PHOTO ON PREVIOUS PAGE: Lowland leopard frog, Cochise County, Arizona. Courtesy of Brian Freiermuth.

difficult. *Rana berlandieri* attains larger body sizes (up to 11.4 cm SVL) and has proportionately larger eyes than *R. yavapaiensis*. Coloration of the two species is similar, but *R. yavapaiensis* generally has more extensive reticulation between the blotches on the hind legs, and its ventral coloration is often less dusky than *R. berlandieri* (Stebbins 2003). *Rana berlandieri*'s call consists of a low trill often followed by grunts, whereas *R. yavapaiensis* calls with higher-pitched notes that are given in rapid succession, often followed by lower-pitched chucks (Stebbins 2003, Elliott et al. 2009). Given that there are no known extant *R. yavapaiensis* localities remaining in California and that it is similar in appearance to the nonnative species *R. berlandieri*, positive identifications should be made cautiously. The species are readily distinguishable using molecular data (Hillis and Wilcox 2005, Frost et al. 2006a), which should be used to confirm any potential *R. yavapaiensis* specimens from California.

Taxonomic Relationships

Rana yavapaiensis was recognized as a distinct species in the leopard frog complex primarily on the basis of morphology, reproductive isolation, and allozyme variation (Platz and Platz 1973, Platz 1976, Platz and Frost 1984). The species is morphologically similar to other species of leopard frogs in the southwest. Jaeger et al. (2001) distinguished relict leopard frogs (*R. onca*) from *R. yavapaiensis* using genetic and morphological data. Based on a mitochondrial DNA dataset, Hillis and Wilcox (2005) confirmed a close relationship between these two species to the exclusion of other leopard frog taxa, including several geographically nearby members of the complex.

Frost et al. (2006a) recommended placing this species and many other North American ranids in the genus *Lithobates*, although this proposal and the analyses that support it are controversial (Crother 2009, Frost et al. 2009a, Pauly et al. 2009). We retain the traditional taxonomy here to maintain stability and pending further analyses.

Life History

Life history characteristics of California populations of *Rana yavapaiensis* are poorly known. The species apparently breeds opportunistically during winter rains (Stebbins 1972), and breeding has been documented to occur from late December through March in California (Storer 1925, Ruibal 1959). Elsewhere in the range, breeding has been documented from October to April (Platz and Platz 1973, Collins and Lewis 1979, Frost and Platz 1983, Sartorius and Rosen 2000). The reproductive biology of *R. yavapaiensis* has only been studied in Arizona. There, the species is known to experience at least two reproductive peaks within a year (once in the fall, once in the winter or spring), and tadpoles may overwinter (Collins and Lewis 1979, Sartorius and Rosen 2000). However, some authors have observed among-population variation in the occurrence of multiple breeding peaks, and it is unknown whether California populations had one or two breeding peaks per year.

Rana yavapaiensis undergoes marked year-to-year fluctuations in population size throughout its range (Clarkson and Rorabaugh 1989, Sredl et al. 1997, Sartorius and Rosen 2000), which renders isolated populations susceptible to extirpation. This also makes it difficult to confirm the absence or extirpation of populations with single-year surveys, emphasizing the importance of multiyear surveys for this species.

Habitat Requirements

Habitat requirements for *Rana yavapaiensis* are poorly understood, particularly in California. The species was historically found in slow-moving water along the San Felipe Creek drainage and the Lower Colorado River (Storer 1925, Stebbins 1972). The species has been found predominantly in marshy areas with bulrushes, cattails, and grasses with a willow overstory (Storer 1925, Jennings and Hayes 1994a, Jennings and Hayes 1994b), but it is unknown whether this vegetation type is required for population persistence. The species also expanded into artificial canals and ditches in the Imperial Valley as agriculture developed in the region (Storer 1925, Klauber 1934), as is the case currently for *R. berlandieri* in Imperial County. It is unknown whether *R. yavapaiensis* can persist in these artificial habitats or whether they represent non-sustaining sink habitat requiring immigrants from nearby source populations.

Aquatic dissolved salt levels probably limit the distribution of this species, at least in some situations. Ruibal (1959) examined salt tolerance in adults and eggs from the San Felipe Creek drainage and found that salinities observed throughout most of the drainage were lethal to eggs (though not to adults) and that suitable areas for breeding were limited to the springs and seeps that fed the drainage. Whether salt concentration was always a limiting factor in California, or agricultural practices led to unnaturally high salt levels in some water bodies, is unknown.

Distribution (Past and Present)

No extant populations are presently known in California (Jennings and Fuller 2004). The distribution of *Rana yavapaiensis* was historically patchy, even before recent declines. In California, the species was historically present in suitable habitat along the Lower Colorado River, the Imperial Valley, and the San Felipe Creek drainage (Platz 1988, Stebbins 2003). Outside of California, the species historically ranged along the Lower Colorado River from northern Mexico to Arizona, from near sea level to 1700 m (Platz and Frost 1984, Platz 1988, Jennings and Hayes 1994a, Jennings and Hayes 1994b, Stebbins 2003). The last confirmed record in California is from 1965 in an irrigation ditch east of Calexico, Imperial County (Jennings and Hayes 1994a).

Trends in Abundance

Severe declines have occurred throughout the known California range of *Rana yavapaiensis*, and currently there are no known extant populations. Repeated surveys since 1965 have failed to locate this species (Vitt and Ohmart 1978, Clarkson and Rorabaugh 1989, Jennings and Hayes 1994b). In addition, in 1976 Hurricane Kathleen apparently modified the surface drainage patterns around San Sebastian Marsh, Imperial County, eliminating the wetland habitat that supported the species previously (E. Ervin, pers. comm.). *Rana yavapaiensis* also appears to be declining through parts of its range outside of California (Clarkson and Rorabaugh 1989, Stebbins 2003).

Nature and Degree of Threat

The declines in *Rana yavapaiensis* occurred before extensive collections were made or studies were carried out. As a consequence, threats to this species in California are poorly understood, with few actual data supporting any of the potential threats considered here. Possible threats that contributed to its decline include direct impacts from agricultural runoff, which has been shown to be highly detrimental to other species in the leopard frog complex (Relyea 2008), habitat alteration, including water availability and/or flow regimes (Hayes and Jennings 1986), and predation by or competition with introduced bullfrogs, predaceous fishes, and invertebrates (Clarkson and Rorabaugh 1989). Some recent declines in the closely related *R. onca* appear to be linked to encroachment of dense emergent vegetation into open water habitats (Bradford et al. 2004), and this process could plausibly also affect *R. yavapaiensis*. All of these factors were occurring simultaneously within the range of *R. yavapaiensis* along with declines, making it difficult to disentangle their effects (Hayes and Jennings 1986). In addition, over 13,000 km of ditches in the Imperial Valley were burned and subsequently sprayed with oil during this time, and this presumably adversely affected these frogs (Twining and Hensley 1943).

Chytridiomycosis has been documented as contributing to declines in *R. yavapaiensis* populations in Arizona (Bradley et al. 2002), and this disease is a concern for any remaining California populations. An additional concern is the possibility of competition or hybridization with *R. berlandieri* in California. *Rana berlandieri* was introduced into California well after *R. yavapaiensis* declined (Platz et al. 1990), so it is presumably not involved in the initial decline of the species. However, as it continues to expand its range in southern California, *R. berlandieri* may pose a risk to any remaining *R. yavapaiensis* populations (Rorabaugh et al. 2002). Hybridization has been documented between other species pairs of the leopard frog complex, including rare natural hybridization between *R. yavapaiensis* and the Chiricahua leopard frog (*R. chiricahuensis*) (Platz and Frost 1984). Molecular phylogenic analyses suggest that *R. berlandieri* is more closely related to *R. yavapaiensis* than to *R. chiricahuensis*, implying that natural hybridization between *R. berlandieri* and *R. yavapaiensis* may be possible. Because *R. berlandieri* is now far more common in California than *R. yavapaiensis*, ongoing hybridization, should it occur, may result in genetic swamping of any remaining populations.

Status Determination

Rana yavapaiensis has undergone severe declines and has not been documented in California in over 40 years, and there is a strong possibility that the species is already extirpated statewide. However, it remains possible that the frog is present in scattered isolated localities that have not been surveyed, or that frogs have gone undetected despite surveys.

If any populations persist, it is likely that they are vulnerable to the causes of initial decline throughout most of the California range of this species. Such populations, which are almost certainly small and/or isolated, would also be vulnerable to the natural fluctuations in population size that occur in this species. This natural vulnerability could be exacerbated by changing precipitation regimes in the southeastern part of California, where increasing temperatures, declines in precipitation, and greater year-to-year variation in rainfall are expected to occur due to climate change (Cayan et al. 2008b, PRBO 2011).

Management Recommendations

If new surveys locate remaining populations of this species, the habitat supporting these frogs should be protected while further study is carried out. Without a better understanding of this species' life history in California, establishing an effective management program will be difficult. If native California populations are not found, *Rana yavapaiensis* is a potential candidate for assisted reintroductions from nearby populations in Arizona, particularly in areas where introduced *R. berlandieri* are not present or have been eliminated. More generally, such future introductions should be attempted in habitats that are as pristine as possible, and are free of introduced anurans of any species, introduced predatory fishes (including mosquitofish), and pathogenic fungi.

Monitoring, Research, and Survey Needs

Survey efforts need to be renewed along the San Felipe Creek drainage, the Imperial Valley, and the Lower Colorado River. Although the most likely areas for remnant populations are those that have been the least impacted by agriculture and development, even degraded agricultural habitat can he utilized by *Rana yavapaiensis*, and therefore should be surveyed. Because populations are prone to large yearly fluctuations, surveys shonld he repeated over multiple years in both the wet and dry seasons. Surveys for larvae should also be undertaken since tadpoles are often more reliably detected than adults. If any remaining populations are located, the habitat surrounding these areas should be protected, and researchers should begin a monitoring program to quantify and track population sizes. Any suspected *R. yavapaiensis* populations should be confirmed using a set of molecular markers, both to firmly establish species identity and to check for hybridization between *R. yavapaiensis* and *R. berlandieri*. Because hybridization is a concern, both mitochondrial and nuclear markers should be used. Given the difficulty in distinguishing the two species, we recommend that populations of presumptive *R. berlandieri* be sampled for genetic material using nonlethal means (such as toe clips) and checked for diagnostic molecular markers to confirm that no native *R. yavapaiensis* DNA is present.

Should surveys discover extant populations of *R. yavapaiensis*, research into the basic life history and the causes of decline in California will be a prerequisite to developing an effective management program. Life history studies with a particular focus on habitat suitability should be undertaken on any populations that are located or reestablished.

COUCH'S SPADEFOOT

Scaphiopus couchii Baird 1854

Status Summary

Scaphiopus couchii is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 56% (62/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Scaphiopus couchii is a medium-sized (5.7–9.1 cm SVL) anuran with a black keratinized spade on the heel of each hind foot and a vertically elliptical pupil (Stebbins 2003). The dorsal coloration is variable, ranging from green or greenish-yellow to brownish-yellow with a pattern of darker markings forming lines, spots, or a reticulating network (Grismer 2002, Stebbins 2003). Males are generally greener and have less conspicuous dorsal patterning than females (Grismer 2002, Stebbins 2003). The ventral surface is whitish (Grismer 2002). The call is a short (~1 s) low groan that declines in pitch and has been described as sounding similar to the bleating of a sheep (Elliott et al. 2009). Within its range, *S. couchii* can be distinguished from all other frogs by the presence of a conspicuous black spade on the hind feet and a vertically oriented pupil. Specimens that

Couch's Spadefoot: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	0
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	0
v. Ecological tolerance (10)	10
vi. Population trend (25)	15
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	62
Total Possible	110
Total Score/Total Possible	0.56

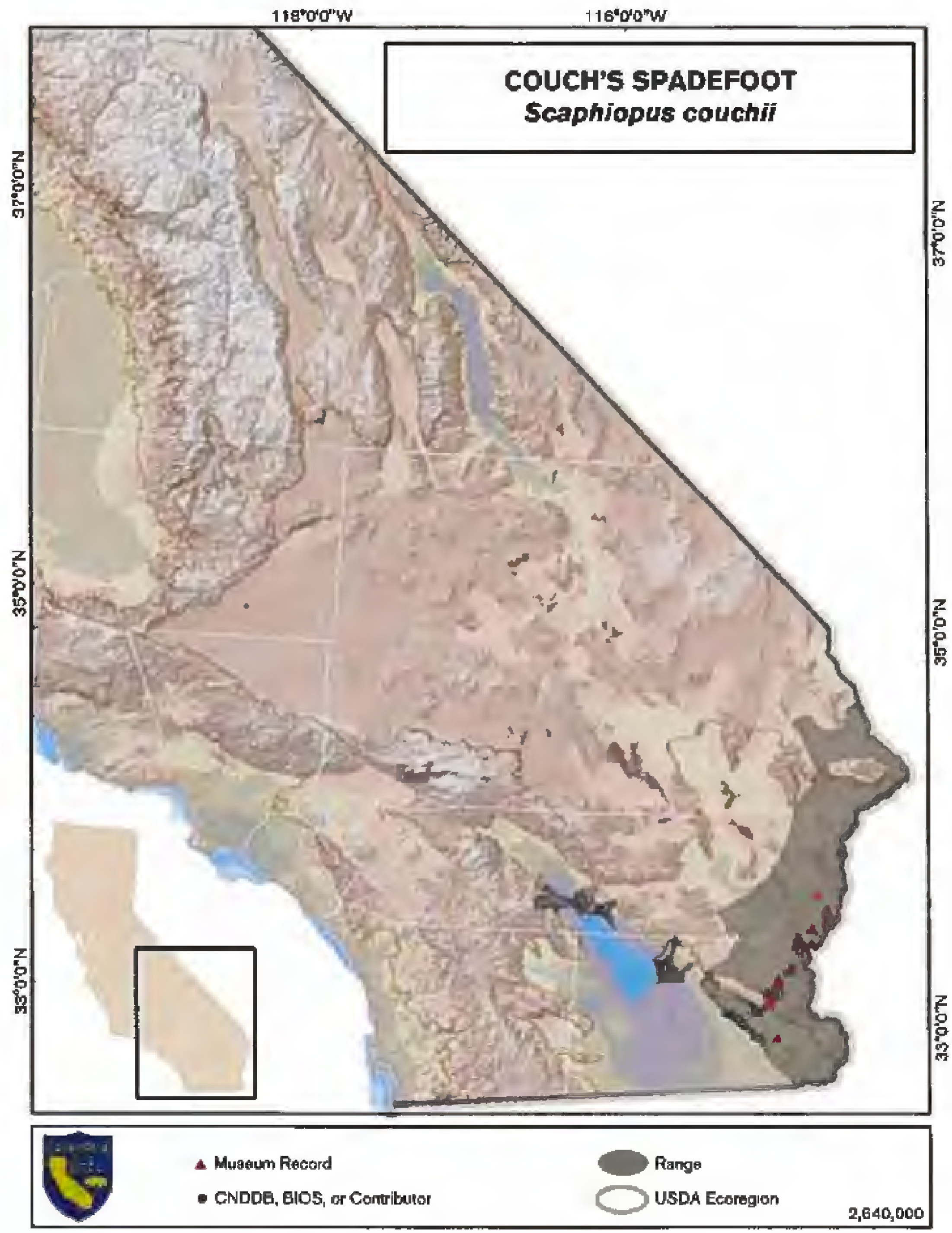


PHOTO ON PREVIOUS PAGE: Couch's spadefoot, Cochise County, Arizona. Courtesy of Rob Schell Photography.

have been unearthed from burrows, or have newly emerged, may be covered in a dark hardened layer of skin that soon sloughs off (Mayhew 1965).

Taxonomic Relationships

Scaphiopus couchii was initially described based on morphology, and its distinctiveness has not been questioned since this time (Baird 1854). Intraspecific variation has only been examined in a small portion of the range in Arizona and New Mexico (Chan and Zamudio 2009). Little genetic structure was observed among the populations studied.

Life History

Scaphiopus couchii is xeric-adapted and primarily fossorial, spending the majority of its life in self-constructed burrows and emerging only during and immediately after intense summer rains (Mayhew 1965, McClanahan 1967, Stebbins 2003). This species has been found to be active on the surface after summer monsoon rainstorms in August and September, but not after winter rainstorms in October, December, or January (Mayhew 1965), suggesting that the activity period is limited to the summer in California as it is throughout the rest of its range. However, California differs from the rest of the range in that it receives the majority of its rainfall during the winter, suggesting that little local adaptation has occurred in this species (Mayhew 1965). Outside of California, surface activity is also tied to rain events, although *S. couchii* has (rarely) been found active on the surface during periods of high humidity, even when no recent rainfall has occurred (Mayhew 1962).

Emergence behavior is elicited by the low-frequency sound of rain falling on the desert soil, not from the rain itself (Dimmitt and Ruibal 1980a). Frogs emerge from deep (20–90 cm) burrows on the first night following the first heavy summer rain (Shoemaker et al. 1969, Dimmitt and Ruibal 1980a). Most breeding for a season usually occurs on this first night of activity (Woodward 1982). Following this, the species may forage intermittently for up to 2 months, although much of this time is also spent in shallow (2–10 cm) burrows, which the frogs dig to avoid desiccation (Dimmitt and Ruibal 1980a). One feeding event can likely provide enough energy to allow an individual to persist for at least one year (McClanahan 1967, Dimmitt and Ruibal 1980b). Females deposit their eggs in ephemeral pools that form following intense summer rains (Woodward 1982). The development rate of this species is remarkably fast, with eggs hatching in as little as one day and metamorphosis occurring within 8–10 days if sufficient food is available (Mayhew 1965, Newman 1989, Morey and Janes 1994). Tadpoles are tolerant of a wide range of water temperatures (up to 39–42.5°C) such as are frequently encountered within the breeding pools (Brown 1969). This species likely does not breed every year in California and may skip reproduction and remain underground in suboptimal years (Mayhew 1962).

Habitat Requirements

Scaphiopus couchii requires soils that are soft enough to allow burrowing. The species appears to prefer areas that contain at least some vegetation, although burrowing in completely open areas is also known (Mayhew 1965, McClanahan 1967). This taxon also requires the presence of temporary desert rain pools that retain water for at least 8 days to allow sufficient time for metamorphosis. The area in California in which *S. couchii* occurs receives an average of about 6.5 cm of rainfall per year, and its fine-scaled distribution may be linked to the amount of runoff that collects in localized areas (Mayhew 1965). The distances traveled between upland retreats and breeding sites are not known, nor are the precise terrestrial habitat requirements of adults or juveniles.

Distribution (Past and Present)

Scaphiopus couchii ranges throughout much of Texas, Mexico, southern Arizona, and southern New Mexico, from near sea level to 1800 m (Stebbins 2003). Scattered, localized populations also occur in central Arizona and southern Colorado (Stebbins 2003). In California, this species ranges from the Colorado River west at least to the vicinity of the Algodones Dunes, ranging as far north as Chemehuevi Wash (~9 km north of Vidal Junction) and south to the vicinity of the United States–Mexico Border (Mayhew 1962, Tinkham 1962, Jennings and Hayes 1994a). A few observational records exist in the vicinity of the Salton Sea, and although these appear to be credible, verification is needed that populations are extant in this area.

This taxon's range in California is likely relictual from more mesic periods and is probably more fragmented now than it once was (Mayhew 1965). This species was not known to occur in the state until 1962 (Mayhew 1962, Tinkham 1962), and no significant declines have been documented since that time.

Trends in Abundance

No historical or current abundance data are available for this taxon within California. Human activities have both created and destroyed breeding sites for the species (S. Morey, pers. comm.), but no quantitative studies have documented the overall impacts of these activities on the species across California.

Nature and Degree of Threat

Scaphiopus couchii is likely persisting closer to its physiological limits in California than it is elsewhere within its range (Mayhew 1965). The California range is both hotter and drier than most of the rest of the range, and most of the limited rainfall occurs outside of the monsoon, during a time when *S. couchii* is usually inactive. The current populations in California likely persist due to the presence of local conditions that allow for the collection of sufficient quantities of water, such as the presence of basins on the eastern base of the Algodones Dunes and pools that form along desert washes. The relatively fragmented nature of the species' California distribution and the physiological conditions under which it lives make it susceptible to localized extirpations due to habitat modification that destroys temporary pools and due to the effects of climate change. Recent models (PRBO 2011) indicate that average temperature will increase significantly, by more than 2°C in most months in the Sonoran/Colorado Desert of California. Given that *S. couchii* may already be near its physiological temperature limits, this may have an enormous impact on its viability in the state. In addition, some precipitation projections include an overall decrease of up to 45% (PRBO 2011), and increased variation in year-to-year precipitation (Cayan et al. 2008b), which could have severe detrimental impacts on this species by decreasing the number of years in which enough rainwater collects to allow breeding. Essentially, if the interpretation is correct that the California population exists at the physiological limits of the species' capacity, then predicted changes in rainfall and temperature may seriously reduce its range in the state.

Off-highway vehicle usage in the Algodones Dunes has degraded habitat in many areas (R. Fisher, pers. comm.). Noise generated by off-highway vehicle usage has been implicated in eliciting emergence in this species by mimicking the sound of falling rain that it uses as an emergence cue (Brattstrom and Bondello 1979). Temporary and permanent anthropogenic water sources associated with livestock (cattle ponds) and perhaps agriculture may help to provide suitable breeding habitat that is important to the persistence of this species.

Status Determination

The small and fragmented range of this taxon, coupled with its sensitivity to habitat disturbance through off-highway vehicle use and predicted climate change, justifies its Priority 3 status.

Management Recommendations

The primary, immediate management goal for *Scaphiopus couchii* is to protect existing habitat from further impact. Off-highway vehicle use and larger modifications (solar projects, mining) may negatively alter both the hydrology of breeding pools and the suitability of soil for burrowing. In particular, if pools are modified such that they dry faster (through either more rapid draining or overall smaller size), their hydroperiod may become too short to allow metamorphosis. Specific areas requiring protection should be determined by the surveys outlined below. In the future, the impacts of projected climate change may seriously threaten this species in California, and proactive management may be required to counteract this threat; such management could include relocating populations to cooler or more mesic sites, deepening and maintaining the hydroperiod of natural breeding sites, and potentially creating completely novel breeding pools that can hold water if the climate changes.

Monitoring, Research, and Survey Needs

Range-wide surveys need to be undertaken for this taxon to identify suitable remaining habitat, determine the sizes of extant breeding populations, and to further characterize the species' range in California. To our knowledge, the northernmost population at Chemehuevi Wash has not been resurveyed since its original description in 1962 (R. Fisher, pers. comm.), and this is an important area in need of surveys. As the species distribution in California is patchy, largely in remote regions of the state, and given that the species does not emerge every year, care should be taken to search desert pool habitats even in areas where this anuran has not yet been documented. Surveys should ideally take place during the first night following the first major summer (monsoonal) rain event. Surveyors should be experienced with this frog's call (Elliott et al. 2009), as this will likely be the easiest way to find populations, and pools should be surveyed for tadpoles within a few days after they fill during summer rains.

The movement ecology of this taxon and its potential to recolonize previously extirpated areas are unknown and are a topic in need of further study, particularly so in California where populations appear to be fragmented. Additional study of its physiological limits would also be helpful in establishing a more informed management plan, now and in the face of future climate changes. In particular, the severity of drought and the number of years between breeding events that can be tolerated are critical pieces of information for the long-term management of this species. Landscape ecological information, including the amount of terrestrial habitat needed, the relationship between population size and pool basin size, inundation duration and frequency, and the movement frequency of animals between breeding sites would all be valuable for future management considerations. Additional information on habitat use itself, including the extent that ongoing railroad and water diversion projects within the range subsidize or detract from potential habitat for this species, is also a critical research need that would inform ongoing management of this species.

Finally, given the spotty distribution of the species and the potential for genetic isolation among sites, multi-locus population genetic studies using microsatellites or single nucleotide polymorphisms of all extant California populations would provide a badly needed estimate of the extent to which populations are subdivided and therefore the optimal management strategies to protect genetic diversity. In addition, given how widespread the species is across the southwestern United States, genetic data comparing the uniqueness of the California population is essential for range-wide management.

WESTERN SPADEFOOT

Spea hammondii (Baird 1859)

Status Summary

Spea hammondii is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 69% (76/110). During the previous evaluation, it was also considered a Species of Special Concern under the name *Scaphiopus hammondii* (Jennings and Hayes 1994a).

Identification

Spadefoot toads as a group have catlike eyes with vertical pupils, a single black spade on each hind foot, and indistinct paratoid glands (Stebbins 2003). *Spea hammondii* is dusky green or gray dorsally, often with irregular markings (Stebbins 2003). Tubercles on the skin are tipped with orange or red, and the irises are usually pale gold (Jennings and Hayes 1994a, Stebbins 2003). The ventral surface is white to light gray without markings (Stebbins 2003). Adults are 4–6 cm SVL (Stebbins 2003). Larvae can reach approximately 7 cm in TL and their eyes are set close together when viewed from above (Stebbins 2003). This species is unlikely to be confused with other sympatric anurans.

Taxonomic Relationships

North American spadefoots have had a confusing taxonomic history. Studies using allozymes and morphology (Wiens and Titus 1991) and mitochondrial DNA (Garcia-Paris et al. 2003) support the species status of *Spea hammondii*,

Western Spadefoot: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	7
v. Ecological tolerance (10)	10
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	76
Total Possible	110
Total Score/Total Possible	0.69

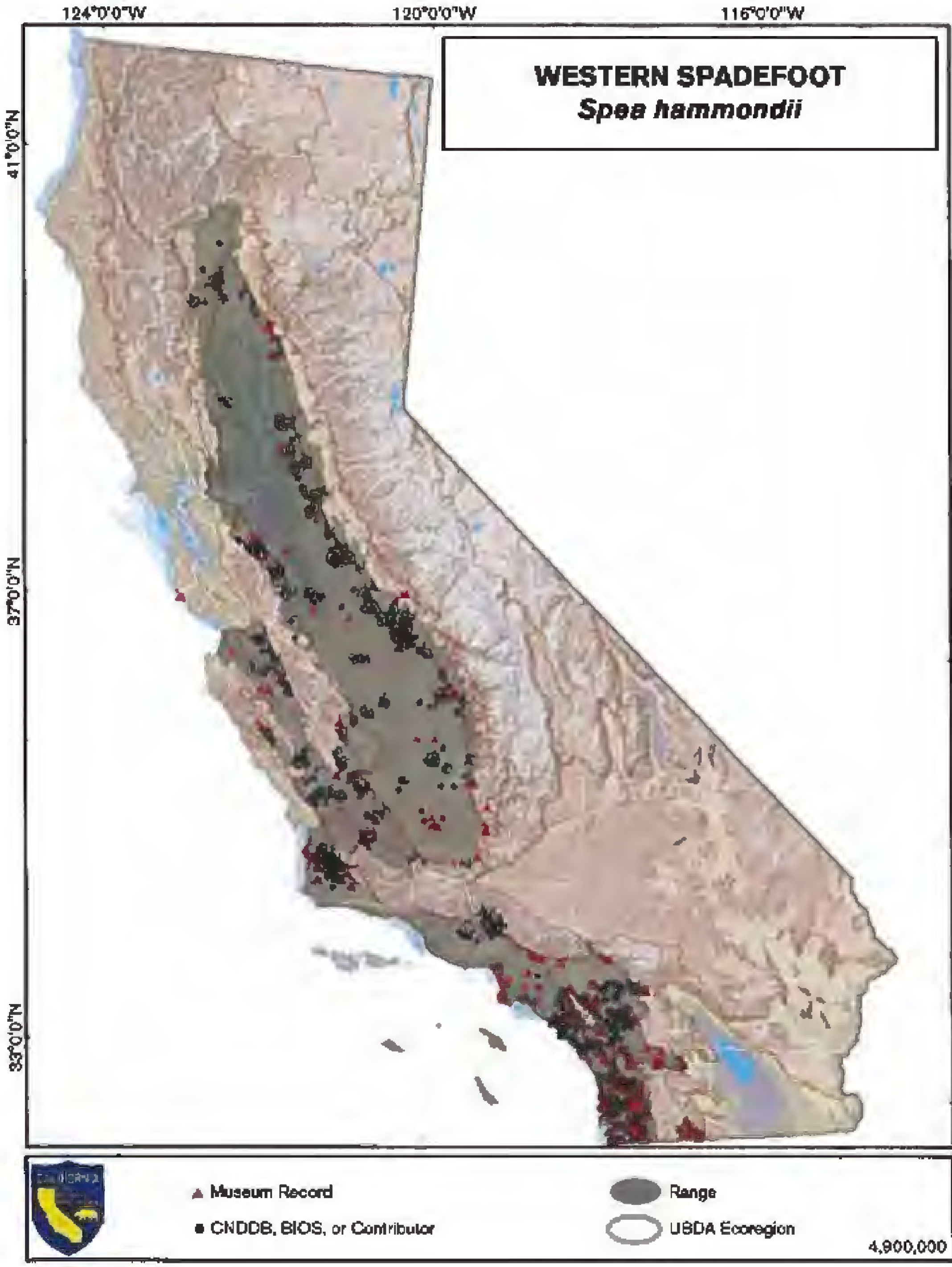


PHOTO ON PREVIOUS PAGE: Western spadefoot, Sacramento County, California. Courtesy of Robert Thomson.

placing it sister to a clade consisting of the Great Basin spadefoot (*S. intermontana*) and the Plains spadefoot (*S. bombifrons*). This arrangement is consistent with Kluge (1966) and Sattler (1980). Relationships within *Spea* are still unresolved however, and cryptic taxa may exist within *S. hammondii* (García-París et al. 2003) and *S. intermontana* (Wiens and Titus 1991). Ongoing phylogeographic work should clarify the extent of intraspecific variation in the species. Preliminary data indicate that some mitochondrial introgression has occurred between *S. intermontana* and *S. hammondii* in southern California, but not the Central Valley portions of the species' range (P. Spinks, unpublished data).

Life History

Adult *Spea hammondii* are terrestrial, moving from summer refugia to ephemeral water bodies to breed in the spring following warm late winter or spring rains (Storer 1925, Burgess 1950, Stebbins 1954, Feaver 1971, Brown 1976, Morey 1998). Breeding aggregations can consist of more than 1000 individuals (Jennings and Hayes 1994a). Breeding occurs over a 2–3 week period, during which males can be heard chorusing intermittently (Brown 1976, Morey and Reznick 2004). Additional bouts of breeding can occur, and pools can contain cohorts of different ages (Morey 2005). Onset of breeding activity varies depending on rainfall and region. For example, heavy rains in 1991 resulted in breeding occurring only in March (San Luis Obispo and Riverside Counties; Morey and Reznick 2004). In the two following years, breeding occurred between January and March (Morey and Reznick 2004). Breeding has also been documented in August, and from October to December in San Diego County (Ervin et al. 2005, Ervin and Cass 2007). It is unknown how common early-breeding behavior is, but the October 2004 events may have been in response to very dry conditions, and many of these larvae ultimately succumbed to desiccation. The previous year, 2003, set a record rainless period, and the breeding in 2004 occurred after the first measurable rain in 181 days (Ervin et al. 2005).

Females lay 300–500 eggs in clusters of 18–25 (Stebbins 1951, Stebbins 1985) that usually hatch in 3–4 days (Morey 2005). Morey and Reznick (2004) surveyed vernal pools in San Luis Obispo and Riverside Counties and found that the larval period lasted an average of 58 days. In the laboratory, the minimum time for larval development was estimated to be 14 days (Morey and Reznick 2004). Males raised experimentally under high food conditions developed secondary sexual characters by the beginning of their first breeding season after metamorphosis, while females of the same age had adult coloration but underdeveloped ovaries (Morey and Reznick 2001). It is unknown how long it takes to reach maturity in the field, but based on this experimental work males probably mature 1–2 years after metamorphosis and females at least 2 years after metamorphosis. Most individuals are mature at 4–4.5 cm SVL (Storer 1925, Morey and Guinn 1992).

Larvae are frequently at risk of desiccation due to pools drying before development is complete. In Fresno County, 17 out of 23 vernal pools dried before larvae metamorphosed (Feaver 1971). Across 20 populations in San Luis Obispo and Riverside Counties, Morey and Reznick (2004) observed that 15% of ponds dried before larvae metamorphosed. As pools dry, larvae experience increased daily variation in temperature, increased ammonia levels, increased water hardness, and decreased depth (Morey and Reznick 2004). These factors lead to crowding and decreased growth rate. While several cues are operating simultaneously, water reduction alone is sufficient to trigger accelerated development within 24 hours (Denver 1997a, Denver 1997b, Denver et al. 1998, Boorse and Denver 2003). In the field, there is a positive correlation between hydroperiod and mass at metamorphosis (Morey and Reznick 2004). In the lab, animals reared at low density were larger (4.96 g) at metamorphosis and metamorphosed sooner (77.8 days) than animals maintained at high density (2.9 g, 87.8 days; Morey and Reznick

2001). Survivorship of metamorphs was also higher for animals that were larger at metamorphosis, regardless of larval density (Morey and Reznick 2001). Effects of the larval rearing environment persisted for several months after metamorphosis, but small metamorphs were able to catch up in growth if terrestrial food availability was high.

Little is known about terrestrial activity, although most movement and surface activity is thought to be nocturnal (Morey 2005). Juveniles leave natal pools shortly after metamorphosis in April–June presumably seeking refugia, although their terrestrial habitat is unknown (Morey 2005). Adults and juveniles retreat to burrows by late summer, with juveniles capable of digging burrows 10–20 cm deep even in hard, dry soil (Morey and Reznick 2001). Mammal burrows may also be used (Stebbins 1951).

Larval diet has not been studied, although larvae of other spadefoot species are generalists, consuming animals, plants, and organic detritus (Pomeroy 1981, Pfennig 1990). Cannibal morph larvae with broad heads and enlarged jaw muscles are known from San Luis Obispo and Riverside counties, but it is unknown how common they are throughout the species' range (Morey 2005). Adults are generalized predators on terrestrial arthropods and other prey, including beetles, moths, flies, and earthworms (Morey and Guinn 1992).

Habitat Requirements

Spea hammondii occurs in grasslands, oak woodlands, coastal sage scrub, and chaparral vegetation in washes, floodplains, alluvial fans, playas, and alkali flats (Stebbins 2003, Morey 2005). Temporary pools are used for breeding, but *S. hammondii* will also readily breed in artificial water bodies such as cattle ponds (Morey 2005). Vernal pools used by *S. hammondii* for breeding had an average ponding duration of 81 days (range 36–127, $n = 9$, San Luis Obispo and Riverside Counties) (Morey and Reznick 2004). Pools with at least some successful recruitment lasted on average 3 weeks longer than larval development time (Morey and Reznick 2004). Pool temperature during larval development ranged from 11°C to 32°C (Morey and Reznick 2004). Brown (1967) found that water temperatures between 9°C and 30°C were necessary for larval development (eggs collected from Riverside County).

Perennial pools containing introduced predators such as crayfish, fish, or bullfrogs are often unsuitable for successful recruitment (Jennings and Hayes 1994a). However, in southern California, ephemeral pools utilized by introduced species with predatory aquatic stages, such as the African clawed frog (*Xenopus laevis*), can still function as breeding habitat for *S. hammondii* (confirmed by the presence of dispersing metamorphs), but the effects these introduced species have on overall recruitment levels are unknown (Ervin and Fisher 2001, Ervin and Burkhardt 2006).

Distribution (Past and Present)

Spea hammondii occurs in the Central Valley and bordering foothills across southern California from Shasta County south into northwestern Baja California, including the Coast Ranges south of Monterey, from sea level to 1365 m (Jennings and Hayes 1994a, Ervin et al. 2001, Stebbins 2003; S. Barry, pers. comm.). Jennings and Hayes (1994a) concluded that as of the 1990s, over 80% of historically occupied habitat in southern California and 30% of habitat in northern California were no longer suitable due to development and habitat conversion. In surveys throughout the Central Valley, Fisher and Shaffer (1996) reported *S. hammondii* as virtually extirpated from the Sacramento Valley and at a reduced density in populations of the eastern San Joaquin Valley.

Trends in Abundance

Current or historical abundance data are largely unavailable or anecdotal, and little recent data is available. Recent surveys of Mather Airport (formerly Mather Air Force Base) in Sacramento County estimated that breeding adults numbered in the few dozens, although this was based on short-duration surveys and limited data (A. Chang, unpublished data). Morey and Guinn (1992) reported an average of 1.16 individuals/km of roadway during a relatively wet winter (1982–1983) and 0.68 individuals/km during a drier winter (1984–1985) in the San Joaquin Valley.

Nature and Degree of Threat

The major threat to *Spea hammondii* is habitat loss and fragmentation due to agriculture and urban development. Other threats include invasive species and climate change. Davidson et al. (2002) found that currently occupied sites had less surrounding urban development than extirpated sites. Extant populations also occur at higher elevations than extirpated sites on average, possibly due to invasive species being more common at lower elevation (Fisher and Shaffer 1996, Davidson et al. 2002). *Spea hammondii* is sensitive to invasive species such as crayfish, bullfrogs, and mosquitofish; however, many of these species cannot persist in the highly ephemeral breeding habitats *S. hammondii* uses (Jennings and Hayes 1994a, Morey 2005). Bullfrogs have been documented to prey on *S. hammondii* (Morey and Guinn 1992, Balfour and Ranlet 2006), although the impact of this predation on overall abundance is unknown.

Spea hammondii may be at risk from climate change because breeding is dependent upon temperature and rainfall cues, and larval development requires ephemeral pools to persist long enough to complete development (Morey and Guinn 1992, Jennings and Hayes 1994a). Mean annual temperatures are projected to increase throughout the range of *S. hammondii*, with warmer winters and summers and earlier spring warming expected (reviewed in PRBO 2011). The frequency of extremely hot days is predicted to increase by up to 25 days per year in some parts of the range (Bell et al. 2004). There is less certainty about future precipitation patterns, with estimates ranging from little change to roughly 30% decreases in rainfall (Snyder and Sloan 2005, PRBO 2011). Changes in temperature and precipitation will likely affect vernal pool hydrology (e.g., Pyke 2005) and may also affect the timing of breeding, though how *S. hammondii* will respond to these changes needs further study. The largely unsuccessful early breeding observed by Ervin et al. (2005) may be indicative of the kinds of mismatches in environmental cues and breeding behavior that this species may suffer under climate change. The probability of large (>200 ha) wildfires is expected to change very little in the Central Valley (Westerling and Bryant 2008). In the more northern coastal part of the range, the probability of large fires is expected to increase (Westerling and Bryant 2008), and the area burned is expected to increase by up to 50% (Lenihan et al. 2008). In the southern part of the range where wildfire is common, there is little consensus on future fire dynamics because of the difficulty in modeling Santa Ana weather events (Westerling et al. 2004, Westerling and Bryant 2008). The largely subterranean lifestyle of *S. hammondii* may make it relatively resistant to the effects of fire. However, wildfires occurring during dispersal may be particularly detrimental due to direct mortality and habitat degradation and this issue requires more study. Vegetation shifts due to climate change are expected to be modest in the Central Valley, where land use is a more important determinant of habitat type (PRBO 2011). Elsewhere in the range, chaparral and shrublands are expected to decrease in area, while grassland is expected to increase (Lenihan et al. 2008, PRBO 2011). The impact of these shifts may be modest as *S. hammondii* uses all of these habitat types.

Status Determination

Ongoing habitat loss and extirpations throughout the range of *Spea hammondii* warrant Priority 1 Species of Special Concern status.

Management Recommendations

Remaining sites should be protected from urban and agricultural development, with emphasis on larger habitat blocks that allow for more natural metapopulation dynamics to persist. The fact that *Spea hammondii* readily breeds in anthropogenic structures can be exploited to create breeding habitat in response to habitat loss and potentially also to climate changes that affect natural vernal pool phenology. Terrestrial habitat is likely not so easily restored, and minimizing or eliminating disturbance around breeding habitat would help protect adults (see the "Monitoring, Research, and Survey Needs" section). Efforts to remove introduced predators from breeding habitat should be considered. In some cases, cattle grazing operations may be beneficial to *S. hammondii*. Over 3 years in Sacramento County, Marty (2005) found that experimentally grazed vernal pools experienced fewer drying and refilling cycles within a season, and had a longer maximum inundation period (115 days) than ungrazed treatments (65 days) or treatments where grazing occurred seasonally (65–78 days).

Monitoring, Research, and Survey Needs

Research is needed into terrestrial habitat use (Jennings and Hayes 1994a, Morey 2005), including juvenile dispersal, adult migration patterns and distances, and the importance (if any) of rodent burrows for all age classes. This information is important for determining how much and what kinds of terrestrial habitat to protect around breeding sites. For example, Morey and Reznick (2001) found that the quality of juvenile terrestrial habitat in terms of food availability compensated for stressful larval conditions. Additional study on which environmental conditions promote post-metamorphic survival will aid in management planning. It is also unknown what proportion of adults breed each year and how long individual adults spend at breeding sites (Morey 2005). Underground habitat use is poorly known, including feeding and dormancy patterns. Remaining populations are likely highly fragmented, and research is needed into connectivity among populations at both the local and the regional levels; additional landscape ecology and genetic studies would help determine patterns of differentiation (Jennings and Hayes 1994a). Finally, comparative studies of this species in the Central Valley and southern California would help determine the extent of biological variation in life history patterns across this ecological gradient.

SOUTHERN LONG-TOED SALAMANDER

Ambystoma macrodactylum sigillatum **Ferguson 1961**

Status Summary

Ambystoma macrodactylum sigillatum is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 66% (73/110). It was not considered a Species of Special Concern during the previous evaluation (Jennings and Hayes 1994a).

Identification

Ambystoma macrodactylum sigillatum is a medium-sized (4.1–8.9 cm SVL) salamander with a broad head and large eyes (Stebbins 2003). The dorsal ground coloration is black or dusky brown with a yellow dorsal stripe that is usually divided into blotches on the body and into fine spotting on the head and tail (Ferguson 1961, Petranka 1998, Stebbins 2003). Small whitish-blue flecks are present on the sides of the body, and the ventral surface is dark brown (Stebbins 2003). The larvae have large bushy gills and a dorsal fin that extends to near the forelimbs (Petranka 1998).

Metamorphosed individuals of this species are unlikely to be confused with any other salamanders within its range. Other subspecies of *A. macrodactylum* have similar body proportions but differ in the size, extent of blotching,

Southern Long-Toed Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	15
iii. Population concentration/migration (10)	10
iv. Endemism (10)	3
v. Ecological tolerance (10)	3
vi. Population trend (25)	20
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	73
Total Possible	110
Total Score/Total Possible	0.66

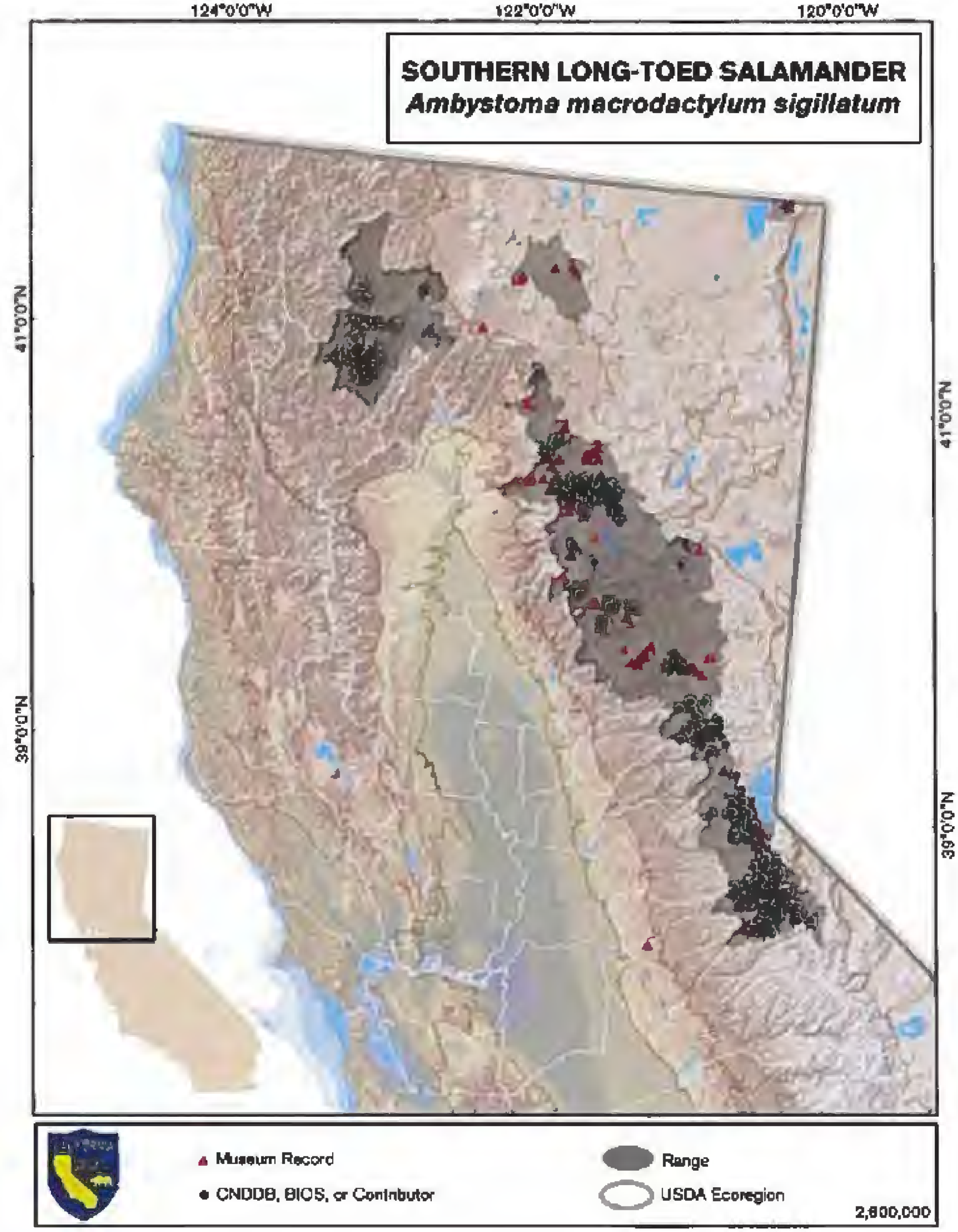


PHOTO ON PREVIOUS PAGE: Southern long-toed salamander, Butte County, California. Courtesy of Robert Hansen.

and coloration of the dorsal stripe, and their ranges do not overlap in California. Differentiating larvae from co-occurring newts (*Taricha granulosa*, *T. torosa*) requires careful attention. Newt larvae generally have small, narrow heads and few gill rakers (5–7 on the anterior side of the third gill arch), whereas *A. macrodactylum* larvae have broad heads and 9–13 gill rakers on the anterior side of the third arch (Stebbins 2003).

Taxonomic Relationships

Ambystoma macrodactylum sigillatum is one of five currently recognized subspecies of long-toed salamander (Petranka 1998, Stebbins 2003). *Ambystoma macrodactylum* has been widely recognized as a distinct species since its initial description by Baird (1854). Since this time, a number of different species and subspecies have been described. The current five-subspecies arrangement stabilized after the work of Ferguson (1961), which described *A. m. columbianum* (eastern long-toed salamander) and *A. m. sigillatum*, as well as the work of Russell and Anderson (1956), which described the geographically isolated *A. m. croceum* (Santa Cruz long-toed salamander) from Santa Cruz and Monterey Counties. Ongoing genetic studies indicate that several of these subspecies may warrant full species status (Savage 2008). *Ambystoma macrodactylum sigillatum* was described based on the size, color, and pattern of the dorsal band, as well as vomerine tooth counts (Ferguson 1961). Although it intergrades morphologically with *A. m. columbianum* at the northern edge of its range (Ferguson 1961), ongoing genetic analyses support recognition of *A. m. sigillatum* as a distinct species (Savage 2008).

Life History

Ambystoma macrodactylum sigillatum is a pond-breeding salamander that often has a prolonged larval stage. The life history of this taxon varies widely depending on elevation and climate (Petranka 1998). Here we have summarized data for *A. m. sigillatum*, where possible, and described the variation present across the species where the life history is highly variable and/or uncertain.

Adults emerge from hibernation and migrate to breeding habitat after the first thaw. Mating begins shortly after adults enter the breeding habitat, usually in May or June, with lower-elevation populations usually being able to breed earlier than higher-elevation populations (Anderson 1967, Howard and Wallace 1985). Elsewhere in the *A. macrodactylum* range, primarily at low elevations where the climate is mild, breeding is not delayed by winter freezes, so reproduction starts with the onset of fall rains (Ferguson 1961, Nussbaum et al. 1983). As in other *Ambystoma* species, mating follows a pattern of courtship and spermatophore deposition. Females oviposit on vegetation, rocks, sticks, or directly on the pond bottom 2–3 days following courtship and mating (Anderson 1961, Stebbins 2003). The eggs are laid singly or in clumps of up to 100 eggs (Petranka 1998, Stebbins 2003). The pattern of egg deposition varies geographically in this species: *A. m. sigillatum* tends to lay eggs singly or in long loose clusters in relatively deep water (Anderson 1967), although this is variable. Eggs hatch in 2–5 weeks, with longer incubation periods required at higher elevations and lower water temperatures (Anderson 1967, Nussbaum et al. 1983, Petranka 1998). The larval period can be as short as 50 days in temporary pools at lower elevations but may last 2 years in the highest elevations in permanent pools (Nussbaum et al. 1983, Pilliod and Fronzuto 2005). Size at metamorphosis varies widely from 2.3 to 4.8 cm SVL (Howard and Wallace 1985). This species is able to tolerate a relatively wide range of water temperatures, with larvae overwintering under the ice at near freezing temperatures but then selecting the warmest areas available throughout the summer (up to 24.5°C). Presumably these temperatures allow for more rapid larval growth and development (Anderson 1968b).

Ambystoma macrodactylum sigillatum is a generalist predator, as both larva and postmetamorph, that feeds on a variety of small insects, crustaceans, and spiders (Anderson 1968a). Larvae and males in the aquatic environment will prey on zooplankton, insect larvae, and small snails (Anderson 1968a, Nussbaum et al. 1983). In the lab, larvae are also known to take frog (primarily *Pseudacris*) tadpoles and conspecific larva (Anderson 1968a, Nussbaum et al. 1983). Females apparently do not feed in the aquatic environment, which may simply reflect the short amount of time they spend there during the breeding season (Anderson 1968a).

Habitat Requirements

Ambystoma macrodactylum, as a species, occurs in a larger variety of habitat types than any other salamander in the Northwestern United States (Ferguson 1961, Nussbaum et al. 1983). Suitable habitats for *A. m. sigillatum* include arid grassland and sagebrush communities, dry woodlands, coniferous forests, alpine meadows, and a wide variety of intermediate habitat types (Ferguson 1961, Petranka 1998, Pilliod and Fronzuto 2005). In some areas, this species is abundant in disturbed agricultural areas (Nussbaum et al. 1983). Elsewhere in the range, landscape genetic studies indicate that populations that persist in highly modified habitats do so with increased population isolation, probably increasing susceptibility to local extirpations (Goldberg and Waits 2010).

At high elevations (above 2450 m in the Sierra Nevada and 2100 m in the Klamath Mountains), where breeding occurs late and larval development is prolonged, some populations of *A. m. sigillatum* require permanent water bodies for breeding because larvae overwinter prior to metamorphosis (Anderson 1967; K. Leyse, pers. comm.). If these overwintering sites are shallow (1–2 m in depth), as is common in the Tahoe region of the Sierra Nevada, few larvae seem to survive the winter (K. Leyse, pers. comm., unpublished data). Spring-fed water bodies may increase the likelihood of successful overwintering, though more data are required to verify this. This subspecies also persists far more readily in fishless water bodies (see the "Nature and Degree of Threat" section).

The species is known to utilize hardwood forests, meadows, and granite slopes for upland habitat. Further study on the extent and types of upland habitat that this species requires are needed.

Distribution (Past and Present)

Ambystoma macrodactylum sigillatum ranges from southwestern Oregon (south of the Calapooya divide, Lane and Douglas Counties) through the Trinity Alps, Warner Mountains, Sierra Nevada, and adjacent areas of northwestern California reaching as far south as Carson Pass (Ferguson 1961, Brode 1967, Bury 1970a, Pilliod and Fronzuto 2005). The known elevational range for this taxon is from near sea level to 3000 m (Stebbins 1966, Nussbaum et al. 1983), although the distribution in California is restricted to the higher end of this range. The presence of isolated populations of the species *A. macrodactylum* in Santa Cruz and Monterey Counties, California (*A. m. croceum*), and in southeastern Oregon suggests that the species may have been historically distributed more broadly throughout the west. If so, the present day range likely reflects a range contraction as climate has changed over the last several thousand years.

Localized, present-day changes in distribution appear to be ongoing in several parts of California. In the historically fishless Klamath Siskiyou bioregion, *A. m. sigillatum* are 44 times more likely to be present in lakes without fish than lakes that contain fish. Because these fish have been introduced during the last 150 years, it is likely that some lakes where *A. m. sigillatum* does not occur represent localized extirpations as a result of fish predation (Welsh et al. 2006). A similar pattern occurs in the north central Sierra Nevada near Lake Tahoe. Here, *A. m. sigillatum* are present in 92.3% of fishless sites, but only 37.5% of fish-containing sites (Leyse 2005). In the Klamath Mountains, *A. m. sigillatum* was documented at 25 of 118 sites in surveys conducted between 1999 and 2001. Salamanders were present at only 15 of these sites when they were resurveyed in 2008 (K. Pope, pers. comm.). The overall geographic extent of the *A. m. sigillatum* range appears to still be intact, but it is clear that localized extirpations are occurring in several areas.

Trends in Abundance

Abundances of *Ambystoma macrodactylum sigillatum* have declined throughout relatively large areas of the California range. The Klamath Mountain surveys described above documented 4126 individuals at 25 occupied sites in 1999–2001 but only 569 individuals at the 15 occupied sites in 2008 (K. Pope, pers. comm.). Few historical abundance data are available, but overall current abundance of larvae at lower-elevation sites appears to be low (K. Leyse, pers. comm.). Population genetic estimates of population trends suggest that regional populations exchange few migrants and that effective population sizes are small (Savage et al. 2010).

Nature and Degree of Threat

Trout introductions are the largest threat to remaining populations of *Ambystoma macrodactylum sigillatum*. Welsh et al. (2006) found that the absence of introduced fish was a major predictor of *A. m. sigillatum* presence even after controlling for other environmental variables. Aside from the local effect of fish on individual water bodies, fish introductions appear to affect *A. macrodactylum* populations at the scale of entire watershed basin. In Idaho, basins with higher introduced fish densities had significantly lower densities of *A. macrodactylum* (Pilliod and Peterson 2001). The authors postulated that much of the remaining fishless habitat in fish-containing basins is too shallow for most larvae to successfully overwinter and that the deeper, fish-containing pools no longer acted as stable source populations for the basin. This led to a destabilization of normal source–sink dynamics, causing declines throughout the entire basin. These results suggest that the presence of fish at the basin scale is a significant conservation risk, irrespective of whether patches of fishless habitat remain within the basin (Pilliod and Peterson 2001). Where *A. m. sigillatum* persist in the presence of fish, larval densities are very low both in deeper fish-containing pools and in adjacent fishless pools (K. Leyse, pers. comm.). When larvae are found in fish-containing pools, they tend to hide under rocks or are only captured in overnight trapping, indicating that they may alter their behavior in response to the presence of predators (K. Leyse, pers. comm., though see Tyler et al. 1998). Declines due to the presence of fish have also been documented elsewhere in *A. macrodactylum*'s range (Liss and Larson 1991, Liss et al. 1995, Tyler et al. 1998). In Montana, introduced trout were linked to *A. m. krausei* extirpations. Salamander recolonization following local trout extirpations strongly indicated that trout were the actual causal agent of declines (Funk and Dunlap 1999).

Climate change also poses a threat for *A. m. sigillatum*. Many of the remaining pools that this species utilizes are shallow. Projected shifts to earlier and faster snowmelt in the Sierra Nevada could have complex and possibly negative effects on this species by changing the hydrology of lakes and ponds (Cayan et al. 2008b, Franco et al. 2011, PRBO 2011). As many of these pools appear to be spring fed, any changes to hydrology of the springs could also have severe impacts (Leyse 2005).

Disease and environmental contaminants may also pose threats for remaining populations of *A. m. sigillatum*. Lethal ranavirus infections of *A. m. sigillatum* were recently detected in Lassen Volcanic National Park (Bunck et al. 2009). This species is also susceptible to iridovirus infection and exposure to atrazine, a commonly used herbicide (Forson and Storfer 2006). *Bd* has been detected in a single adult salamander at Carter Meadow in Lassen National Forest, although the load was low. Prevalence of *Bd* appears to be low for this species and no evidence of die-offs or illness due to this pathogen is known (K. Pope and J. Piovia-Scott, unpublished data).

Status Determination

Ongoing serious declines in distribution and abundance are the primary reasons for this Priority 2 status.

Management Recommendations

The presence of relatively deep fishless pools appears to be important to the continued persistence of this species, particularly at the highest elevations. As such, fish stocking should be limited in areas where *Ambystoma macrodactylum sigillatum* occurs. Where stocking does occur, mitigation strategies outlined by Appendix K of California Department of Fish and Wildlife hatchery and stocking program environmental impact report should be followed (ICF Jones and Stokes 2010).

Monitoring, Research, and Survey Needs

Declines due to fish predation have now been amply demonstrated, so continued monitoring on the effects of fish predation is less important than work related to fish removal. If predaceous fish can be successfully removed from areas supporting this species, occasional monitoring should be undertaken to detect unauthorized reintroductions, particularly in areas that experience high human impact and to document recolonization dynamics by the salamanders. An important management question centers on the relative importance of permanent and temporary pools to metapopulation dynamics across elevations. That is, it may be that at lower elevations, temporary fish-free pools are the primary source of successful recruitment, and deeper lakes can therefore be maintained as fishing resources, whereas at the highest elevations, the species can only persist if permanent, fish-free habitats are common. The type and extent of upland habitat utilized by this species is also in need of further study. In particular, the extent of upland habitat that populations require in order to persist has not been studied in this taxon. Climate change could also have different impacts on the upland phase of the life cycle, in addition to the impacts that are projected for the aquatic part of the life cycle. In addition, populations are still under considerable risk from disease, and monitoring efforts focused on detecting the presence of ranavirus and *Bd* should be continued.

SANTA CRUZ BLACK SALAMANDER

Aneides flavipunctatus niger Myers and Maslin 1948

Status Summary

Aneides flavipunctatus niger is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 48% (53/110). This taxon was not previously considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Aneides flavipunctatus niger is a medium-sized plethodontid salamander (5.1–9.5 cm SVL) (Stebbins 2003). The adult dorsal coloration is either solid black or black with a few small white flecks (Myers and Maslin 1948). Juveniles (<4.0 cm SVL) have brassy dorsal pigmentation with white to blue-white spots (Lynch 1981). The ventral coloration is black or dark gray (Myers and Maslin 1948). The nasolabial grooves and costal grooves are well defined, and most individuals (95%) have 17 costal grooves (Lynch 1981, Stebbins 2003). *Aneides flavipunctatus niger* has rounded toe tips, counter to the squared toe tips typical of *Aneides*. Its limbs are short relative to the trunk, with 3–5 costal grooves between adpressed limbs. The heads of males are larger than those of females, and are roughly triangular with prominent, protruding upper jaw teeth (Stebbins 2003).

Santa Cruz Black Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	10
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	7
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	3
Total Score	53
Total Possible	110
Total Score/Total Possible	0.48

PHOTO ON PREVIOUS PAGE: Santa Cruz black salamander, Santa Cruz County, California. Courtesy of William Flaxington

Aneides flavipunctatus niger could be confused with the co-occurring arboreal salamander (*A. lugubris*). Adult *A. lugubris* are grayish to brownish above with yellow flecks that are often concentrated on the sides, squarish toetips, and a pale whitish venter (Stebbins 2003). Juvenile *A. f. niger* have green pigmentation, while *A. lugubris* juveniles do not.

Taxonomic Relationships

Aneides flavipunctatus niger is recognized as a subspecies based on geographic isolation from other populations, morphological and color variation, and ecology (Myers and Maslin 1948). Allozyme studies by Larson (1980) and subsequent reanalysis by Highton (2000) suggested that *A. f. niger* is a distinct lineage. More recent analyses of mitochondrial DNA data supported the allozyme analyses and identified another potentially distinct lineage of *A. flavipunctatus* in the Mount Shasta Region (Rissler and Apodaca 2007). Further genetic studies are ongoing and should help resolve these taxonomic issues. Current work is expanding sampling throughout the range of *A. f. niger* and includes both mitochondrial and nuclear markers (S. Reilly, pers. comm.).

Life History

Little is published on the life history of *Aneides flavipunctatus niger*, and we therefore rely on information from the northern subspecies, the specked black salamander (*A. f. flavipunctatus*) when data from *A. f. niger* are lacking (see the "Distribution" section). *Aneides flavipunctatus niger* is a terrestrial salamander that can be active year-round in streamside microhabitats (Lynch 1974). Like the majority of salamanders, it is most active on the surface at night, and more so during rain events. Females lay eggs in July or early August (Petranka 1998). In the laboratory, field-collected *A. f. flavipunctatus* from Mendocino County stayed with clutches until the young hatched (N. Staub, pers. obs. in Staub and Wake 2005), but it is unknown whether A. *f. niger* females also attend eggs in the field. Lynch (1981) examined 112 adult females across the range of *A. flavipunctatus* (including *A. f. niger* populations) and found that females carried 5–25 enlarged ovarian follicles, with fecundity increasing with body size. In the southern populations sampled in this study (which would contain *A. f. niger* samples), an average-sized female was 63 mm SVL, with an estimated clutch size of 9 (Lynch 1981). One record of a natural clutch of *A. f. niger* eggs was found more than 20 cm belowground (Van Denburgh 1895). Like many plethodontid salamanders, eggs undergo direct development, and fully formed, small juveniles appear at the surface shortly after the onset of fall rains, often in October or November (Lynch 1981).

No diet information has been published on *A. f. niger*. We presume that it is a generalized predator of small arthropods and other invertebrates. *Aneides flavipunctatus flavipunctatus* in northern coastal California are generalized predators that eat small invertebrates, including millipedes, beetles, termites, hymenopterans, flies, and collembolans (Lynch 1985).

Habitat Requirements

Aneides flavipunctatus niger is restricted to mesic forests in the fog belt of the outer Coast Range (Myers and Maslin 1948). While salamanders in the genus *Aneides* are sometimes quite arboreal, *A. f. niger* is a ground-dweller (Myers and Maslin 1948). *Aneides flavipunctatus niger* occurs in moist streamside microhabitats and is frequently found in shallow standing water or seeps (Myers and Maslin 1948, Lynch 1974; S. Barry pers., comm.). In these moist microhabitats, *A. f. niger* has been found under stones along stream edges and under boards near creeks (Myers and Maslin 1948). *Aneides flavipunctatus niger* also occurs in talus formations or rock rubble (S. Reilly, pers. comm.).

Distribution (Past and Present)

Aneides flavipunctatus niger is endemic to California and has a small range in the woodlands of the Santa Cruz Mountains in western Santa Clara, northern Santa Cruz, and southernmost San Mateo Counties. *Aneides flavipunctatus*

flavipunctatus occurs from Sonoma County north along the coast into southwestern Oregon and east to Shasta County (Stebbins 2003). Museum specimens exist for the Santa Lucia Mountains (LACM 141882-141883); however, we are unaware of other records for this region, and recent searches in this area have not been successful (S. Reilly, pers. comm.). Lynch (1981) reported that almost all localities of *A. flavipunctatus* (including sites within the range of *A. f. niger*) occurred below 600 m elevation in mesic forests that do not experience sustained freezes.

Some populations of *A. f. niger* have presumably been lost to development. Such losses are most likely to have occurred along the east slope of the Santa Cruz Range as older ranchland has been converted to subdivisions (S. Barry, pers. comm.). However, there is very little documentation of the historical distribution of this taxon.

Trends in Abundance

As for many plethodontids, documenting abundances is exceedingly difficult because *Aneides flavipunctatus niger* spends the majority of its time underground. No reliable population estimates exist for any sites, and therefore no declines in population abundance have been quantitatively documented. Some declines are likely to have taken place due to development and disturbance within the limited geographic range of this taxon. *Aneides flavipunctatus niger* is reported to have been abundant and easily found in the late 1950s, relatively abundant in the 1970s, and difficult to find in recent years (D. Wake, pers. comm.). Range-wide sampling efforts over the last few years have yielded only a handful of specimens (<15) at a few sites, including the UC Santa Cruz campus (S. Reilly, pers. comm.). This anecdotal evidence suggests that declines may have occurred and are possibly ongoing.

Nature and Degree of Threat

Aneides flavipunctatus niger habitat is vulnerable to the effects of logging, spring capping, and roadbuilding. The Peninsula Open Space Trust has acquired some of the vulnerable property in the northern part of the range, but there is still some risk of further ranchland subdivision (http://www.openspacetrust.org; S. Barry, pers. comm.). Climate change may pose some threats to this taxon, particularly given its small range and habitat specificity. Within the range of *A. f. niger*, mean annual temperatures are predicted to increase, though little change is expected in precipitation (reviewed in PRBO 2011). If conditions become significantly warmer and drier, this may affect opportunities for surface activity, although use of moist streamside microhabitats may minimize this effect. The frequency and size of fires in the Coast Ranges is expected to increase up to 50% by the end of the century, although impacts on the forested habitats used by *A. f. niger* are likely to be less severe than in more open habitats (Fried et al. 2004, Lenihan et al. 2008, Westerling and Bryant 2008). The extent of grassland vegetation is predicted to increase, and forested areas are predicted to decrease within the range of *A. f. niger*, which may negatively affect habitat availability (Lenihan et al. 2008).

Status Determination

Aneides flavipunctatus niger is an endemic salamander with a small geographic range in an area with some risk of additional development. However, ongoing declines and population losses have not been well documented, resulting in a Priority 3 designation.

Management Recommendations

Further protection of habitat is key for managing this taxon. In particular, special attention should be given to preserving forests, streamside and spring microhabitats, and natural talus formations within the Santa Cruz Mountains and to maintaining and enhancing connectivity between habitat patches.

Monitoring, Research, and Survey Needs

Basic ecological and life history information is almost entirely lacking for this taxon, as are estimates of current population abundances, limiting our ability to make more specific management recommendations. Surveys of microhabitats such as streams and seeps in forested areas should be conducted, though disturbance of microhabitat in order to find animals needs to be balanced with concerns regarding continuing decline. These surveys may be more effective if artificial cover objects are placed in suitable habitat, allowing for more comparable survey efforts among localities and increased detectability. Animals are most likely to be encountered at night when surface conditions are moist. Surveys are needed to establish estimates of abundance and to monitor population sizes over time. Upland terrestrial habitat usage is poorly known, and upland surveys would be useful for determining whether riparian buffers would be beneficial for *Aneides flavipunctatus niger*. Ecological and/or genetic studies of movement ecology and landscape genetics would be useful for understanding connectivity among populations and the permeability of different vegetation types.

INYO MOUNTAINS SALAMANDER

Batrachoseps campi Marlow, Brode, and Wake 1979

Status Summary

Batrachoseps campi is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 50% (55/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Batrachoseps campi is one of the largest and most robust members of the diverse plethodontid genus *Batrachoseps* (to 6.1 cm SVL) (Stebbins 2003). The head is relatively broad, and the tail is short compared to other *Batrachoseps* species. The body coloration is dark brown to blackish, with grayish or silvery dorsal spotting which ranges from very sparse to a continuous network. Individuals sometimes have a silvery or greenish cast overall (Stebbins 2003).

This species is the only salamander within its range and thus is unlikely to be confused with other species in the field. With the exception of the Kern Plateau salamander (*B. robustus*) and the largest individuals of the Tehachapi slender salamander (*B. stebbinsi*), other nearby *Batrachoseps* species are noticeably less robust and do not occur east of the Sierra crest. *Hydromantes* species may appear superficially similar

Inyo Mountains Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	5
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	10
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	0
viii. Projected impacts (10)	10
Total Score	55
Total Possible	110
Total Score/Total Possible	0.50

PHOTO ON PREVIOUS PAGE: Inyo Mountains salamander, Inyo County, California. Courtesy of Adam Clause.

but have five toes on the hind feet rather than four, as is the case in *Batrachoseps* (Stebbins 2003).

Taxonomic Relationships

This species is a member of the *Plethopsis* subgenus of *Batrachoseps*, which also includes the Oregon salamander (*B. wrighti*) from north central Oregon, and *B. robustus* from the Kern Plateau and western margins of Owens Valley in eastern California (Wake et al. 2002). *Plethopsis* can generally be characterized as a stout, robust group of *Batrachoseps* with relatively broad heads. *Batrachoseps campi* is morphologically distinguishable from other *Plethopsis* based on the presence of silvery iridophores, lack of dorsal stripe, and lack of white flecks ventrally (Marlow et al. 1979, Wake et al. 2002, Stebbins 2003). In addition, the species is genetically distinct at allozyme and mitochondrial loci (Yanev 1978, Yanev and Wake 1981, Jockusch and Wake 2002).

Life History

The life history of *Batrachoseps campi* is in need of further study. Its habitat differs somewhat from other closely related *Batrachoseps* species (e.g., *B. robustus, B. wrighti*), but information from these taxa is still likely to apply to *B. campi* in several respects. Surface activity occurs at night (Macey and Papenfuss 1991a) during which time the species presumably feeds on a variety of small insects. A life history study of the species is likely to provide important information for future management.

Habitat Requirements

Batrachoseps campi appears to be largely restricted to small patches of riparian habitat associated with perennial springs and limestone fissures in canyons of the Inyo Mountains. Localities where this species has been found contain wet rocks and fissures in close proximity to perennial water (Hansen and Wake 2005a). Salamanders are usually found under wet rocks or in clumps of moist ferns or other cover (Hansen and Wake 2005a). The species retreats into fissures and rock crevices when surface conditions are not favorable. Habitat surrounding these localized springs consists of Mojave Desert and Great Basin vegetational associations, which are unsuitable for the species. Individuals have only been found away from immediate proximity to flowing water at high-elevation sites in areas of pinyon–juniper woodland (Giuliani 1996, Hansen and Wake 2005a).

Distribution (Past and Present)

Batrachoseps campi is known from a small number of localities on the eastern and western slopes of the Inyo Mountains (Jennings and Hayes 1994a), although additional populations (presumably few) may be discovered in currently unsurveyed sites (Hansen and Wake 2005a). The known elevational range of the species extends from 490 to 2600 m (Macey and Papenfuss 1991a, Hansen and Wake 2005a).

Trends in Abundance

Populations may have declined or been extirpated at a few sites due to habitat modification, though population abundance data are essentially lacking (Papenfuss and Macey 1986). Although data are scarce, most known populations appear to be stable.

Nature and Degree of Threat

The primary threat to this taxon is habitat modification. The overall species range is very small (<20 ha total occupied habitat) and within that range consists of very small, isolated patches of suitable habitat (Hansen and Wake 2005a). The populations in each of these patches are isolated, so recolonization following extirpation is unlikely (Yanev and Wake 1981). Flash floods have scoured the canyon bottoms at some localities, destroying the riparian habitat, though salamander populations appear to persist and slowly recover (Giuliani 1996, Hansen and Wake 2005a). Damage to the sensitive riparian microhabitat from the capture and containment of springs (spring capping), mining, water diversion, and feral burro activity has occurred at other localities (Papenfuss and Macey 1986). Much of the species' range is unprotected and is vulnerable to further modification.

Status Determination

Due to its small range size and isolated populations, this species is inherently vulnerable to decline. The springs that are essential to its existeuce are scarce within the species' range and are vulnerable to impacts from water diversion and habitat degradation from humans, livestock, and feral mammals. There are few data on the habitat requirements of this species and the extent to which the isolated population can withstand these impacts. For all of these reasons, a Priority 3 status is justified.

Management Recommendations

The primary management priority for *Batrachoseps campi* is to protect existing habitat. Restoration of degraded habitat would be helpful. However, given the dearth of information on habitat requirements, it is very difficult to know what kinds of restoration would most benefit the species. Thus, restoration efforts need to be informed by the research and monitoring efforts outlined below. Until that time, the riparian areas around desert springs should be protected from modification, specifically with respect to changes in hydrology and vegetation. Some populations, such as the one at Barrel Spring, Inyo County, California, are likely to be sensitive to relatively minor changes in hydrology (D. Wake, pers. comm.).

Monitoring, Research, and Survey Needs

While the key management priority for this species is simply to protect habitat and minimize disturbances, restoration efforts would require basic research on the size, habitat requirements, and occupancy of sites throughout the species' limited geographic range. In the course of this work, surveyors would need to undertake basic life history research to gather information on population sizes (both census and genetically determined effective population sizes), yearly activity cycles, habitat occupancy, and basic ecological data. Because habitat protection alone is likely to be sufficient to safeguard this species, it may be best to carry out this work only in areas where disturbance to the habitat can be minimized.

Additional desert spring habitat near the known distribution needs to be searched during times when surface moisture is high enough to bring salamanders to the surface, although minimizing damage to these rare habitats is a critical priority. Higher-elevation populations may be more dispersed across the landscape, and surveys should take this into account. Monitoring efforts need to be initiated at localities that have experienced habitat degradation to quantify the ability of *Batrachoseps campi* to tolerate habitat changes that occur as springs are managed for human or livestock needs.

LESSER SLENDER SALAMANDER

Batrachoseps minor Jockusch, Yanev and Wake 2001

Status Summary

Batrachoseps minor is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 71% (78/110). This taxon had not yet been described at the time of the previous Species of Special Concern revision and was therefore not evaluated.

Identification

Salamanders in the genus *Batrachoseps* are generally characterized as elongate, slender plethodontid salamanders with extremely reduced limbs, elongate, worm-like bodies, and extremely long tails that are often longer than the SVL of the animal. Many species have been identified in the last two decades, many of which are morphologically cryptic and some of which have extremely small ranges. *Batrachoseps minor* is the smallest species of *Batrachoseps* (up to 3.4 cm SVL). The coloration is dark blackish brown on the sides and dorsum, sometimes with a lighter brown or tan dorsal stripe along the back (Stebbins 2003). Dense white speckles are present on the ventral surface (Jockusch et al. 2001).

This species is morphologically similar to the more common and microsympatric black-bellied slender salamander (*B. nigriventris*), though its limbs and feet are relatively more

Lesser Slender Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	10
iii. Population concentration/migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	3
vi. Population trend (25)	25
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	10
Total Score	78
Total Possible	110
Total Score/Total Possible	0.71

PHOTO ON PREVIOUS PAGE: Lesser slender salamander, San Luis Obispo County, California. Courtesy of William Flaxington.

robust (Hansen and Wake 2005b). Subadults, in particular, can be difficult to tell apart in these species, particularly in some preserved specimens. Molecular identification may be required in some of these cases.

Taxonomic Relationships

Batrachoseps minor was previously included in *B. pacificus* (sensu lato). Populations now regarded as *B. minor* were recognized largely on the basis of mitochondrial DNA and allozymes, though some morphological features distinguish this species from other members of the *B. pacificus* complex (Jockusch et al. 2001). *Batrachoseps minor* is closely related to the San Simeon slender salamander (*B. incognitus*), and the garden slender salamander (*B. major*) (Jockusch et al. 2001, Jockusch and Wake 2002).

Life History

The life history of *Batrachoseps minor* has not been studied. The species presumably feeds on very small insects and other terrestrial invertebrates and exhibits similar ecological characteristics as other members of the *B. pacificus* complex.

The species is microsympatric throughout the entirety of its range with *B. nigriventris*, which is both more widespread and more common than *B. minor* within the range (Hansen and Wake 2005b). It is possible the *B. nigriventris* ecologically replaces *B. minor* at lower elevations (Hansen and Wake 2005b), though the extent or effects of competition between these species has not been studied.

Habitat Requirements

Batrachoseps minor is found on steep north and east-facing mesic slopes within its known range (Jockusch et al. 2001). Known localities have a canopy of oak, tanbark, madrone, and laurel with a poison oak thicket understory (S. Sweet, pers. comm.). These sites remain damp much longer than surrounding slopes, and are 2–3°C cooler at the litter/soil interface (S. Sweet, pers. comm.). Very few localities are known, and habitat requirements need further study.

Distribution (Past and Present)

Batrachoseps minor is found only in north central San Luis Obispo County. It is present in the southern part of the San Lucia Range above 400 m, ranging from the vicinity of Black Mountain south and east into the Paso Robles and Santa Rita drainages (Jockusch et al. 2001). Populations farther south have been assigned to this species based on morphology and molecular information (E. Jockusch, pers. comm.).

Trends in Abundance

This species was apparently once common within its range. Many specimens were collected throughout the 1970s before the species was described, but the species subsequently became much more difficult to find (Jockusch et al. 2001; D. Wake, pers. comm.). Few specimens have been reported in the literature in the last decade, although several unreported sightings are known, and populations may now be increasing to some degree (Hansen and Wake 2005b; E. Jockusch, pers. comm.; D. Wake, pers. comm.; S. Sweet, pers. comm.). During 1971–1975, field crews associated with the Museum of Vertebrate Zoology undertook 10 field trips that collected 265 *Batrachoseps* from sites known to support *B. minor*. This collection comprised 206 *B. minor* (77% of the total) and 59 *B. nigriventris* (S. Sweet, pers. comm.). In 12 surveys conducted since 2011, 27 *B. minor* have been found along with 60 *B. nigriventris* (31% of the total; S. Sweet, pers. comm.), suggesting that the frequency with which *B. minor* is detected relative to *B. nigriventris* has decreased and that the total number of *Batrachoseps* found is smaller today than it was previously. No obvious changes in habitat or plant cover between the early 1970s and the present that might explain these changes have been observed (S. Sweet, pers. comm.).

Nature and Degree of Threat

Little information is available concerning any aspect of the biology of this species, making threats difficult to characterize with certainty. Some habitat modification resulting from land conversion to vineyards has occurred within the range, and the invasion of exotic plants has caused changes to the understory in some areas (Hansen and Wake 2005b; D. Wake, pers. comm.); both of these factors are presumably detrimental to the species' persistence. That said, the extent to which such land conversion has occurred has been disputed (S. Sweet, pers. comm.) and a large amount of apparently suitable habitat still remains in the general region. The species was formerly detected in large numbers at wineries (Hansen and Wake 2005b; E. Jockusch, pers. comm.; D. Wake, pers. comm.). Other factors contributing to the declines deserve further study. As this species seems to be limited to relatively mesic areas within its range, changing hydrology and temperature associated with climate change has the potential to render much of the current habitat unsuitable for this species. The marked declines in abundance over the last few decades may indicate a degree of sensitivity to habitat or climatic conditions or, alternatively, may simply represent a temporary and cyclical decline associated with moderate-term changes in climate (rainfall specifically; S. Sweet, pers. comm.). Here, we interpret the observed pattern with precaution in mind, treating the documented declines in abundance as real and noncyclical, but acknowledging that an alternative possibility exists and that further study and published data are needed.

Status Determination

Batrachoseps minor is a California endemic and has an exceedingly small geographic range. Large apparent declines have occurred since the 1970s, and the threats to this taxon are poorly understood, leading to a Priority 1 status.

Management Recommendations

Given what is currently known about this species, little can be done in terms of management. Few sites have been confirmed (using molecular data) to support *Batrachoseps minor*, and these sites should be protected from further modification that is likely to be detrimental to salamander populations. Additional information on the range, habitat requirements, and environmental sensitivity of the species is needed to help guide future management.

Monitoring, Research, and Survey Needs

Batrachoseps minor is poorly known biologically, and published accounts of even the most basic habitat and ecological data are largely lacking for the species. Additional and ongoing surveys for this taxon are needed to help determine its range, both geographically and ecologically. However, careful attention needs to be paid to effective identification of specimens that are found. Because *B. minor* is so similar in appearance to *B. nigriventris*, and the two species occur in microsympatry, surveyors need to have extensive experience distinguishing different *Batrachoseps* species from each other. Subadult specimens of *B. minor* may require molecular identification unless and until field-validated morphological characters can be identified. As the status of remaining populations is unknown, a reasonable management policy would be that no *Batrachoseps* from the known or suspected range of *B. minor* be removed from the wild unless the collector has extensive experience identifying these species. Rather, individuals should be photographed and nondestructively sampled, preferably by removing a small portion from the end of the tail (~2 mm) and genotyped to establish identification. If a few replicate DNA sequences from both the nuclear and mitochondrial genomes could be established as reliable barcoding genes, DNA typing could be accomplished quickly and inexpensively. Surveys should take place when surface conditions are appropriately moist to enhance the likelihood of finding populations of this elusive salamander. The chances of finding *B. minor* without disturbing its natural habitat would likely be increased by establishing a transect of artificial cover objects (plywood boards) throughout the known range. Nighttime surveys during rain events might also be productive. In addition, nearby areas should continue to be surveyed for this species, as its distribution could potentially be larger, both ecologically and geographically, than is presently known. Higher-elevation areas, such as those in the vicinity of Santa Rita and Old Creek Road, San Luis Obispo County, should be surveyed if access to private land in these areas can be established. It is possible that the known localities occur near the lower elevational range of the species, and larger populations exist at higher elevations (E. Jockusch, pers. comm.). Recent and repeated surveys in some of these areas have failed to detect this species, which suggests elevation may not be an important factor (S. Sweet, pers. comm.). Nevertheless, the species is clearly less detectable than it was decades ago and additional published data are needed to better characterize the known distribution and abundance. Additional research into potential causes of the declines in detectability should also be pursued. In particular, screens of museum specimens for the presence of pathogenic fungi might be fruitful (D. Wake, pers. comm.), as could study of decadal scale climate and rainfall patterns within the species known range (S. Sweet, pers. comm.).

RELICTUAL SLENDER SALAMANDER

Batrachoseps relictus Brame and Murray 1968

Status Summary

Batrachoseps relictus is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 60% (66/110). It was also considered a Species of Special Concern during the previous evaluation (Jennings and Hayes 1994a); however, the range of the species has since been greatly reduced as a consequence of taxonomic revisions.

Identification

As is typical of its genus, *Batrachoseps relictus* is a small, elongate, worm-like salamander with a slender body, long tail, and tiny limbs. The dorsal coloration is blackish brown with a lighter, often indistinct dorsal stripe that may be reddish, yellowish, or dark brown (Stebbins 2003). *Batrachoseps relictus* is one of the smallest members of its genus. SVLs of mature animals collected at the type locality in the lower Kern River Canyon (see the "Distribution" section) averaged 30.2 mm, while those from Breckenridge Mountain averaged somewhat larger at 39 mm SVL (Jockusch et al. 2012). *Batrachoseps relictus* also has relatively few trunk vertebrae, with a modal number of 17 from the type locality (Brame and Murray 1968) and counts as low as 17 occurring with low frequency in the

Relictual Slender Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	10
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	7
vi. Population trend (25)	15
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	66
Total Possible	110
Total Score/Total Possible	0.60

PHOTO ON PREVIOUS PAGE: Relictual slender salamander, Kern County, California. Courtesy of William Flaxington.

Breckenridge Mountain populations (Jockusch et al. 2012).

Several other species of *Batrachoseps* occur in the same region of the southern Sierra Nevada, and geographic range is the best way to distinguish animals in the field. Individuals from the upper Kern River Canyon (Greenhorn Mountains slender salamanders, *B. altasierrae*) that were previously considered a part of *B. relictus* (see the "Taxonomic Relationships" section) have relatively longer trunks, smaller heads, shorter limbs, and smaller feet (Jockusch et al. 2012). Female *B. altasierrae* have fewer maxillary teeth, and the vomerine teeth in both sexes are patchily distributed, compared to being arranged in rows in *B. relictus* (Jockusch et al. 2012).

In the lower Kern River Canyon, the range of *B. relictus* overlaps with Kern Canyon slender salamanders (*B. simatus*) and the yellow-blotched ensatina (*Ensatina eschscholtzii croceator*; Brame and Murray 1968). Unlike *B. relictus*, *B. simatus* is not closely associated with water, and populations of *B. relictus* at elevations where *B. simatus* occurs are likely extirpated (see the "Distribution" section). *Ensatina eschscholtzii croceator* is a larger, more robust salamander and is easily distinguished by conspicuous yellow blotches on the dorsum and a much larger body form (Stebbins 2003).

Taxonomic Relationships

The populations included in *Batrachoseps relictus* have changed considerably since its original description. Brame and Murray (1968) considered several geographically disjunct populations as belonging to *B. relictus*, most of which are now recognized as distinct species (Yanev 1978, Yanev 1980, Jockusch et al. 1998, Wake and Jockusch 2000, Jockusch et al. 2001). Populations in the Sierra Nevada from the Merced River to the Kern River were considered a part of the *relictus* group (Yanev 1980) and were split into four allopatric species by Jockusch et al. (1998). At that time, *B. relictus* was thought to range from the Tule River drainage to the lower Kern River Canyon, including populations in the Greenhorn Mountains (Jockusch et al. 1998, Jockusch and Wake 2002). Since then, populations from the upper Kern River have been found to be morphologically distinct from salamanders at the *B. relictus* type locality, and have been described as the new species *B. altasierrae*, the Greenhorn Mountains slender salamander. (Jockusch et al. 2012). Populations of *Batrachoseps* on Breckenridge Mountain were discovered in 1979. Jennings and Hayes (1994a) designated this putative taxon as a Species of Special Concern. Recent morphometric analyses have shown that populations from Breckenridge Mountain are most similar to *B. relictus* from the type locality, and are now included as *B. relictus* (Jockusch et al. 2012). Given the description of the new taxon *B. altasierrae*, the classification of Breckenridge Mountain populations as *B. relictus*, and the presumed extirpation of the type locality (see the "Distribution" section), extant *B. relictus* only occur on Breckenridge Mountain under the current taxonomic arrangement.

While we follow the recommendations of Jockusch et al. (2012) to recognize *Batrachoseps* from Breckenridge Mountain as *B. relictus*, it is important to note that their phylogenetic analyses of mitochondrial DNA show these populations as nested within *B. simatus*, the Kern Canyon slender salamander. Jockusch et al. (2012) argued that allozyme data and unpublished nuclear data recovered a different pattern that corroborated the distinctiveness of *B. relictus*, and that the mitochondrial DNA results were potentially explained by introgression from *B. simatus* into *B. relictus*. This interpretation appears to be reasonable. However, given the complexity of this group, it remains possible that additional work may lead to further taxonomic revisions.

Life History

Very little is known about the natural history of *Batrachoseps relictus*, and much of the ecological literature published under this name refers to what is now classified as *B. altasierrae*. *Batrachoseps relictus* on Breckenridge Mountain (1700–2000 m elevation) have been found surface active under cover objects from May to early

October (Jockusch et al. 2012). At lower elevations in the Kern River Canyon, animals have been collected between January and May, suggesting that surface activity is possible over most of the year and varies with elevation. Association with aquatic microhabitats likely facilitates extended periods of surface activity (see the "Habitat Requirements" section). Like other plethodontid salamanders, *B. relictus* is a direct developer that lays terrestrial eggs. Females have been found with yolked ova or eggs in May and June (Jockusch et al. 2012). A communal nest with roughly 125 eggs and 20 adults was discovered beneath a rock in a seep during June 1979 at the high-elevation site on Breckenridge Mountain (R. Hansen, pers. obs., in Jockusch et al. 2012; observation incorrectly ascribed to *B. simatus* in Stebbins 1985). Diet has not been studied in *B. relictus*. Presumably they use their projectile tongues to catch small invertebrates, as do other *Batrachoseps* species (Hansen and Wake 2005c).

Habitat Requirements

Individuals from the type locality in the lower Kern River Canyon have been found associated with perennial springs, seeps, and small creeks in oak woodland below 750 m (Hilton 1948, Brame and Murray 1968). This close association with water was described as "semiaquatic" by Brame and Murray (1968). Animals have been found under cover objects with water beneath them and observed in the water (Hilton 1948, Jockusch et al. 2012). On Breckenridge Mountain the dominant vegetation type at extant localities is pine–fir forest (Jockusch et al. 2012). East of Squirrel Meadow at 2000 m elevation, *Batrachoseps relictus* is typically associated with a small seep and sandy or gravel substrate (Jockusch et al. 2012). Use of upland habitat away from water is unknown, but two adults were found 45 m upslope from seep habitat at the Squirrel Meadow site (Jockusch et al. 2012). At Lucas Creek, the lower-elevation extant locality on Breckenridge Mountain (1665 m), all *B. relictus* to date have been found under cover objects along a 750 m stretch of stream (Jockusch et al. 2012).

Distribution (Past and Present)

The type locality is in the lower Kern River Canyon, 150 yards above the junction of state Highway 178 and the road turnoff to Democrat Hot Springs and Resort (Brame and Murray 1968). Despite repeated, careful searches, *Batrachoseps relictus* have not been seen at the type locality since 1970 (Jockusch et al. 2012; incorrectly reported as 1971 elsewhere). Extirpation of the type locality may have been caused by the degradation of the sensitive seep and spring habitat due to the construction of Highway 178 (Hansen 1988). With the presumed extirpation of the type locality, *B. relictus* is now thought to be restricted to two localities on Breckenridge Mountain, and has the smallest known range for any described species of *Batrachoseps*. Populations north of the Kern River including the Greenhorn Mountains are no longer considered a part of *B. relictus* (see the "Taxonomic Relationships" section). The known elevation range is from 480 m in the Lower Kern Canyon River up to 2000 m on Breckenridge Mountain (Jockusch et al. 2012).

Trends in Abundance

Declines are suspected at one extant site, the area east of Squirrel Meadow on Breckenridge Mountain. The locality was first discovered in 1979 but later degraded by construction of a logging road through *Batrachoseps relictus* habitat. Salamanders were not seen at this site for two decades, with declines presumed to be due to habitat degradation from road construction, wildfire, and timber harvest (Jockusch et al. 2012). More recent surveys of the site have found that populations appear to be rebounding to some degree (Jockusch et al. 2012). Whether such variation in abundance over time is typical, due to detection difficulty, or actual anthropogenic declines is unknown.

Nature and Degree of Threat

The major threat to *Batrachoseps relictus* is habitat degradation, particularly of sensitive spring and seep habitat. Climate change is expected to increase temperatures in the Sierra Nevada, although changes in precipitation and fire regime are highly uncertain and large regional variation is expected across the mountain chain and at different elevations (reviewed in PRBO 2011). If conditions become warmer and drier, this would presumably negatively affect *B. relictus* populations, although microhabitat characteristics are likely key to determining surface activity and population stability. Large reductions in snowpack are predicted for the Sierra Nevada (reviewed in PRBO 2011), which may decrease the availability of streamside habitat for *B. relictus*.

Status Determination

The extremely limited geographic range of *Batrachoseps relictus*, the small number of known extant populations, and apparent extirpation of the type locality contribute to a Priority 1 Species of Special Concern designation for the species.

Management Recommendations

Protecting the two remaining localities from habitat degradation is critical to the persistence of *Batrachoseps relictus*. Given the extremely sensitive and restricted range of the species, any habitat modification should be avoided where the species still occurs. Road construction should be avoided, and road use and maintenance activities should be restricted, or ideally eliminated altogether. Timber harvest and use of heavy equipment in or near seeps and streams should be eliminated. If the type locality is confirmed to be extirpated, then repatriation of the species to the type locality may be appropriate. However, the lack of genetic information from this site (no genetic samples exist) and the overall state of flux in the classification of southern Sierra Nevada *Batrachoseps* may argue against such reintroductions pending further molecular systematics work on the group as a whole. Although the extent and use of upland habitat is unknown, protection of riparian buffers would almost certainly benefit this species in disturbed areas. In addition, it is probably reasonable to assume that livestock grazing should be eliminated from areas where the species still occurs, at least until field ecological studies indicate that grazing is compatible with the salamander's habitat requirements.

Monitoring, Research, and Survey Needs

Basic life history and population biology information is severely lacking for this species, and represents a critical research need. A key survey need is to attempt to locate additional populations, particularly at mid-elevations on Breckenridge Mountain, which are largely unexplored (Jockusch et al. 2012). High-priority sites for surveys include streamside and seep habitats on the north face of the mountain. Monitoring should continue at the lower Kern River Canyon localities to confirm extirpation. Populations at the higher-elevation Breckenridge Mountain locality went undetected for many years, and it remains possible that animals could be rediscovered at the type locality. If so, the collection of genetic samples would be invaluable to support or refine the current taxonomy of the species, and to help determine patterns of connectivity among remaining populations.

CALIFORNIA GIANT SALAMANDER

Dicamptodon ensatus (Eschscholtz 1833)

Status Summary

Dicamptodon ensatus is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 66% (56/85). This species was not previously considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Dicamptodon ensatus is a large (6.3–17.3 cm SVL) robust salamander with a very large head and stout limbs. The dorsal coloration is a coppery tan to dark brown irregular marbled pattern on a tan to light reddish brown background. The venter is paler and usually unmarked, although marbling often extends onto the chin, throat, and under the legs. The marbling coloration is often brighter in young metamorphs compared to adults. The tail is laterally compressed, the skin is smooth, and post-metamorphic juveniles and adults lack tubercles on their feet (Stebbins 2003).

Larvae are of the stream type, with short bushy gills and a tail fin that begins at the insertion of the hind limbs and extends posteriorly to the tail tip. Larval dorsal coloration is light brown, and ventral coloration is white to yellowish white (Nussbaum 1976). There is also a pale

California Giant Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	10
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	10
v. Ecological tolerance (10)	10
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	3
Total Score	56
Total Possible	85
Total Score/Total Possible	0.66

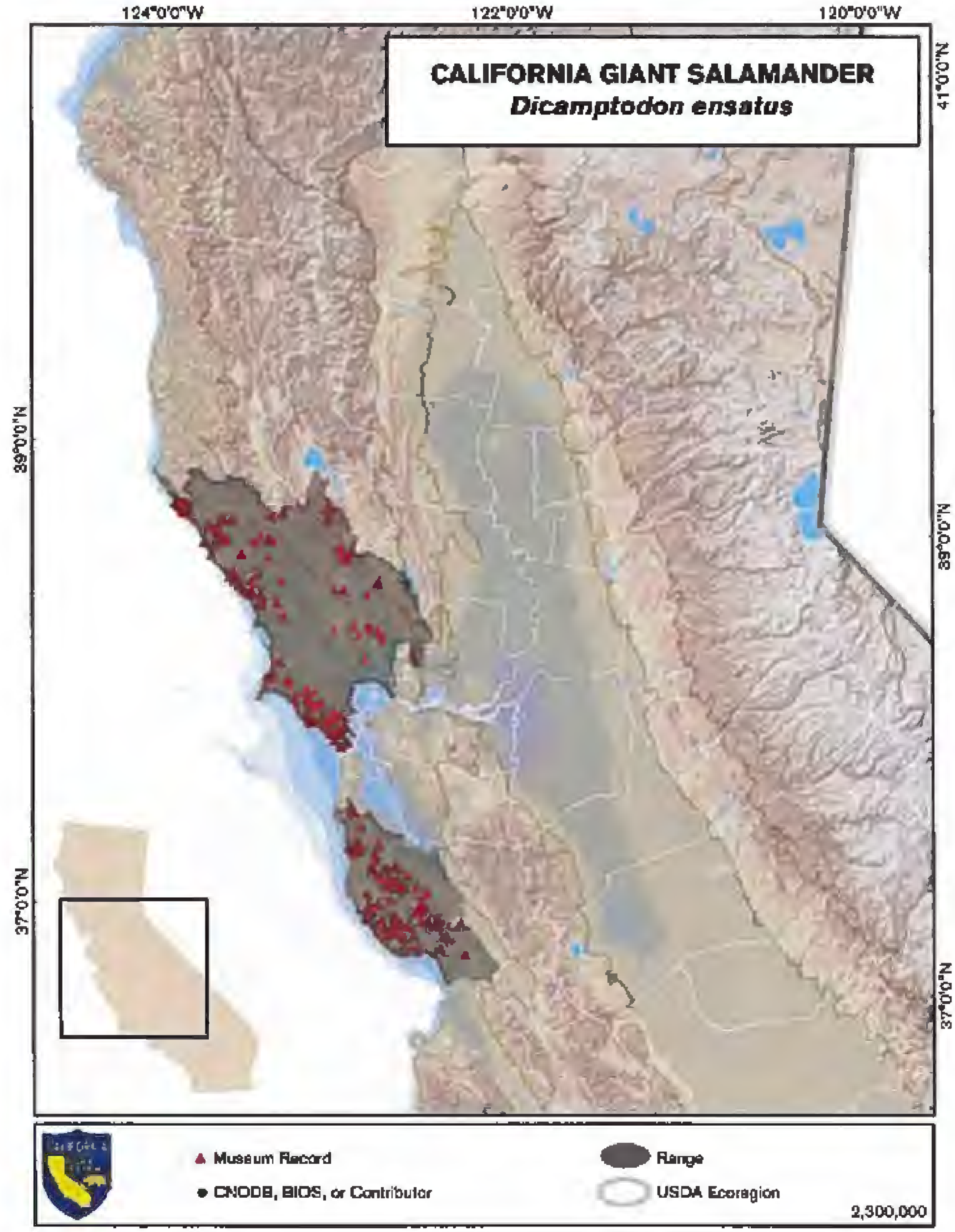


PHOTO ON PREVIOUS PAGE: California giant salamander, Santa Cruz County, California. Courtesy of Nicholas Hess.

eye stripe behind each eye, and the snout is depressed (Petranka 1998). The toe tips of larvae are black and cornified (Petranka 1998).

In California, *D. ensatus* is largely indistinguishable from the more widely distributed coastal giant salamander (*D. tenebrosus*) based on morphology alone. However, both geographic range and genetic markers distinguish these two species.

Taxonomic Relationships

Good (1989) split California *Dicamptodon* into two species, *D. tenebrosus* in the north and *D. ensatus* in the south, on the basis of allozyme data. A 4.7 km hybrid zone exists between the two species approximately 10 km north of Gualala in Mendocino County (Good 1989). Otherwise, the two species are allopatric.

Life History

Adult *Dicamptodon ensatus* are terrestrial and return to streams to breed during the fall rainy season (Kessel and Kessel 1943a) and in the spring (Stebbins 2003). One *D. ensatus* nest of approximately 70 eggs was found under a submerged wooden plank in a rapidly flowing stream in the Santa Cruz Mountains, San Mateo County, during June (Henry and Twitty 1940). Female *D. tenebrosus* guard nests through hatching (Nussbaum et al. 1983), and an adult female *D. ensatus* was found near the Santa Cruz Mountains nest (Henry and Twitty 1940), suggesting that both species may guard their eggs. Eggs in early developmental stages are pure white and approximately 5.5 mm in diameter (Petranka 1998). The larval stage lasts approximately 18 months, with larvae growing 8–12 mm in TL per month during the warmer months in their first year. Larvae reach 10 cm TL within a year of hatching and metamorphose in late summer at 13–14 cm TL (Kessel and Kessel 1943a, Kessel and Kessel 1943b, Kessel and Kessel 1944). The prevalence of paedomorphosis in this species is unknown, although it can be quite common in *D. tenebrosus*. A paedomorphic population of *D. ensatus* has been reported from caves on the UC Santa Cruz campus (B. Sinervo, unpublished data).

Bury (1972) reported gut contents of 12 adults from Del Norte, Humboldt, and Marin Counties (i.e., a mix of *D. ensatus* and *D. tenebrosus*). Eight out of 12 specimens contained one or more vertebrates, including California slender salamanders (*Batrachoseps attenuatus*), lizards, mice, shrews, and voles. Other prey included large invertebrates such as land snails and smaller invertebrates such as beetles and crickets (Bury 1972). Cannibalism has been documented in adults (Anderson 1960). No diet data from larvae are available for this species, though they are presumed to have similar diets to larval *D. tenebrosus* (Petranka 1998), which primarily consume aquatic insects and other invertebrates (Parker 1994).

Habitat Requirements

Dicamptodon ensatus occurs in mesic coastal forests (oak woodland and coniferous forest; Petranka 1998), and coastal chaparral habitat is used in southern Marin County and San Mateo County (N. Waters, pers. comm.). Very little is known about terrestrial habitat use by adults and metamorphs, although adults are occasionally found surface active or under cover objects in wet conditions (Petranka 1998). One unusual record exists of an adult *D. ensatus* in a tree vole (*Arborimus pomo*) nest 2.4 m off the ground, the only account of arboreality in this species (D. Hamilton and W. Roberts, unpublished data in Forsman and Swingle 2007).

Breeding and larval development occurs in cold permanent and semipermanent streams (Petranka 1998). Larval habitat use is poorly studied. In one stream, small larvae were found in slow-moving water near the banks during heavy flows, and as flows decreased they moved into the main stream channel where larger larvae occurred (Kessel and Kessel 1943a, Kessel and Kessel 1943b).

Distribution (Past and Present)

Dicamptodon ensatus is endemic to California, occupying a small range from sea level to

900 m in elevation along the coast in two isolated areas near San Francisco Bay (Stebbins 2003). North of the Bay, they occur in the outer Coast Ranges from near the southern border of Mendocino County south through Marin County, and the inner Coast Ranges in Napa, Sonoma, Lake, and Solano Counties (Good 1989). South of the Bay, they occur in the Santa Cruz Mountains in San Mateo, Santa Clara, and Santa Cruz Counties (Good 1989; N. Waters, pers. comm.). *Dicamptodon ensatus* has not been recorded in the East Bay (Stebbins 2003). Nussbaum (1976) mentioned an unconfirmed sight record from the Santa Lucia Mountains in Monterey County. Multiple surveys by several researchers over the decades have attempted to verify this account with no individuals detected (N. Waters, pers. comm.). While extirpations have not been documented, urbanization, agriculture, and timber harvest have likely resulted in some population losses, particularly due to development in the southern part of the range (Bury 2005; S. Barry, pers. comm.)

Trends in Abundance

Given the paucity of information, this species is currently considered data deficient for the population trend metric. However, it is likely that abundance has been reduced in habitats disturbed by urbanization, roadbuilding, logging, or water diversions (Bury 2005).

Nature and Degree of Threat

The Santa Cruz Mountains isolate is currently largely contained within a network of public parkland, though the extent of possible losses in this region due to past development is poorly understood (N. Waters, pers. comm., S. Barry, pers. comm.). Coast Range populations in the north are likely subject to negative effects from timber harvest and development, though this area is less urbanized than the southern part of the range. Disturbances such as clear-cutting and road construction can lead to lower abundances in *Dicamptodon tenebrosus* (Corn and Bury 1989, Welsh and Ollivier 1998). Other threats include fragmentation of riparian habitat, water diversions for municipal and agricultural use, and road mortality (N. Waters, pers. comm.).

Climate change may negatively impact *D. ensatus*, although uncertainty in climate projections coupled with limited ecological information makes assessing risk difficult. Mean annual temperature is expected to increase while projected changes in precipitation are likely modest, leading to warmer and possibly drier conditions in northwestern and central California (reviewed in PRBO 2011). At the same time, upwelling is expected to intensify (Snyder et al. 2003, Lebassi et al. 2009). This may increase fog development and contribute to cooler, moister conditions along the coast, potentially ameliorating effects of warming or drying within the range of *D. ensatus*. The frequency and extent of wildfire is expected to increase in the region encompassing the southern part of the range, with predicted increases in area burned of up to 50% (Fried et al. 2004, Lenihan et al. 2008, Westerling and Bryant 2008). How fire regime will change in the northern part of the range is less well understood (reviewed in PRBO 2011). Effects of wildfire on *D. ensatus* are unknown, though mortality and habitat degradation due to fire has been documented in other stream-breeding amphibians (e.g., Gamradt and Katz 1997, Pilliod et al. 2003). In northwestern California, vegetation communities are expected to shift from moist conifer to drier mixed evergreen forest, with reductions in Douglas fir and redwood forest in particular (Lenihan et al. 2008, PRBO 2011), which may impact the availability of *D. ensatus* habitat.

Status Determination

Dicamptodon ensatus is an endemic, ecologically specialized salamander with a small geographic range that is restricted to an area with a high human population density. These factors combine to place it at high risk of habitat loss and disturbance. However, data are not available to determine whether ongoing declines and population losses have occurred, resulting in a Priority 3 designation for this species.

Management Recommendations

We know little about the basic biology of this species, which makes it difficult to formulate management recommendations beyond minimizing disturbances to existing habitat. Habitat protection may be particularly important for small headwater streams where siltation and other stream disturbances are known to severely impact other *Dicamptodon* species. Construction and use of roads should be eliminated or minimized within *D. ensatus* habitat, particularly during the breeding season. Riparian buffer vegetation should be retained in areas that are developed or harvested, though efficacy of buffers and optimal buffer widths for this taxon are unknown.

Monitoring, Research, and Survey Needs

Distribution, abundance, habitat requirements, and life history of *Dicamptodon ensatus* all need further study. Most research to date has focused on the more widespread *D. tenebrosus* to the north and was conducted before the two species were recognized as distinct. This substantial knowledge gap needs to be addressed with basic ecological studies. Nothing is known about dispersal in this species, especially the importance of movement through terrestrial habitats. Both mark–recapture and landscape genetic studies are needed for *D. ensatus*. Studies are also needed that examine the efficacy of streamside buffers in ameliorating the effects of disturbance on stream habitats. Such studies should be replicated both north and south of San Francisco Bay, given that these are completely isolated population segments living in different habitats. Distributional surveys are particularly needed in the Inner Coast Range portion of the northern range (N. Waters, pers. comm.).

While larvae are easy to find by searching aquatic habitats, transformed *D. ensatus* are infrequently encountered using typical amphibian survey techniques. For example, only 12 individuals were captured in 18,032 trap nights over 3 years of pitfall trapping along 840 m of drift fence in suitable habitat at Point Reyes National Seashore (G. Fellers and D. Pratt, unpublished data, in Fellers et al. 2010). In the same study, no *Dicamptodon* were detected under 84 coverboards during nearly 2000 coverboard checks. However, culvert removal using heavy equipment uncovered aggregations of >20 adults at the same study sites, suggesting that terrestrial sampling may severely underestimate abundance (Fellers et al. 2010). Another account from Santa Cruz County reported several adults and eggs getting washed out of a drill hole made 6 m into a hillside to access a subterranean spring (Dethlefsen 1948). These reports suggest that metamorphosed individuals may be largely subterranean in their habits, a possibility that needs further investigation.

SOUTHERN TORRENT SALAMANDER

Rhyacotriton variegatus Stebbins and Lowe 1951

Status Summary

Rhyacotriton variegatus is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 75% (83/110). Previously it was also considered a Species of Special Concern, although at a lower priority level. Additional research on ecology and phylogeography since Jennings and Hayes (1994a) supports this change in status.

Identification

Rhyacotriton variegatus is a small to medium-sized salamander (5 cm SVL) (Welsh and Lind 1992, Tait and Diller 2006), with a small head and a short, laterally compressed tail (Stebbins 2003). Expanded square-shaped glands lateral and posterior to the vent in adult males distinguish this genus from all other North American salamanders (Petranka 1998). *Rhyacotriton* has large bulging eyes, with eye diameter roughly equal to the distance between the anterior edge of the eye and the tip of the snout (Stebbins 2003). The dorsal ground color is brownish to olive, and the venter is yellow to yellowish green with a sharp, abrupt demarcation between the dorsal and ventral coloration (Petranka 1998). California *R. variegatus* are heavily speckled with small dark spots

Southern Torrent Salamander: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	3
v. Ecological tolerance (10)	10
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	10
Total Score	83
Total Possible	110
Total Score/Total Possible	0.75

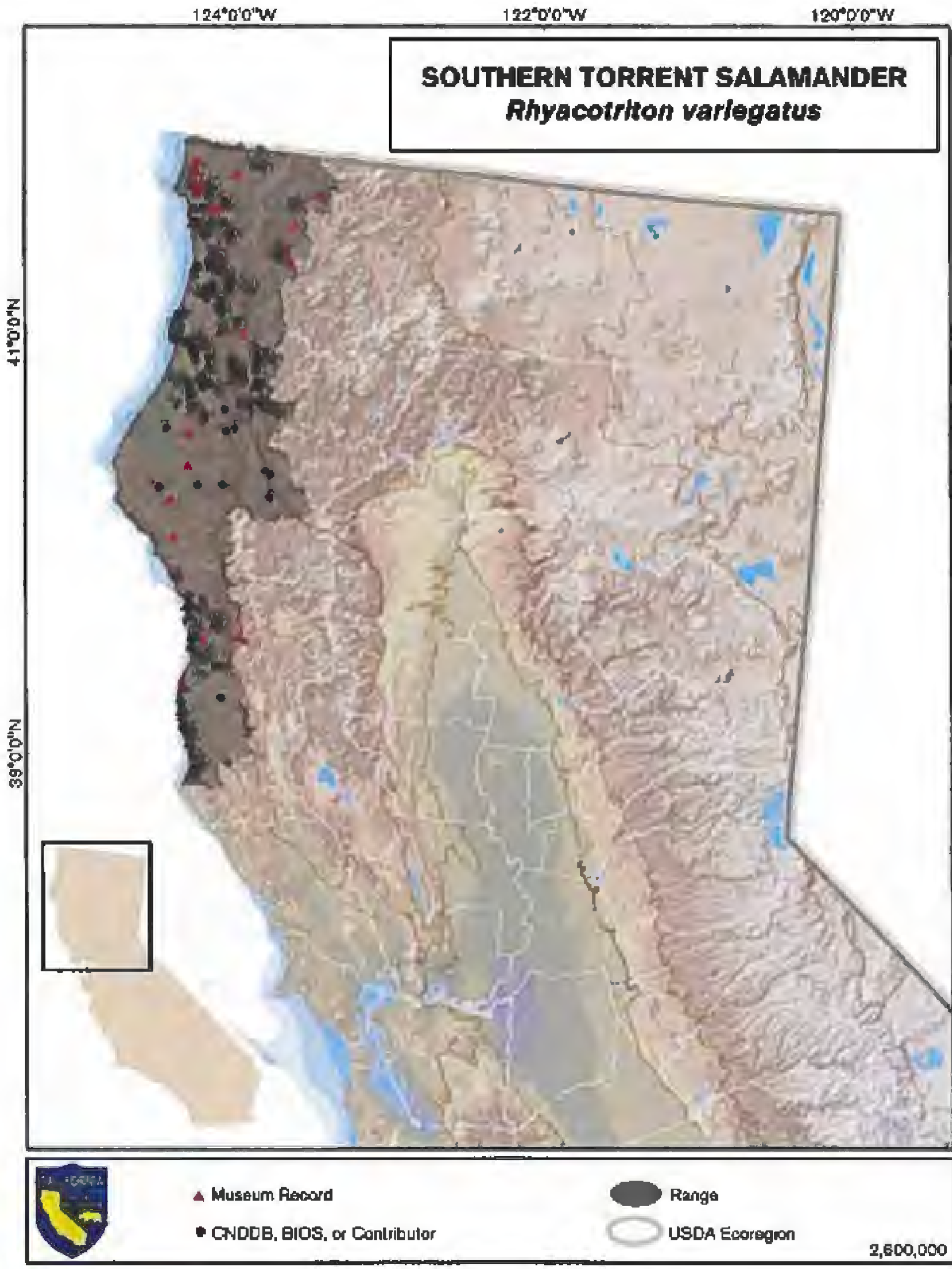


PHOTO ON PREVIOUS PAGE: Southern torrent salamander, Mendocino County, California. Courtesy of Robert Thomson.

on the dorsum and venter (Good and Wake 1992).

Larvae are of the stream type and have morphological adaptations unique to headwater specialists (Valentine and Dennis 1964). Larvae have short stubby gills and a tail fin that does not extend anteriorly onto the trunk. The dorsum is light brown above, the venter is cream to yellow, and the body is sprinkled with dark speckling above and below except on the tail fin. The eyes are prominent and dorsally positioned (Petranka 1998).

Taxonomic Relationships

Rhyacotriton variegatus has been recognized as a species since 1992 based on protein variation (Good and Wake 1992). Miller et al. (2006) identified three mitochondrial DNA clades within *R. variegatus*. The California clade/southern Oregon clade split occurs at the Smith River in California, a common biogeographic boundary. Miller et al. (2006) concluded that the California clade constitutes an evolutionarily significant unit (sensu Moritz 1994). The California clade is endemic to the state with a ~50% smaller range than the species as a whole, and the southern Oregon clade animals in California have an extremely small range. Although Miller et al. (2006) recognized these clades as potential management units, we consider them as a single taxon here pending additional research on their geographic ranges and genetic distinctiveness using additional molecular markers.

Life History

Breeding may occur throughout much of the year. Males produce sperm year-round, with peak production from February through April (Humboldt County; Tait and Diller 2006). California females have been found carrying spermatophores from February through June (Stebbins and Lowe 1951, Tait and Diller 2006), and females from an Oregon population had cloacal spermatophores as late as October (Nussbaum and Tait 1977).

Females produce smaller clutches than most similarly sized stream-breeding salamanders (Petranka 1998), with gravid females carrying from 4 to 16 ovarian eggs (Nussbaum and Tait 1977, Good and Wake 1992, Tait and Diller 2006). Karraker (1999) found a nest with 11 cream-colored eggs deposited singly beneath a small boulder in a first-order stream channel in Humboldt County.

Developmental times are slow, with oviposition to sexual maturity taking approximately 4.5 years (Nussbaum and Tait 1977, Tait and Diller 2006). Time from oviposition to hatching is roughly 8 months (Karraker 1999), with time from oviposition to absorption of yolk probably closer to a year (Tait and Diller 2006). Peak oviposition is in August and September in California, with peak hatching occurring in the spring (Humboldt County; Tait and Diller 2006). Larval development from hatching to metamorphosis takes 2–2.5 years (Nussbaum and Tait 1977, Tait and Diller 2006). After metamorphosis, an additional 1–1.5 years of growth is required before sexual maturity is attained (Nussbaum and Tait 1977, Tait and Diller 2006).

The extended reproductive period and overwintering of larvae result in overlapping size cohorts in streams (Welsh and Lind 1992, Tait and Diller 2006). Hatchlings are 14–16 mm SVL (Tait and Diller 2006), and size at metamorphosis is around 35 mm SVL (Nussbaum and Tait 1977, Good and Wake 1992, Tait and Diller 2006). In Humboldt County, larval growth rates were recorded as 2.3 mm/year in Six Rivers National Forest (Welsh and Lind 1992) and 8.9 mm/year in a more coastal site in the Mad River drainage (Tait and Diller 2006). Larvae and adults weighed more in the spring than fall at one site, suggesting active foraging and growth over the winter months (Welsh and Lind 1992).

Adults are active at air and water temperatures of 5–10°C, lower than those known for any other aquatic salamander (Stebbins and Lowe 1951, Stebbins 1955, Brattstrom 1963). The average critical thermal maximum for adults and larvae are also lower than reported for other salamanders (larvae: 26.7°C; adults:

27.9°C; Bury 2008b). Welsh and Lind (1996) observed signs of stress in adults at 17.2°C. Thermal tolerances of eggs are unknown (Bury 2008b).

Very few data are available on movement or diet in this species. One mark–recapture study at a single headwater stream/seep site in Humboldt County found extremely low levels of movement, with approximately 1 m/year of movement for adults and 2 m/year for larvae on average (Welsh and Lind 1992). However, unrecaptured animals may have moved longer distances (20% of originally marked animals were recaptured). The diet of *Rhyacotriton variegatus* appears to be generalized on aquatic and semiaquatic invertebrates, with amphipods and collembolans the most abundant prey (Bury and Martin 1967).

Habitat Requirements

Rhyacotriton variegatus occurs within a relatively narrow range of ecological conditions that are typical of late-seral forests. These conditions include cold, clear, flowing permanent seeps and headwater to low-order streams with coarse, rocky substrates in mesic to moist forests (Welsh and Lind 1988, Welsh 1990, Welsh and Lind 1991, Welsh and Lind 1996, Vesely and McComb 2002, Welsh et al. 2005, Ashton et al. 2006, Welsh and Hodgson 2011). Key habitat requirements are the maintenance of cold water temperatures (6.5–15°C) and presence of loose substrates composed of gravel and cobble (Diller and Wallace 1996, Welsh and Lind 1996, Stoddard and Hayes 2005, Welsh et al. 2005, Bury 2008b, Welsh and Hodgson 2008). In the Mattole Watershed, *R. variegatus* occurred primarily in undisturbed headwater channels and was never detected in streams where canopy closure was less than 91% or water temperatures were warmer than 13.5°C (Welsh and Hodgson 2011). *Rhyacotriton variegatus* is extremely desiccation intolerant (Ray 1958), although it will occasionally venture away from the stream channel and use riparian and forest habitat in the wet season (Vesely and McComb 2002; Vesely and McComb, pers. obs., in Welsh and Lind 1996).

Rhyacotriton variegatus is sensitive to fine sediment load and embeddedness (Welsh and Lind 1996, Welsh and Ollivier 1998) and has been found to be positively associated with high-gradient streams, particularly in areas with timber harvesting. This may be due to stream network processes that flush fine sediments out of high-gradient reaches (Corn and Bury 1989, Diller and Wallace 1996, Stoddard and Hayes 2005, Ashton et al. 2006). In a review of seven studies of *R. variegatus* habitat associations, Welsh and Hodgson (2008) found that the species occurred at sites where fine sediment ranged from 2% to 40%, and zero detections occurred when more than 65% of the coarse substrate was embedded with fine sediment.

Distribution (Past and Present)

Rhyacotriton variegatus occurs patchily at elevations below 1469 m throughout the Pacific Coast Ranges of Oregon and California, from the Little Nestucca River and Grande Ronde Valley in Oregon to near Alder Creek in Mendocino County in California (Good and Wake 1992). Populations also occur in the Cascade Range in Oregon (Good and Wake 1992, Miller et al. 2006). A previously reported disjunct population in the McCloud River, Siskiyou County, appears to be based on incorrectly identified museum specimens of the southern long-toed salamander (*Ambystoma macrodactylum sigillatum*) in the California State University, Chico collection.

Suitable microhabitat is patchily distributed in California, and *R. variegatus* is only found in suitable sites about half of the time. Random stratified sampling of 117 sites throughout the geographic range in California found that 45% of sites contained suitable microhabitat, but only 62% of those sites were occupied (Welsh and Lind 1992). Sampling of 38 different sites in the same region selected for the US Forest Service "Old-growth Wildlife Project" found suitable microhabitat in 79% of sites, with *R. variegatus* present in 47% of suitable sites (Welsh and Lind 1992). Systematic stratified sampling of 53 mixed conifer–hardwood stands on public lands in northern California found *R. variegatus* at 62% of sites (Welsh and Lind 1996).

Some of the variation in distribution can be explained by forest age and timber harvest histories, with *R. variegatus* more often found in older, unharvested stands. Welsh (1990) surveyed spring and seep habitats in 34 forest stands in the Coast Ranges in California and southern Oregon ranging from 30 to 560 years old and at elevations of 150–1500 m. *Rhyacotriton variegatus* was found in 70% of old-growth stands, 50% of mature stands, and 11% of young stands. Recent surveys of the Mattole Watershed in northern California (Humboldt and Mendocino Counties) found *R. variegatus* mostly in late-seral headwater tributaries, habitats that are now rare in the watershed (Welsh et al. 2005, Welsh and Hodgson 2011). However, occupancy rates were higher in young forests along the coast where temperatures are mediated by the maritime climate: *R. variegatus* was found in 48% of 30 m sampling reaches and 80% of entire stream reaches in stands less than 80 years old (Diller and Wallace 1996).

Exact figures are difficult to come by, but most of the historical coastal old-growth habitat in California is now gone (85–96.5% gone; references in USFWS 1997). In addition to habitat modification, several investigators have hypothesized that *Dicamptodon* predation may restrict *Rhyacotriton* distribution to small headwater streams (e.g., Stebbins 1955, Nussbaum 1969, Welsh and Lind 1996, Welsh and Ollivier 1998). However, Rundio and Olson (2001) found that *R. variegatus* larvae were unpalatable to *D. tenebrosus* larvae, surviving 90% of encounters in experimental trials.

Trends in Abundance

Estimates of abundance are not available for time periods before timber harvesting became a prominent factor in landscape management, but the highest documented abundances over the last several decades have been in late-seral sites, supporting the idea that abundances are reduced in response to disturbances such as timber harvest and road building. *Rhyacotriton variegatus* can be locally abundant, with densities of up to 22 salamanders/m^2 recorded in suitable streamside habitat at an old-growth site in Six Rivers National Forest, Humboldt County (Welsh and Lind 1992). However, most sites in that study yielded 1–5 captures/10 m^2 (Welsh and Lind 1992). By sampling across the range of *R. variegatus* in California and across stands of different ages, Welsh and Lind (1996) documented a much lower mean density of 0.68 salamanders/m^2. In young stands in coastal northern California (<80 years old), Diller and Wallace (1996) found that densities were 0.18–5.5 salamanders/m^2. Welsh et al. (2000) reanalyzed Welsh and Ollivier's (1998) data from sites in Prairie Creek Redwoods State Park in Humboldt County for comparison to encounter rate data reported by Wroble and Waters (1989) from timber company lands in the same county. *Rhyacotriton variegatus* was found at the rate of 0.72 salamanders/hour on parkland compared to 0.05 salamanders/hour on harvested lands (Welsh et al. 2000). In Oregon, densities averaged 0.29 salamanders/m^2 on forested lands versus 0.04 salamanders/m^2 on logged habitat (Corn and Bury 1989).

Nature and Degree of Threat

Major threats to this species include timber harvesting, road building, rural development, marijuana cultivation, and climate change. *Rhyacotriton variegatus* is sensitive to the impacts of timber harvesting and roadbuilding due to direct impacts of heavy equipment and indirect effects on temperature, humidity, and sediment load (Welsh et al. 2000, Welsh and Hodgson 2008). Several researchers have argued that declines and extirpations will continue due to timber harvesting and related land management practices (e.g., Welsh et al. 2000, Ashton et al. 2006, Olson et al. 2007, Welsh and Hodgson 2008). While *R. variegatus* can persist in some harvested areas, particularly in coastal forests where the effects of logging may be ameliorated by the milder climate (e.g., Welsh 1990, Diller and Wallace 1996; S. Barry, unpublished data), it occurs in more sites and with higher density in older stands.

Habitat loss and degradation due to rural residential development and marijuana cultivation is a growing concern for this species in California. Every new house built in forested lands requires a source of water, which is often provided by diverting headwater streams. In some cases, *R. variegatus* has been observed to occur above but not below such diversions (M. van Hattem, pers. comm.). This threat is likely to increase in the near future. For example, the Humboldt County General Plan is currently being updated, with some proposals considering a doubling or tripling of rural development. Marijuana cultivation also presents a water diversion threat to this species, as well as potential negative impacts due to grading, roadbuilding, and the application of herbicides and pesticides (e.g., Thompson et al. 2014).

Rhyacotriton variegatus has slow developmental times and low vagility, leading to potentially high susceptibility to rapidly changing environmental conditions. Expected climate changes within its range over the next 100 years include increased temperatures, changes in hydrology, changes in fire regime, and vegetation shifts. Mean annual temperatures are expected to increase throughout the range of *R. variegatus* in California (reviewed in PRBO 2011). The frequency of extremely hot days is projected to increase, with roughly 9 additional days over 32.2°C (Bell et al. 2004). Such temperatures exceed the critical thermal maxima for adults and larvae of *R. variegatus*, although water temperatures, microhabitat structure, and behavioral thermoregulation may ameliorate these effects. For coastal populations, upwelling is expected to intensify, which may increase fog development and contribute to cooler, moister conditions (Snyder et al. 2003, Lebassi et al. 2009). Coastal populations may therefore continue to provide more favorable climatic conditions than areas farther inland. Potential changes in precipitation are less clear, with some models predicting modest increases, others modest decreases, and others reductions in rainfall of up to 28% (reviewed in PRBO 2011). Warmer temperatures will result in less precipitation stored as snow, and reductions of 30–80% are predicted for snowpack accumulation in northwestern California (Snyder et al. 2004, Cayan et al. 2008b). The timing of spring snowmelt has shifted later in the spring in this region over the last 50 years (Stewart et al. 2005), though the timing of future shifts is unknown. Reductions in water availability due to reduced snowpack and possibly reduced precipitation will affect the timing and magnitude of stream flows. This may negatively affect habitat quality and availability for all life stages of this highly aquatic salamander. How fire regime will be affected by climate change in northwestern California is not well understood. Some models predict little change in fire regime or even decreases in area burned along the northern coast (Fried et al. 2004, Lenihan et al. 2008), while increases in area burned have been predicted for the southern coast of northwestern California (Lenihan et al. 2008). Westerling et al. (2011) projected a 100% increase in area burned in northwestern California under some scenarios. How fire affects *R. variegatus* needs further study, although direct mortality and habitat degradation due to fire have been documented in other stream-breeding amphibians (e.g., Gamradt and Kats 1997, Pilliod et al. 2003). Vegetation communities are expected to shift from moist conifer to drier mixed evergreen forest, with reductions in Douglas fir and redwood forest in particular (Lenihan et al. 2008, PRBO 2011). It is unclear what effect these shifts may have on *R. variegatus* because stream conditions and forest age seem to be more important indicators of habitat quality than forest type.

Status Determination

Rhyacotriton variegatus is a Priority 1 Species of Special Concern due to its high degree of habitat specificity resulting in a patchy distribution in isolated habitat islands, high degree of genetic variation among management units, and association with late-seral forests that are now rare and often ecologically compromised by timber harvesting (Good and Wake 1992, Welsh and Lind 1996).

Management Recommendations

Rhyacotriton variegatus populations would benefit from forest management activities that maintain cold water temperatures and low sedimentation levels such as decreasing the use and building of roads, decreasing timber harvest, and leaving riparian vegetation intact in harvested areas. Suitable microhabitats should be surveyed for *R. variegatus* presence during the wet season when salamanders are more likely to be detected before such areas are disturbed (Tait and Diller 2006, Olson et al. 2007). Monitoring activities themselves can damage sensitive microhabitats (L. Diller, pers. comm.), and personnel should be well trained in techniques to minimize such negative effects. Occupied microhabitats in particular should be protected from direct impacts of heavy equipment. In areas where timber harvest occurs, vegetation should be left intact around *R. variegatus* habitat, particularly to maintain canopy cover, though the width and configuration of such buffers is an important research need detailed below. In the absence of more detailed research, Olson et al. (2007) recommend using relatively wide buffers on the order of 40–150 m to maintain obligate riparian species. In addition to buffers along streams, habitat should be left intact around seeps ("leave islands"; reviewed in Olson et al. 2007). Marijuana cultivation appears to pose a growing threat to maintenance of high-quality habitat for this species. Enforcement and regulation of marijuana cultivation is an ongoing issue in California and we suggest that the environmental impact of such activities be considered. Little is known about use of upland habitats, but protection of large channel networks and associated seeps and springs to maintain aquatic and upland connectivity would likely help maintain populations of *R. variegatus* (Welsh and Lind 1992, Vesely and McComb 2002, Olson et al. 2007, Welsh 2011).

Monitoring, Research, and Survey Needs

Several studies have been conducted to determine the presence/absence of *Rhyacotriton variegatus* across the landscape, and such surveys should continue. A critical research need is studies that monitor population abundance over time, particularly under different timber harvesting regimes. Given the long life span and slow development time of this species, such long-term studies might provide insights that shorter, single-season analyses would miss. When possible, population estimates in managed forests should be compared to *R. variegatus* abundance in nearby undisturbed mature forest stands (i.e., reference populations) to assess the impacts of disturbance (Welsh 2011). Additional studies on movement ecology and dispersal beyond localized movements would aid in designing management strategies to promote habitat connectivity. The extent to which upland versus aquatic habitats are used for dispersal is unknown and is crucial for determining whether buffers should be focused around continuous waterways, upland linkages between waterways, or both (Welsh and Lind 1992, Olson et al. 2007, Welsh 2011).

Experiments that test the efficacy of buffer strips for maintaining favorable habitat conditions in harvested areas would also be valuable. Buffer strips from 6 to over 90 m wide have been proposed for maintaining riparian fauna under a range of management scenarios (reviewed in Olson et al. 2007). Stoddard and Hayes (2005) recommended buffer strips >46 m wide for *Rhyacotriton*. Similarly, riparian buffer strips 40 m wide around first through third-order streams in Oregon supported similar salamander abundance (including *R. variegatus*) as unharvested stands (Vesely and McComb 2002). Welsh and Hodgson (2008) recommend stream temperatures <15°C to maintain populations. The relationship between the size and aspect of a subbasin, the amount of the surrounding area harvested, the resulting maximum stream temperature, and how much buffer would be required to ameliorate any critical biological temperature thresholds are important research needs (Welsh et al. 2005). Temperature is not the only factor that can be influenced by management activities however, and other indicators of habitat quality such as embeddedness should be measured as well (Olson et al. 2007).

Because *R. variegatus* is patchily distributed, monitoring studies should first identify areas with suitable habitat. In surveys for *R. variegatus* in Douglas fir/hardwood forests in the Klamath region, Welsh and Lind (1992, 1996) defined minimum essential microhabitat for *R. variegatus* as an area of at least 10 m^2 of flowing water (e.g., a patch of spring seep or first- or second-order streams) at least 75 m away from a forest edge. Within these sites, aquatic searches seemed most effective at detecting *R. variegatus*, as they are rarely encountered using techniques such as terrestrial pitfall trapping (e.g., Welsh 1990). Sampling should be done in the spring when *R. variegatus* are most abundant (Welsh and Lind 1992, Ashton et al. 2006, Tait and Diller 2006).

Landscape genetic studies that quantify levels of connectivity within and across stream systems would help to better delimit local management units as well as important dispersal corridors for this species. Studies similar to recent analyses on another western stream salamander, the Idaho giant salamander (*Dicamptodon aterrimus*) (Mullen et al. 2010), would be particularly instructive as a way to examine the relationship between stream connectivity and salamander gene flow.

RED-BELLIED NEWT

Taricha rivularis (Twitty 1935)

Status Summary

Taricha rivularis is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 81% (69/85). During the previous evaluation, *T. rivularis* was determined to not merit Species of Special Concern status (Jennings and Hayes 1994a). *Taricha rivularis* ranked high enough to warrant status in the current evaluation, although very little information is available on population distribution or abundance trends.

Identification

All species in the genus *Taricha* are stocky, medium-to-large newts with granular skin, dark dorsal coloration, and indistinct or absent costal grooves (Petranka 1998, Stebbins 2003). *Taricha rivularis* has bright, tomato red ventral coloration and reaches up to 8 cm SVL (Stebbins 2003). In all members of the genus *Taricha*, breeding males seasonally acquire smooth skin and an enlarged tail fin (Petranka 1998). Larvae have a stream-type-like morphology where the tail fin does not extend all the way to the shoulders (Stebbins 2003). The range of *T. rivularis* overlaps with the range of the rough-skinned newt (*T. granulosa*), and the

Red-Bellied Newt: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	10
v. Ecological tolerance (10)	10
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	69
Total Possible	85
Total Score/Total Possible	.81

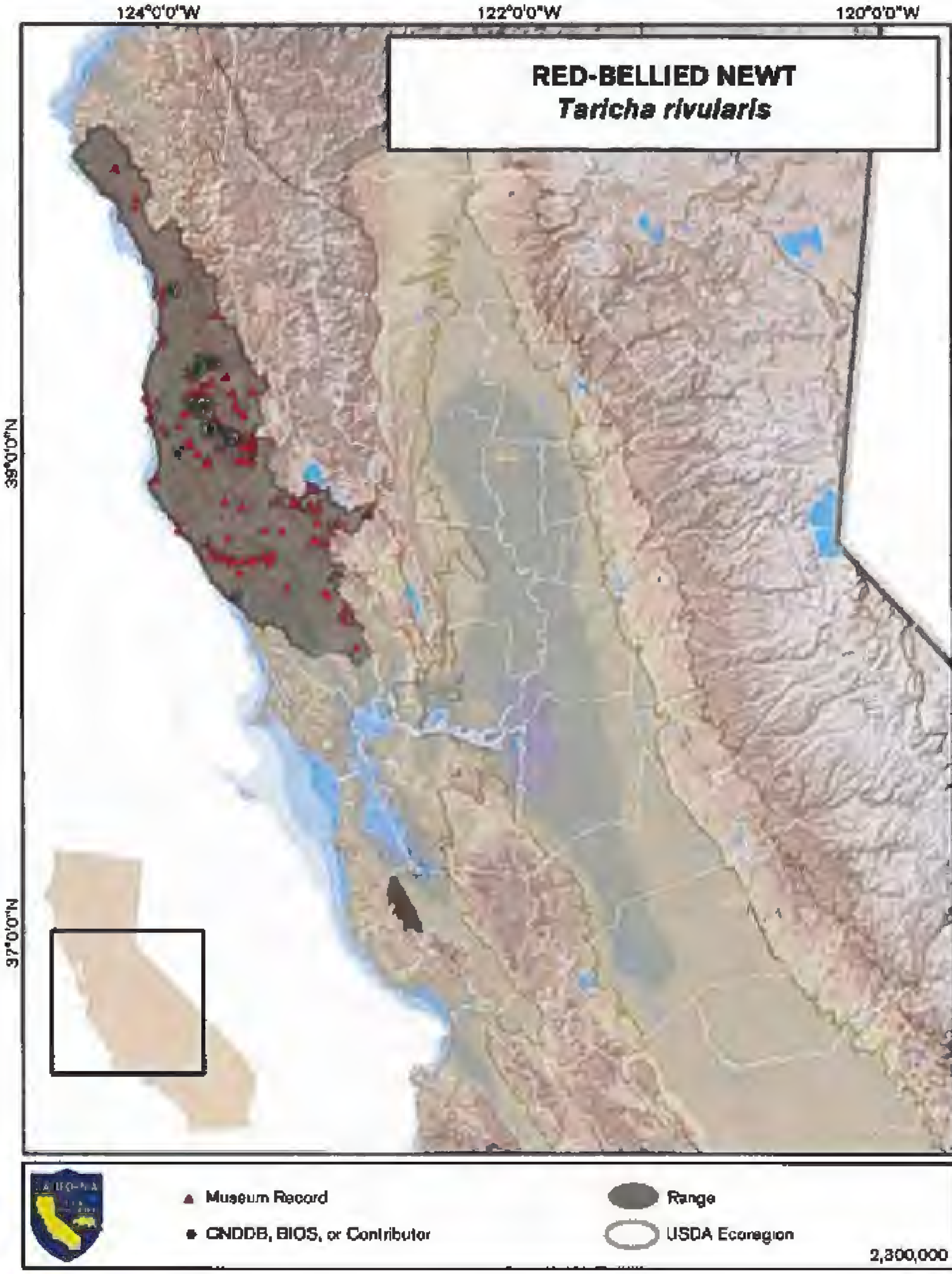


PHOTO ON PREVIOUS PAGE: Red-bellied newt, Mendocino County, California. Courtesy of Adam Clause.

southeastern edge of its range overlaps with the Coast Range newt (*T. torosa*). These species can be distinguished based on several morphological and color characteristics. In addition to distinctive red ventral coloration, *T. rivularis* has dark brown eyes, compared to the yellow or silvery irises in the other species (Twitty 1935).

Taxonomic Relationships

Taricha rivularis was described on the basis of the clear morphological differences existing between it and other California *Taricha* (Twitty 1935), and its species status has never been questioned. Gene flow among populations was previously thought to be very low because animals return to the same stream areas for breeding and show very strong homing behavior (Hedgecock and Ayala 1974, Hedgecock 1978; see the "Life History" section). Kuchta and Tan (2006a) found low levels of allozyme and mitochondrial DNA divergence among four populations in the north and south of the range, which may suggest that gene flow is higher than previously thought. Although *T. rivularis* shows a high degree of philopatry, long-distance movements are well documented, and this may explain the observed low levels of divergence (Kuchta and Tan 2006a).

Life History

Breeding coincides with the receding of streams after heavy winter rains (Twitty 1942). Adults are terrestrial, and the aquatic breeding phase lasts from February to May, with most breeding occurring between March and early April (Twitty 1955, Packer 1960, Twitty 1966, Stebbins 1985). Males typically breed annually, whereas most females breed every 2–3 years (Twitty 1961, Twitty et al. 1964). Adults have been observed returning to the same ~15 m segment of creek to breed across multiple years (Twitty 1959, Packer 1962, Packer 1963, Twitty et al. 1967a). Adults tend to use a small reach of stream during the breeding season, although movements of a couple hundred meters within a season have been observed (Packer 1962). Adults are also capable of moving several kilometers across years and have excellent homing abilities (Twitty 1959, Packer 1962, Twitty et al. 1964, Twitty et al. 1967a). After breeding, adults leave streams but usually remain in the same drainage (Twitty et al. 1967b). Fall rainfall triggers movement, but heavy rainfall can inhibit overland movement (Packer 1960, Grant et al. 1968), and sustained rainfall, increased stream volume, or increased sediment load can stimulate animals to temporarily leave breeding streams (Packer 1960). Little is known about terrestrial habitat use by metamorphs. Underground retreats are used from May to October, and adults forage on the surface before and as they migrate to streams (Twitty 1966, Licht and Brown 1967, Marks and Doyle 2005).

Eggs are attached in a single layer to the bottom of stones or submerged vegetation in fast-flowing water (Twitty 1935, Twitty 1942). The average size of an egg mass is 10 eggs (range 6–16) (Twitty 1935, Riemer 1958, Twitty 1964), and as many as 70 egg masses have been observed attached to a single stone (Twitty 1935, Twitty 1942). The incubation period in the lab is 16–34 days, with faster development times at warmer temperatures (Licht and Brown 1967). Larvae hatch at a minimum of 10 mm TL (Riemer 1958, Twitty 1964) in mid to late April and metamorphose in late August (Licht and Brown 1967) at 45–55 mm TL (Stebbins 1951). There is no evidence that larvae overwinter in streams (Riemer 1958, Twitty 1964). It is unknown how far or to what habitats metamorphs travel, but they go into hiding shortly after metamorphosis (Twitty 1955, Twitty 1961, Twitty 1966, Twitty et al. 1967b). Juveniles are not captured in terrestrial habitats when adults are abundant, suggesting that they remain underground, or at least in a distinct, unknown microhabitat, for several years (Twitty et al. 1967a). It takes approximately 5 years to reach sexual maturity (Licht and Brown 1967). Hedgecock (1978) estimated life spans on the order of 20–30 years based on Twitty's (1966) data, and annual survivorship of adults is probably >90% in most years (Twitty 1961). At one site in Sonoma County, 40% of originally marked adult animals were still being recaptured 11 years later (Twitty 1966).

Insects and other small invertebrates presumably make up the bulk of the diet of larvae and adults. In one study, adult stomach contents contained exclusively terrestrial organisms (mostly insects), and adults apparently do not feed while in the water during the breeding season (Packer 1961, Licht and Brown 1967).

Habitat Requirements

Taricha rivularis is found in redwood forests along the coast, although other forest types such as Douglas fir, tan oak, and madrone are also used (Marks and Doyle 2005). Aquatic breeding habitats are moderate to fast-flowing mountain streams with rocky bottoms (Twitty 1935, Stebbins 1951). In the Mattole Watershed (northern Mendocino and southern Humboldt Counties), *T. rivularis* was reported to use both steep headwater and 2–4% gradient step-pool reaches, but was most abundant in lower-gradient plane-bed channels (Welsh and Hodgson 2011). Other features of occupied stream habitats were water temperatures ranging between 15°C and 26°C, a mix of coarse streambed substrates, and intermediate levels of canopy closure (Welsh and Hodgson 2011). Unlike other members of the genus, *T. rivularis* rarely breed in ponds or other standing water habitats (Riemer 1958, Stebbins 1985) and seem to avoid streams used by *T. torosa* (Twitty 1942, Twitty 1955). *Taricha rivularis* will breed in the same streams as *T. granulosa* but tend to use faster-flowing reaches (Twitty 1942).

Distribution (Past and Present)

Taricha rivularis is endemic to California and has the smallest geographic distribution among its congeners (Stebbins 2003). The species occurs in coastal northern California in Sonoma, Lake, Mendocino, and southern Humboldt Counties, at elevations from 150 to 450 m (Stebbins 2003, Marks and Doyle 2005). An isolated population is known from the Stevens Creek watershed in Santa Clara County, although it is unclear if this is an introduction or a native population (Reilly et al., in press). Some habitat has likely been lost to vineyard and other agricultural development in Sonoma and Mendocino Counties, although systematic surveys are not available (H. Welsh, pers. comm.). Some populations have been lost due to damming of creeks and rivers (e.g., Skaggs Spring, which was inundated during the formation of Lake Sonoma). Data from the Mattole Watershed in the mid-1990s documented *T. rivularis* presence in 35% of sampled streams (Welsh et al. 2005), with *T. rivularis* restricted to the forested southern portions of the watershed (Welsh and Hodgson 2011).

Trends in Abundance

Few abundance data are available for this species. Hedgecock (1978) used Twitty's (1961, 1966) census data to estimate that ~60,000 breeding adults occurred along a ~2.5 km stretch of creek in Sonoma County. In the Mattole Watershed, 300 m stretches of randomly selected stream reaches ($n = 83$ stream reaches) yielded 24 metamorphs and 104 aquatic larvae (Welsh and Hodgson 2011).

Nature and Degree of Threat

The paucity of distribution and abundance data makes it difficult to determine the status of most *Taricha rivularis* populations. However, the species has a small range in an area that has experienced high levels of habitat conversion to vineyards and subdivisions, rendering them vulnerable to habitat loss and fragmentation (Marks and Doyle 2005). *Taricha rivularis* may also be experiencing increasing mortality from vehicular traffic (Marks and Doyle 2005), especially during breeding migrations.

Climate change poses potential risks to *T. rivularis* through increased temperatures, changes in hydrology, changes in fire regime, and vegetation shifts. Mean annual temperatures are expected to increase throughout northwestern California (reviewed in PRBO 2011); however, maximum temperature tolerances of *T. rivularis* are unknown. *Taricha rivularis* populations on the coast may be less affected by temperature increases because upwelling is expected to intensify, potentially leading to increased fog development and cooler, moister conditions (Snyder et al. 2003, Lebassi et al. 2009). Potential changes in precipitation are less clear, with some models predicting little change and others reductions in rainfall of up to 28% (reviewed in PRBO 2011). If conditions become warmer and drier, especially in inland sites, this may restrict terrestrial habitat use and overland dispersal. Changes in precipitation may affect stream hydrology, although how *T. rivularis* will respond to such changes is unknown. How fire regime will be affected by climate change in northwestern California is not well understood. Some models predict little change in fire regime or even decreases in area burned along the northern coast (Fried et al. 2004, Lenihan et al. 2008). Increases in area burned have been predicted for the southern coast of northwestern California and inland areas (Lenihan et al. 2008). Westerling et al. (2011) projected a 100% increase in area burned in northwestern California under some scenarios. How fire impacts *T. rivularis* needs more study, although direct mortality and habitat degradation due to fire has been documented in other stream-breeding amphibians in similar habitats (e.g., Gamradt and Kats 1997, Pilliod et al. 2003). Vegetation communities are expected to shift from moist conifer to drier mixed evergreen forest, with reductions in Douglas fir and redwood forest in particular (Lenihan et al. 2008, PRBO 2011). *Taricha rivularis* may not be severely negatively affected by such shifts, as they use multiple forest types.

Status Determination

Taricha rivularis has a small range in an area that has experienced increased levels of habitat loss and fragmentation in recent decades, resulting in a Priority 2 Species of Special Concern status for this endemic salamander.

Management Recommendations

Given the limited ecological information on this species outside of a handful of sites, it is difficult to make management recommendations other than protecting known breeding habitats. Disturbances such as timber harvest, roadbuilding and use, housing development, agricultural development, and water diversions should be minimized or eliminated in *Taricha rivularis* habitat. Occupied habitat should be protected, with a focus on protecting the entire stream network (Olson et al. 2007, Welsh 2011). Retaining streamside buffers on managed lands can help mitigate the effects of logging and roadbuilding, but more research is needed to determine buffer prescriptions, particularly how to protect stream network processes (Olson et al. 2007). The ecological effects of buffer protections may vary across habitat types, and narrower buffers may be effective in more mesic coastal habitat compared to more xeric inland sites. One model recommends riparian management zones 40–150 m wide and patch reserves along headwater streams to accommodate upland habitat use and promote connectivity among drainages (Olson et al. 2007). Given the long-range movements documented in this species, large terrestrial habitat patches may be necessary to maintain connectivity among populations. Any efforts to translocate individuals should also take the strong evidence for adult homing behavior into account, as animals are likely to try and return to their original streams. Construction of new roads should be minimized or avoided in areas where protecting *T. rivularis* is a high conservation priority. To reduce the sedimentation impacts of runoff from roads, forest roads should be disconnected from stream systems (e.g., through the use of ditch-relief culverts). Use of heavy equipment should be avoided or restricted on forest roads when larvae are present in nearby aquatic habitat. Road management strategies should be applied to all forest roads, not just those used for timber harvest. In areas that are known to suffer high road mortality, migration barriers and under-road tunnels may reduce vehicular death (e.g., see review in Schmidt and Zumbach 2008), although research is needed into the design and efficacy of such interventions.

Monitoring, Research, and Survey Needs

Surveys to determine the current distribution of occupied breeding habitats are a first step to documenting potential extirpations. Resurveys of Twitty's field sites along Pepperwood Creek, a tributary along the Wheatfield Fork of the Gualala River in northwestern Sonoma County, would be useful for assessing whether population abundance has changed, as this is one of the few areas where demographic data have been collected (e.g., Twitty 1961, Twitty 1966). However, locating the original sites has proven difficult, and they may occur on private lands that are largely inaccessible (S. Kuchta, pers. comm.). Basic ecological research into habitat preferences (both terrestrial and aquatic) are needed as well as demographic data on all life stages (Petranka 1998, Marks and Doyle 2005). Additional research is needed on dispersal, using both field and genetic techniques. Experiments that moved individuals to different streams found that animals traveled overland to return to their native streams, moving as much as 8 km through terrestrial habitat (Twitty 1959, Twitty et al. 1966). If such terrestrial movements are typical of naturally dispersing animals, then large patches of terrestrial habitat will be needed to maintain connectivity among populations. Finally, although it is assumed that introduced trout and bullfrogs are not a threat to *Taricha* due to their toxic skin secretions, this should be examined for eggs, larvae, and breeding adults. In other California newts, recent experimental research has shown that larval *T. torosa* are highly susceptible to predation by *Ambystoma* (Ryan et al. 2009), and tetrodotoxins have not been isolated from larvae or eggs of *T. granulosa* (Fuhrman 1967).

COAST RANGE NEWT, SOUTHERN POPULATIONS

Taricha torosa (Rathke 1833)

Status Summary

Populations of *Taricha torosa* from the Salinas River in Monterey County south constitute a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 66% (73/110). During the previous evaluation, these populations were also considered Species of Special Concern (Jennings and Hayes 1994a).

Identification

Taricha are stocky, medium-to-large newts (up to 8 cm SVL) with granular skin, indistinct or absent costal grooves, and dark dorsal coloration (Petranka 1998, Stebbins 2003). *Taricha torosa* has yellowish brown to dark brown dorsal coloration and pale yellow to orange ventral coloration (Petranka 1998). Adults that enter aquatic habitats for breeding develop smooth skin and a flattened tail while they are in the aquatic habitat, and the tail fin becomes enlarged in males (Stebbins 2003). Larvae are pond type, with large gill filaments and a large fin, and have two dark, irregular longitudinal stripes running down the back (Stebbins 2003).

Taricha torosa is the only newt in southern California but may be confused with other *Taricha* species in northern California, and with the Sierra newt (*T. sierrae*), where the two overlap in Tulare County. All of the characters for distinguishing among *Taricha* can be

Coast Range Newt, Southern Populations: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	10
v. Ecological tolerance (10)	3
vi. Population trend (25)	15
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	3
Total Score	73
Total Possible	110
Total Score/Total Possible	0.66

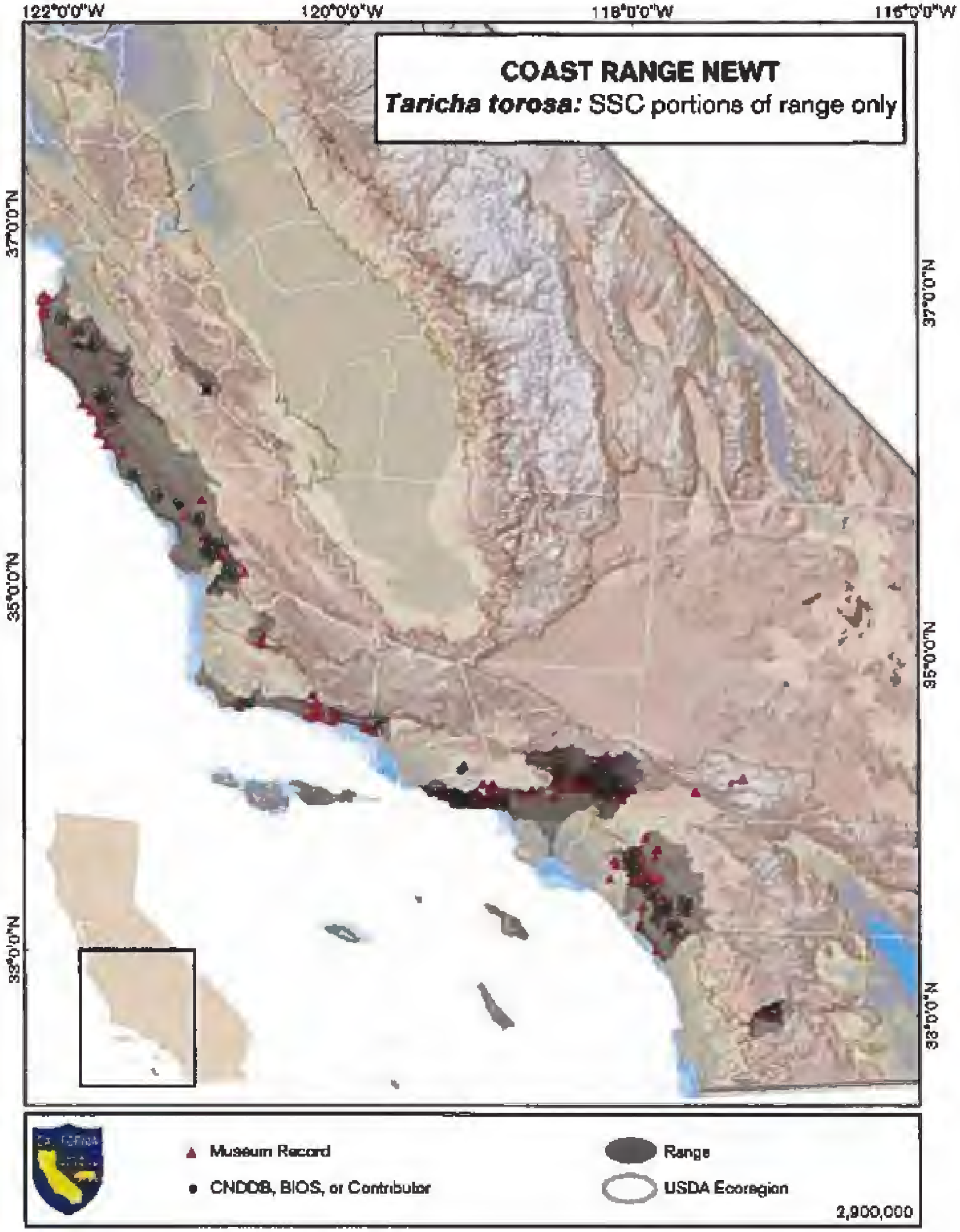


PHOTO ON PREVIOUS PAGE: Coast range newt, southern populations, Los Angeles County, California. Courtesy of Adam Clause.

variable, and in some individuals differentiating the species can be difficult. *Taricha torosa* resembles *T. granulosa* but can be distinguished based on the extensive light ventral coloration that reaches the underside of the eyes, eyes that extend beyond the margin of the head when viewed from above, and palatal teeth in the roof of the mouth forming a Y shape (Stebbins 2003). In *T. granulosa*, the dark dorsal coloration extends beneath the eyes, the eyes are more closely inset and do not extend to the margin of the head when viewed from above, and the teeth in the roof of the mouth are in a V-shaped configuration (Stebbins 2003). *Taricha rivularis* has dark eyes (*T. torosa* has yellow in the eyes), a tomato red venter, and dark coloration under the limbs and over the cloaca (Stebbins 2003). *Taricha sierrae* tends to be darker brown dorsally than *T. torosa* and has a burnt or reddish ventral coloration (Stebbins 2003). *Taricha sierrae* also has more of the lighter ventral coloration on its snout and upper eyelids than *T. torosa* (Twitty 1942, Riemer 1958), and these differences in color pattern are intermediate in hybrid populations (Kuchta 2007).

Taxonomic Relationships

Previously, two allopatric subspecies were recognized: *Taricha torosa sierrae* in the Sierra Nevada and *T. t. torosa* in the Coast Range (Riemer 1958). Phylogeographic work has shown that populations in the southern Sierra are *T. t. torosa* (Tan and Wake 1995), and further molecular work has supported elevation to species status for both subspecies (Kuchta and Tan 2006b, Kuchta 2007). There is a contact zone between the two species around the Kaweah River in Tulare County. Kuchta and Tan (2006b) concluded that while newts from San Diego County do not show long-term evolutionary independence, they still constitute a conservation unit due to genetic differentiation, demographic independence, and geographic isolation.

Life History

Terrestrial adults migrate to aquatic breeding habitats such as ponds, streams, and reservoirs from December to early May, and timing varies by locality, weather, and habitat conditions (Storer 1925, Twitty 1942, Riemer 1958, Gamradt and Kats 1997). Southern populations migrate in March and April (Storer 1925, Brame 1968, Kats et al. 1992) and tend to breed in quiet stream pools (Gamradt and Kats 1996, Gamradt and Kats 1997). No other stream-breeding salamanders occur in the southern part of the range of *Taricha torosa*. Eggs are attached under rocks or to vegetation, with egg masses ranging in size from 7 to 47 eggs (Ritter 1897, Storer 1925, Twitty 1942, Brame 1956, Brame 1968, Mosher et al. 1964). Females may lay 3–6 egg masses at a time, but it is unknown if they breed every year or skip years like *T. rivularis* (Ritter 1897, Twitty 1961, Twitty et al. 1964, Brame 1968). Adults typically leave breeding habitats in early to midsummer (Kats et al. 1994).

Eggs hatch after 4–6 weeks (Kats et al. 1994), and larvae develop for several months, typically metamorphosing in summer or fall (Kuchta 2005). Overwintering has been documented in larvae from Los Angeles (Storer 1925) and Riverside (Carroll et al. 2005) Counties, but given a lack of other reports, this behavior is likely uncommon (Kuchta 2005). Average size at metamorphosis for a Berkeley, Alameda County, population was 47 mm TL, although this probably varies widely depending on local conditions (Ritter 1897). Larvae from a vernal pool in Sonoma County metamorphosed in late July and early August at an average size of 43.8 mm TL (Kuchta 2005). Metamorphosis in permanent water habitats, as are commonly used in the southern part of the range, has not been studied.

Taricha torosa appears to show similar breeding site fidelity, homing ability, and longevity as other *Taricha*, although relatively fewer data are available from *T. torosa*. Watters and Kats (2006) PIT-tagged 36 breeding adults in the Santa Monica Mountains in Los Angeles County in the early 1990s, and recaptured animals for several years. Thirty-nine percent of animals originally tagged were recaptured in subsequent years, some as long as 11 years later, yielding minimum age estimates of 12–14 years. Animals were recaptured on average 15.5 m from the original capture locality. Terrestrial habitat use is poorly studied in juveniles and adults, although overland movements can be substantial. Trenham (1998) recaptured juveniles up to 3.5 km from their natal ponds. Once adults leave breeding sites, they use mesic microhabitats for aestivation during the dry summer (Stebbins 1951, Trenham 1998).

Larvae presumably eat small invertebrates, detritus, and possibly cannibalize conspecifics (Ritter 1897, Kuchta 2005). Aquatic adults will cannibalize eggs and larvae (Ritter 1897, Kats et al. 1992, Hanson et al. 1994). Terrestrial adults are generalist predators consuming a variety of invertebrate prey and the occasional small vertebrate (Ritter 1897, Hanson et al. 1994, Kerby and Kats 1998).

Habitat Requirements

Northern populations occur in mesic forests in hilly or mountainous terrain, while southern populations occur in drier habitats such as oak, chaparral, and grassland (Riemer 1958). Southern populations tend to use permanent streams for breeding, though recruitment may be higher in seasonal reaches that are free of nonnative predatory fish (E. Ervin, pers. comm.). *Taricha torosa* in southern California are also limited by the availability of rocky canyons with clear, cold water (S. Barry, pers. comm.; R. Fisher, pers. comm.). In the Santa Monica Mountains in Los Angeles County, *T. torosa* using a perennial stream laid 89% of their egg masses in pools and 9.5% in runs (Gamradt and Kats 1997). Riffles were rarely used for oviposition (Gamradt and Kats 1997).

Distribution (Past and Present)

Taricha torosa ranges from central Mendocino County south through the Coast Ranges to San Diego County, and also occurs in the southern Sierra Nevada north to Tulare County, from sea level to 1280 m (Stebbins 1959, Tan and Wake 1995). Species of Special Concern status extends only to those populations found in Monterey County and farther south, excluding the southern Sierra Nevada isolate. Our map only shows these populations, though we note that it includes museum specimens from the San Bernardino Mountains that have been questioned (E. Ervin, pers. comm.). *Taricha torosa* is restricted to the Santa Ynez Mountains in Santa Barbara County (S. Sweet, pers. comm.). The southernmost populations of *T. torosa* are highly fragmented and occur in the Santa Monica, San Gabriel, and Santa Ana Mountains (Stebbins 2003). Within San Diego County, populations farthest south are geographically isolated from the rest of the range. Jennings and Hayes (1994a) reported these populations as extirpated; however, since then San Diego populations in the Cuyamaca Mountains have been reported to persist in small isolated pockets of 15–20 breeding adults in the Boulder, Ceder, and Conejos Creek systems (E. Ervin, pers. comm. in Kuchta 2005). Surveys in the 1990s of the foothills and mountains around the Central Valley found *Taricha* species (*T. torosa* and *T. granulosa*) absent from more than half of historically occupied counties (Fisher and Shaffer 1996). Jennings and Hayes (1994a) estimated that a third of localities in southern California have been extirpated. Surveys from 2000 to 2002 in the Santa Monica Mountains and Simi Hills in southern California found *T. torosa* present in 43% (15/35) of streams (Riley et al. 2005). *Taricha torosa* tended to be absent from urban streams, and Riley et al. (2005) hypothesized that this was due to effects on habitat quality from artificial flow regimes, increased presence of introduced species, and possibly also collection pressure.

Trends in Abundance

Historically, *Taricha torosa* was noted as common along the Pacific slope (Klauber 1928, Bogert 1930, Klauber 1930, Dixon 1967, Brattstrom 1988), and it may have been one of the most abundant amphibians in California (Jennings and Hayes 1994a). Populations in the upper Carmel Valley adjacent to the Hastings Reservation in Monterey County numbered in the thousands in the early 1990s but have not been systematically resampled more recently (B. Shaffer and W. Koenig, unpublished data). Southern populations in the Santa Ynez Mountains of Santa Barbara County may have always been small (Jennings and Hayes 1994a). Population size estimates are not available, but populations in the south that used to be in the hundreds are now in the tens (R. Fisher, pers. comm.; E. Ervin, pers. comm., in Kuchta 2005), with populations in San Diego County potentially on the brink of extirpation (S. Kuchta, pers. comm.).

Nature and Degree of Threat

Major threats to *Taricha torosa* include habitat loss and degradation, wildfire, introduced species, and vehicular traffic (Jennings and Hayes 1994a). Sedimentation has caused a large amount of habitat degradation, especially in Los Angeles, Orange, Riverside, and San Diego Counties (Jennings and Hayes 1994a), and *T. torosa* is absent from previously occupied streams in heavily urbanized watersheds (Riley et al. 2005). Wildfire also contributes to habitat degradation. Surveys before and after a chaparral wildfire along a perennial Santa Monica Mountain stream in Los Angeles County documented a roughly 50% reduction in the availability of preferred pool and run habitat due to erosion (Gamradt and Kats 1997). As a result, egg mass density was reduced by two-thirds compared to prefire levels (Gamradt and Kats 1997). Terrestrial adults were observed to produce foamy skin secretions while walking through a prescribed burn area of chamise habitat in Monterey County (Stromberg 1997).

Negative effects of introduced predators on *T. torosa* have been documented. In the Santa Monica Mountains in Los Angeles County, introduced crayfish (*Procambarus clarkii*) and mosquitofish (*Gambusia affinis*) are predators on *T. torosa* and may be contributing to declines (Gamradt and Kats 1996). Stream surveys did not detect either invasive species in the 1980s. Resurveys in the 1990s of previously used breeding habitats found no evidence of breeding in streams with crayfish and mosquitofish present. In one case, *T. torosa* recolonized a reach following floods that removed crayfish, supporting the hypothesis that crayfish exclude newts from breeding habitat. In field and lab trials, survivorship of eggs and larvae was less than 30% in the presence of crayfish. Mosquitofish did not affect egg survivorship but did predate heavily on larvae. Only 46% of larvae survived in the presence of mosquitofish (Gamradt and Kats 1996). Crayfish also aggressively attack and chase adult *T. torosa* out of the water (Gamradt et al. 1997). Native California tiger salamanders (*Ambystoma californiense*) will prey on *T. torosa* larvae where the two co-occur around the Central Valley. However, recruitment is even lower in the presence of hybrids between native *A. californiense* and introduced barred tiger salamanders (*Ambystoma tigrinum mavortium*) (Ryan et al. 2009).

Bd has been documented in 7% (6/90) of *T. torosa* sampled from Santa Clara County (Padgett-Flohr and Longcore 2007), but the role of *Bd* in *T. torosa* declines is unknown. The role of UV radiation in declines is also unknown. Anzalone et al. (1998) reared eggs in field enclosures in the Santa Monica Mountains and found that eggs exposed to UV radiation had 40% survivorship compared to 80% survivorship of eggs when UV was shielded out. However, given that eggs are often attached under rocks and to vegetation, UV is unlikely to be responsible for large-scale declines in the field (Palen and Schindler 2010).

Under climate change, mean annual temperatures are projected to increase throughout the southern range of *T. torosa*, with warmer winters and summers and earlier spring warming expected (reviewed in PRBO 2011). There is less certainty about future precipitation patterns, with estimates ranging from little change to roughly 30% decreases in rainfall (Snyder and Sloan 2005, PRBO 2011). Warmer and potentially drier conditions may affect availability of intermittent and ephemeral waterways used for breeding. Snowpack reductions of up to 90% are predicted in southern California

(Snyder et al. 2004), which will likely result in altered flow regimes. How *T. torosa* may respond to these changes is unknown. The probability and extent of large (>200 ha) fires is expected to increase in the northern part of the special concern range (Fried et al. 2004, Westerling and Bryant 2008). Increases and decreases in fire probability and extent have been predicted for southern California. There is little consensus on future fire dynamics in this part of the range because of the difficulty in modeling Santa Ana weather events (Westerling et al. 2004, Westerling and Bryant 2008). Increases in fire are likely to negatively impact *T. torosa*, largely through habitat degradation but possibly also through direct mortality. Predicted vegetation shifts due to climate change include decreases in chaparral, shrubland, and woodland, and increases in grassland area (Lenihan et al. 2008, PRBO 2011). *Taricha torosa* uses all of these habitat types, and the effects of shifts in their relative abundance and distribution are unknown.

Status Determination

Documented extirpations and reductions in density of remaining populations in southern California, combined with occurrence in an area of high human density, result in a Priority 2 designation for southern populations of *Taricha torosa*.

Management Recommendations

Disturbances such as roadbuilding and road use, housing development, and water diversions should be minimized or eliminated in *Taricha torosa* habitat. Known breeding habitat should be a high priority for protection. Upland terrestrial habitat also needs to be protected, though the extent and configuration of upland habitat required to maintain population connectivity needs more study. Measures to prevent invasion or remove existing nonnative predators are high-priority activities to stabilize populations of this newt. Road mortality is a clear issue in some areas, particularly south of the Santa Monica Mountains. Road signage has been used to try to reduce road mortality in Monterey County, although its effectiveness is not known. Migration barriers and under-road tunnels may reduce vehicular death in key areas, though research is needed into the design and efficacy of such interventions (Schmidt and Zumbach 2008).

Monitoring, Research, and Survey Needs

Research into terrestrial habitat use and movement is critical for understanding habitat requirements and potential corridors of movement among populations, and these should be undertaken for both stream- and pond-breeding sites. Monitoring of sites where invasive species have been removed should be conducted to determine the long-term efficacy of removals and the recovery time and stability of populations following removal. Genetic analyses at the landscape level could be very informative with respect to both metapopulation dynamics and habitat corridor use and should be conducted in both relatively intact (e.g., Santa Monica Mountains) and more fragmented landscapes. Research is also needed into potential management strategies for dealing with wildfire and erosion control in order to protect breeding habitat.

CALIFORNIA LEGLESS LIZARD

Anniella pulchra Gray 1852

Status Summary

Anniella pulchra is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 55% (61/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Anniella pulchra is a medium-sized (11.1–17.8 cm SVL), elongate, legless lizard that is snakelike in body form. This species possesses several characteristics that are related to an underground burrowing lifestyle such as smooth cycloid scales, a shovel-shaped snout, countersunk jaw, a short blunt tail, and the absence of external ear openings (Stebbins 2003). The dorsal coloration is generally metallic light silver or golden with a black middorsal line down the length of the body and black lateral stripes. *Anniella pulchra* typically have a lemon-yellow ventral coloration. Faintly striped variants sometimes occur, and dark-brown and black forms occur on the Monterey peninsula and around Monterey Bay, as well as from Morro Bay, Monterey County, south to Gaudalupe, Santa Barbara County (Stebbins 2003). This species is unlikely to be confused with other lizard species

California Legless Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	7
v. Ecological tolerance (10)	7
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	61
Total Possible	110
Total Score/Total Possible	0.55

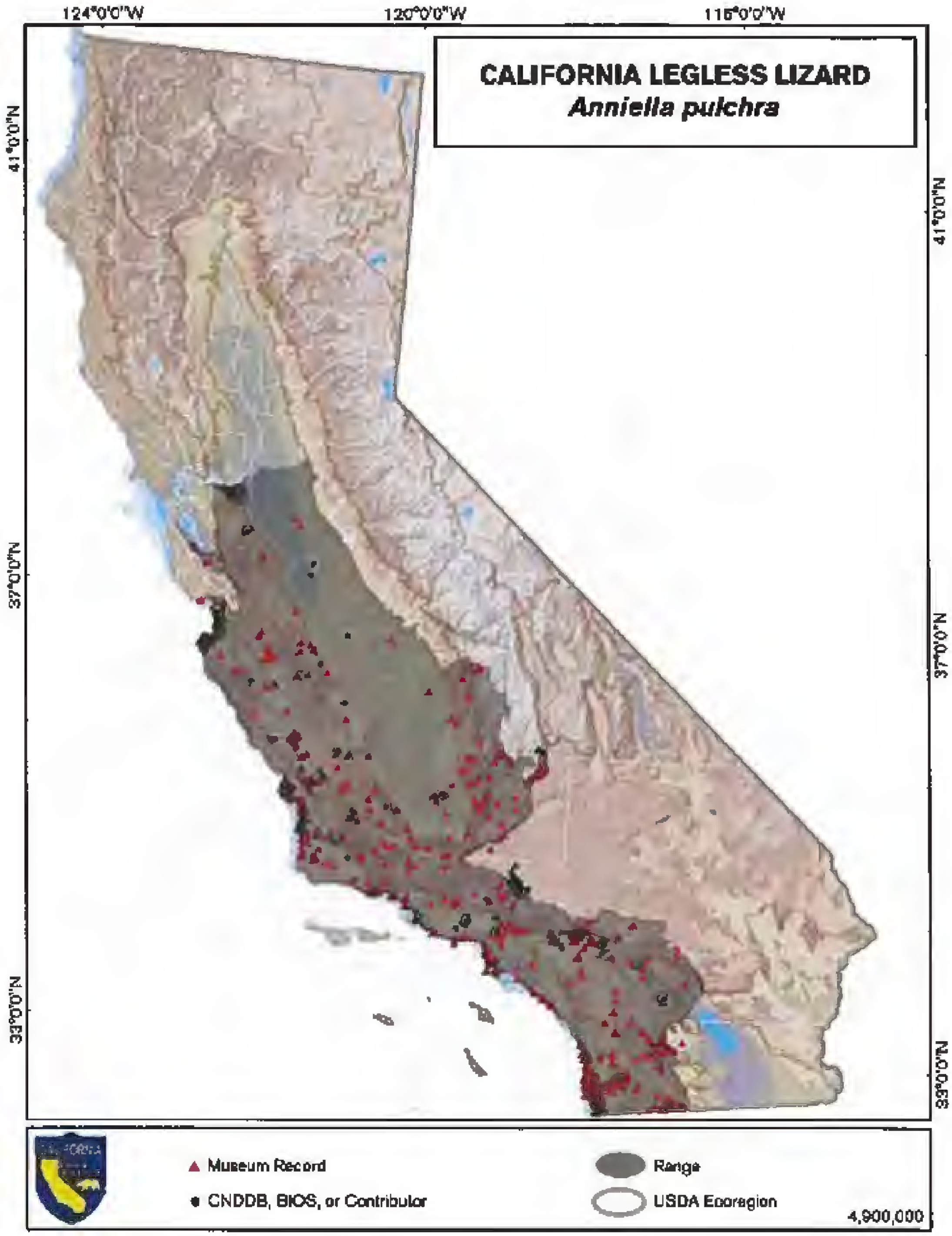


PHOTO ON PREVIOUS PAGE: California legless lizard, Kern County, California. Courtesy of Adam Clause.

in California because it is our only legless lizard. Though *A. pulchra* bears a superficial resemblance to some snake species, the presence of moveable eyelids effectively distinguishes it.

Taxonomic Relationships

Here we treat all California animals as a single species, *Anniella pulchra*. There is substantial evidence for population structure within this species in California from karyotype, allozyme, mitochondrial DNA, nuclear DNA, and morphological studies (e.g., Bezy and Wright 1971, Bezy et al. 1977, Rainey 1985, Pearse and Pogson 2000, Parham and Papenfuss 2009, Papenfuss and Parham 2013). A recent genetic study by Parham and Papenfuss (2009) identified five major lineages within California and documented more extensive genetic diversity within the species than previously reported. Papenfuss and Parham (2013) subsequently proposed that these clades be elevated to species status based on genetic information and some additional data on morphology. This revision occurred as we were finishing onr evaluation of special concern status, and we retain the traditional arrangement here to allow the herpetological community time to evaluate this proposed change in taxonomy.

Life History

Breeding occurs between early spring and July in these live-bearing lizards. Oviductal eggs have been observed between July and October, and 1–4 young are born after a 4-month gestation period (Miller 1944, Goldberg and Miller 1985). Juveniles grow rapidly (2.5–4.4 mm SVL/month) and reach sexual maturity after about 2 years at ~9 cm SVL for males and after about 3 years at ~12 cm SVL for females (Miller 1944, Goldberg and Miller 1985). Life span in the field is unknown, but captive animals have survived for almost 6 years (L. Hunt, pers. comm., in Jennings and Hayes 1994a).

Anniella pulchra is rarely seen active on the surface, but they do use the soil/litter interface for feeding and mating (Miller 1944). Daily activity patterns peak in the morning and evening, though animals have been observed active at night (Miller 1944, Stebbins 1954, Gorman 1957, Bury and Balgooyen 1976, Kuhnz 2000). Coastal and southern populations are likely active year-round, while inland populations (e.g., Sierra Nevada foothills) may enter a period of dormancy during cold months (Banta and Morafka 1968, Zeiner et al. 1988).

Little is known about movement ecology. These fossorial lizards have been found at soil depths from a few to 50 cm below the surface (Miller 1944, Hunt 1984, Kuhnz 2000). Animals have been observed burrowing to a depth of 46 cm in the laboratory (Kuhnz 2000). In one short-term study (~2 months), 10 lizards were recaptured within 10 m of their original capture points (Miller 1944). A two-year PIT tagging study documented an average home range size of 71 m^2 (Kuhnz 2000).

Anniella pulchra prefers lower temperatures than most other California lizards (~21–28°C in lab trials, Bury and Balgooyen 1976; critical thermal maximum 34°C, Brattstrom 1965), which is consistent with a non-basking fossorial lifestyle. Surface activity by this species is likely limited by both ambient and substrate temperature (Miller 1944).

Little is known about the feeding ecology of this species. *Anniella pulchra* is a generalist sit-and-wait insectivore (Coe and Kunkel 1906, Miller 1944) that eats larval insects (e.g., microlepidopterans and beetles), adult beetles, termites, and spiders (L. Hunt, pers. comm. in Jennings and Hayes 1994a).

Habitat Requirements

At a regional scale, *Anniella pulchra* occurs in sparsely vegetated habitat types including coastal sand dunes, chaparral, pine–oak woodland, desert scrub, open grassland, and riparian areas (Stebbins 2003; S. Sweet, pers. comm.). At local scales, this lizard is a microhabitat specialist requiring sandy or loose loamy substrates conducive to burrowing (Miller 1944, Gorman 1957, Cunningham 1959a, Banta and Morafka 1968). Soils that are not used include gravel-sized substrates and those with greater than approximately 10% clay content, resulting in absence of this species from serpentine and shale bedrock (S. Sweet, pers. comm.).

At a Monterey County coastal sand dune site, *A. pulchra* used non-compacted, organic-rich soil preferentially and were most abundant in undisturbed soil types, although they were also found in slightly cemented clay-/silt-rich sands (Kuhnz et al. 2005). Plant community structure also contributed to microhabitat suitability, with *A. pulchra* more common around native shrubs such as silver bush lupine, mock heather, and yellow lupine and less common around nonnative grasses, forbs, and iceplant (Kuhnz et al. 2005). In the Mojave Desert, *A. pulchra* can be found in leaf litter under juniper trees (*Juniperus*) (J. Parham and T. Papenfuss, pers. obs.). Soil moisture may also be a limiting factor for this species (Burt 1931, Miller 1944, Bury and Balgooyen 1976). Kuhnz et al. (2005) found more lizards in the low areas between dunes than in other areas, which may be due to water retention.

Distribution (Past and Present)

Most of the range of *Anniella pulchra* occurs in California, from Contra Costa County south through the Coast Ranges, in parts of the San Joaquin Valley, the western edge of the Sierra Nevada Mountains, the western edge of the Mojave Desert, and northern Baja California (Hunt 1983, Jennings and Hayes 1994a). Although most commonly found within 100 km of the coast, *A. pulchra* ranges in elevation from sea level to about 1800 m (Hunt 1983).

Based on the assumption that *A. pulchra* cannot persist in habitat where soil has been disturbed (e.g., plowing, bulldozing), Jennings and Hayes (1994a) estimated that ~20% of historical habitat is no longer suitable. Parham and Papenfuss (2009) noted that several localities they sampled around Bakersfield in the early 2000s no longer existed by the time their study was published. However, some populations have persisted in developed areas, particularly around fence lines, road verges, utility corridors, and gardens (S. Sweet, pers. comm.). For example, populations that were present in the 1970s were still extant in the 2000s in Fontana, San Bernardino County, in residential areas that were formerly the Delhi Dunes (S. Barry, pers. comm.). *Anniella pulchra* has also been observed in irrigated gardens in Contra Costa County where naturally sandy soils are available (E. Ervin, pers. obs.). The long-term viability of populations in such developed areas is an important research question.

Trends in Abundance

Very few population size estimates are available for this cryptic species. *Anniella pulchra* can be locally abundant, with the highest documented density of 1.67/m² occurring under a single yellow lupine bush in coastal dune habitat at Moss Landing, Monterey County (Kuhnz et al. 2005). Given the high degree of development within its coastal range, we suspect that some populations are declining. In particular, the black form on the Monterey Peninsula may be at great risk given the substantial development pressure in the region.

Nature and Degree of Threat

The greatest threats to *Anniella pulchra* are habitat loss and degradation, and climate change is also a potential emerging threat. Anthropogenic impacts that disturb soil moisture levels or result in soil compaction likely degrade habitat suitability for this species. While some disturbance may be tolerated, development that covers large areas (>8 ha) can potentially cause local extinctions of *A. pulchra* (S. Sweet, pers. comm.). Invasive plants may also have a negative impact on habitat suitability and abundance (Kuhnz et al. 2005). Over the next 100 years, mean annual temperature is expected to increase throughout the range of *A. pulchra* (reviewed in PRBO 2011). There is greater uncertainty in how precipitation will change, with some models predicting decreases in precipitation of up to 37% and other models predicting no change or only moderate declines (Bell et al. 2004, Snyder et al. 2004, Snyder and Sloan 2005, PRBO 2011). Warmer and drier conditions might limit activity to deeper soil depths, although the population impacts of such a shift are unknown. Alterations in vegetation communities due to climate change may pose a larger threat to this species, as increases in grassland habitat are predicted through much of its range with concomitant decreases in preferred open habitat types such as coastal scrub, particularly in southern California (Lenihan et al. 2008, PRBO 2011). The frequency and size of fires in the Coast Ranges is expected to increase up to 50% by the end of the century (Fried et al. 2004, Lenihan et al. 2008, Westerling and Bryant 2008). Fire dynamics are more difficult to predict in southern California, partly due to the role of Santa Ana winds (reviewed in PRBO 2011, Franco et al. 2011). How fire affects *A. pulchra* is unknown. Direct mortality effects may be small due to its subterranean lifestyle, although indirect negative effects may occur through habitat shifts and changes in soil chemistry.

Status Determination

Anniella pulchra is a near-endemic, ecologically specialized lizard with much of its range occurring in heavily populated and impacted coastal areas. Little data is available on the abundance of this cryptic species, particularly in non-dune habitats, which limits our ability to quantify population trends or document extirpations.

Management Recommendations

Protection of dune areas both along the coast and in the Coast Range is critical. In occupied areas, disturbances such as development, agriculture, and off-highway vehicle use should be reduced or eliminated. Activities that compact soil, in particular, should be avoided. Given that *Anniella pulchra* appears to persist in some developed areas provided that sandy soils and native plant communities remain intact, incentivizing or requiring natural landscaping in low-density housing (as has been done in Monterey County for the federally and state endangered Santa Cruz long-toed salamander, *Ambystoma macrodactylum croceum*) may allow lizards to coexist with some development. The spread of nonnative plant species into remaining habitat should be minimized. Eradication of invasive plants and restoration of native vegetation may help increase *A. pulchra* density and should be explored.

Monitoring, Research, and Survey Needs

With a few exceptions (e.g., Miller 1944, Kuhnz et al. 2005), little is known about *Anniella pulchra* abundance across its range. Studies of basic ecology are needed in other parts of the range and in other habitat types. Minimally, surveys summarizing habitat use, soil characteristics, and population density in coastal southern California and the southern Sierra Nevada should be conducted to complement work in Monterey County. Understanding under what conditions this species can persist in human-disturbed habitats would be valuable, particularly with respect to soil characteristics and fragmentation that occurs as a consequence of urbanization and agricultural land use. *Anniella pulchra* co-occurs with Argentine ants (*Linepithema humile*) along the coast, but it is unknown whether this introduced species has any substantial impacts on *A. pulchra*.

Presence and abundance of this cryptic species are both difficult to assess, and more research into the best sampling methods for different habitats would be useful for the development of monitoring efforts. In a comparison of survey techniques in dune habitat in Monterey County, Kuhnz et al. (2005) concluded that time-constrained searches were the most reliable method for detecting *A. pulchra* presence across a range of population densities and dune vegetation types. In time-constrained searches, surveyors searched the surface, under dried vegetation or cover objects, and up to 15 cm below the surface. Kuhnz et al. (2005) noted that all survey methods were poor at detecting lizards at low densities of ~1/100 m^2, and even time-constrained searches greatly underestimated density compared to depletion raking (raking of substrate until one or fewer individuals were found per 40 hours of search effort). However, these results may not apply in general across habitat types. For example, some investigators prefer to use cover objects at inland sites where *A. pulchra* is relatively rare (J. Parham, pers. comm.).

Additional genetic analyses at the population level may be the best way to efficiently determine the effective population size and genetic connectivity of apparently isolated populations. Particularly in conjunction with intensive time-constrained surveys, genetic data can be used to measure habitat-specific gene flow, current population size, and changes in population size. We recommend that appropriate genetic markers be developed and that tissues be collected and deposited in appropriate repositories for such analyses.

COASTAL WHIPTAIL

Aspidoscelis tigris stejnegeri (Van Denburgh 1894)

Status Summary

Aspidoscelis tigris stejnegeri is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 54% (59/110). It was not considered a Species of Special Concern during the previous evaluation (Jennings and Hayes 1994a).

Identification

Aspidoscelis tigris stejnegeri is a member of the *A. tigris* species complex, a group of 8–13 species that are all similar in appearance (Grismer 2002, Reeder et al. 2002, Stebbins 2003). This is a large (6–12.7 cm SVL), extremely active, diurnal lizard with a slim body and a long tail. The dorsal ground color is dark, with a series of lighter tan or beige spots forming stripes down the sides. These stripes may be broken and irregular, suggesting a checkered appearance (Stebbins 2003, Lemm 2006). The ventral coloration is whitish to cream with scattered black spotting which sometimes forms longitudinal lines between the scale rows (Stebbins 2003). The dorsal scales are granular, while the ventral scales are relatively large, rectangular plates (Lemm 2006). The scales on the head are also enlarged dorsally and ventrally, forming plates in front of the gular fold (Lemm 2006). In the San Diego area, juveniles develop a distinctive

Coastal Whiptail: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	3
v. Ecological tolerance (10)	3
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	10
Total Score	59
Total Possible	110
Total Score/Total Possible	0.54

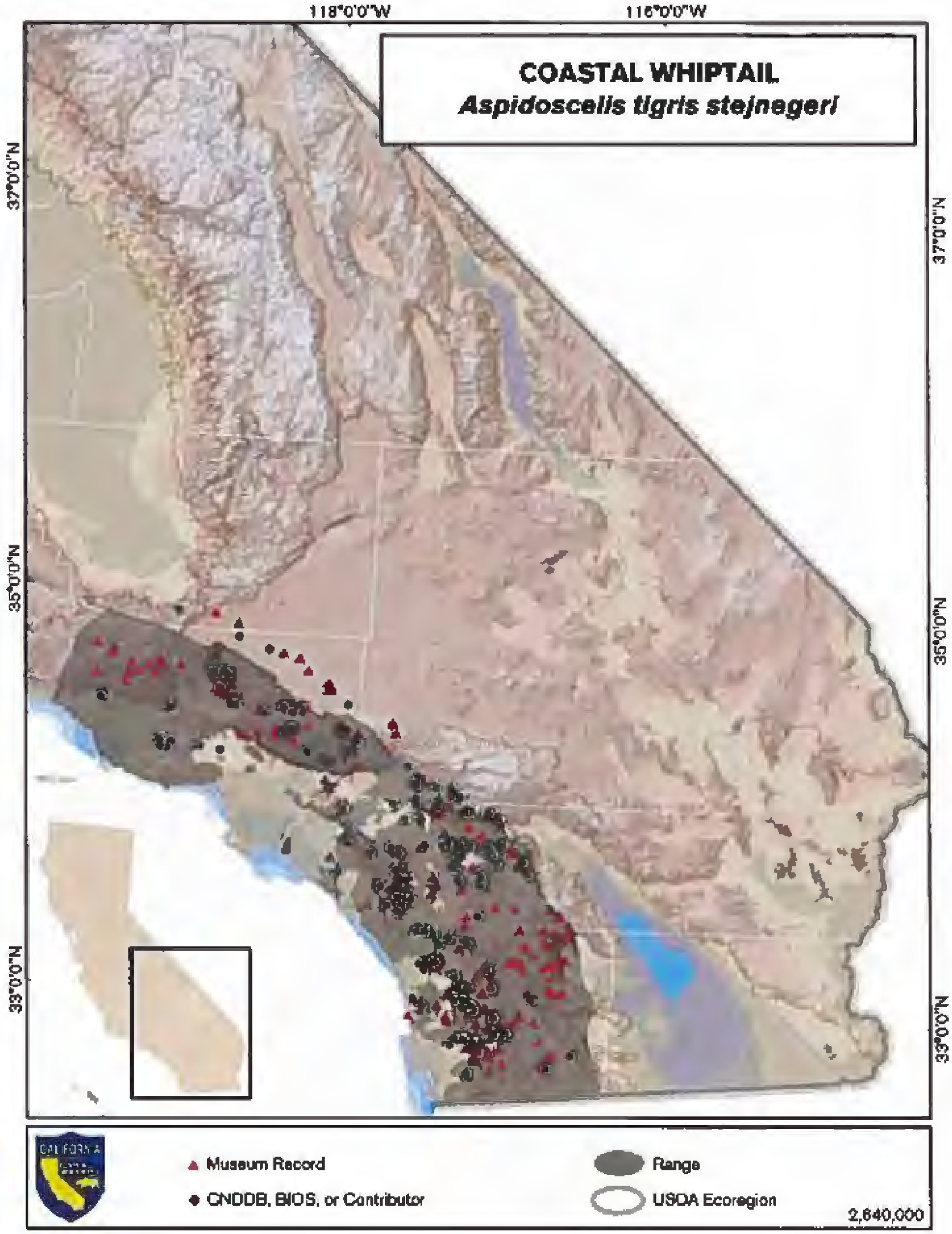


PHOTO ON PREVIOUS PAGE: Coastal whiptail, Los Angeles County, California. Courtesy of Robert Hess.

spotted pattern (Stebbins 2003; R. Fisher, pers. comm.).

Within its range, *A. t. stejnegeri* is only likely to be confused with its congener, the orange-throated whiptail (*A. hyperythra*). Both lizards have similar body shapes and scalation, though *A. hyperythra* is usually smaller (5–7.2 cm SVL) and is marked with well-defined light stripes and an intervening dark ground color (Stebbins 2003). In addition, the males of *A. hyperythra* develop a conspicuous bright orange coloration on the throat and underside of the body and juveniles have bright blue on the tail (Stebbins 2003, Lemm 2006).

Taxonomic Relationships

No modern studies of phylogenetics, phylogeography, or species boundaries exist within the *Aspidoscelis tigris* species complex, although the validity of this subspecies has not been questioned. Reeder et al. (2002) presented a phylogenetic analysis of whiptail lizards of the genus *Cnemidophorus* (sensu lato) and showed that the genus, as historically defined, was not monophyletic. To remedy this, they moved North American whiptails to the genus *Aspidoscelis*, an arrangement that is now widely accepted.

Some confusion surrounds the application of the name *A. t. stejnegeri* in the literature. A closely related whiptail occurs as an insular endemic on Isla Cedros, Baja California, Mexico, which most authors refer to as the subspecies *A. t. multiscutata* (previously, *Cnemidophorus tigris multiscutatus*). However, others have treated *A. t. stejnegeri* as a junior synonym of *A. t. multiscutata* and refer both the insular endemic and the coastal southern California forms to this latter name. Thus, some literature referring to the *A. tigris* subspecies in southern California uses *A. t. multiscutata*. This has sometimes led authors to consider the two names to refer to two separate biological taxa that both occur in southern California (Maslin and Secoy 1986). To clarify, there is only a single member of the *A. tigris* complex in coastal southern California, and its currently accepted name is *A. t. stejnegeri*.

Life History

The life history of *Aspidoscelis tigris stejnegeri* is poorly studied, particularly within its California range, although it is probably similar to other subspecies within the *A. tigris* species complex. This is a diurnally active, wary lizard, which rarely stops moving during its activity period. *Aspidoscelis tigris stejnegeri* is a generalist predator that actively searches for insects, spiders, scorpions, and other small arthropods, including larvae (Grismer 2002, Lemm 2006). Some subspecies in the complex are known to prey upon small lizards, though this has not been documented in *A. t. stejnegeri* to our knowledge. *Aspidoscelis tigris stejnegeri* is a relatively high-temperature specialist that emerges to begin foraging in late morning as the air temperature rises. It can become active as early as mid-March and remain so until early October, although juveniles can remain active into November (Grismer 2002). When active, *A. t. stejnegeri* moves with a distinctive gait, taking a step, halting briefly, then moving again in rapid succession.

Reproduction takes place in spring and summer. Grismer (2002) documented gravid females and courtship behavior in mid-July in Baja California. Courtship may occur earlier in the California populations (Lemm 2006), although few data exist. Hatchlings begin to appear in late July and August in Baja California; again, this may occur earlier in California (Grismer 2002, Lemm 2006).

Habitat Requirements

Aspidoscelis tigris stejnegeri can be found in a wide variety of habitats within the California portion of its range, including coastal sage scrub, chaparral, riparian areas, woodlands, and rocky areas (Lemm 2006). Early observations of this subspecies in California, as well as data from the Baja California portion of the range, indicate that the species prefers sand-and/or gravel-bottomed habitats and brushy areas associated with washes—habitats that have largely been destroyed by development in southern California (J. Grinnell, pers. comm. reported in Van Denburgh 1922). The species continues to persist outside of these preferred habitats, particularly in open chaparral and coastal sage with a gravelly substrate (Grismer 2002, Cooper and Matthewson 2008), although possibly at reduced densities. *Aspidoscelis tigris stejnegeri* requires large blocks of contiguous habitat and is rarely encountered where development and roads have fragmented the available habitat (Case and Fisher 2001, Brehme 2003, Cooper and Matthewson 2008).

Distribution (Past and Present)

Aspidoscelis tigris stejnegeri was formerly present in California from the southern slopes of the Transverse Ranges south to the United States–Mexico border and east to the Peninsular Ranges (Van Denburgh 1922). In Mexico, it ranges farther south between the coast and the western slopes of the Peninsular Ranges, eventually intergrading with the reddish whiptail (*A. t. rubida*) in the Vizcaino region of the central Baja California peninsula. In California the species occurs from sea level to about 1500 m (Lemm 2006).

The species is apparently extirpated, or nearly so, from large areas of the Los Angeles basin and the San Diego region due to habitat loss. By 1922, the species was already scarce in the vicinity of Pasadena, reportedly as a result of habitat loss due to development (J. Grinnell, pers. comm. reported in Van Denburgh 1922). Further declines have occurred throughout the Los Angeles basin and in coastal San Diego County (Stebbins 2003; R. Fisher, pers. comm.). Much of the inland range is still intact, though increasing wildfires may pose a threat (Rochester et al. 2010).

Trends in Abundance

Few data exist regarding historical abundance of this species, although it is susceptible to habitat fragmentation and development. Cooper and Matthewson (2008) reported that the species is rarely encountered in small habitat patches and is an indicator species for large blocks of unfragmented coastal sage and chaparral habitat. Grinnell (1908) reported seeing "many of them" along the lower Santa Ana canyon, San Bernardino County, California, in 1905. This area is now heavily modified and does not provide ideal habitat for this taxon. By 1922, the lizard was reportedly "rare" in the vicinity of Pasadena because of habitat fragmentation and loss (J. Grinnell, pers. comm. reported in Van Denburgh 1922), although Bogert (1930) reported it as being moderately common throughout the southern foothills of the San Gabriel Mountains and most of the Santa Monica Mountains. Atsatt (1913) reported that it was frequently encountered throughout several areas of the San Jacinto Mountains, Riverside County, California. Because habitat fragmentation and loss have continued to occur throughout its range, it is reasonable to assume that declines are continuing.

Nature and Degree of Threat

The primary threat facing *Aspidoscelis tigris stejnegeri* is habitat loss and fragmentation due to development. This species occurs in some of the largest population centers within California and requires relatively large habitat blocks, making it particularly susceptible to urbanization. Further, the increasing frequency and intensity of wildfires in southern California may convert large portions of its remaining habitat to suboptimal grassland, causing further declines in range and/or abundance (Lemm 2006, Rochester et al. 2010, R. Fisher, pers. comm.). Projections from several climate models suggest that the frequency and intensity of wildfires in southern California could increase, although these results appear to be strongly dependent on the model that is employed (Cayan et al. 2008b, Franco et al. 2011, PRBO 2011). If this occurs, additional habitat destruction is likely to occur, negatively impacting this species.

Status Determination

Documented and ongoing declines in the distribution of this species, coupled with ongoing suspected declines in abundance, are the primary contributors to this status. *Aspidoscelis tigris stejnegeri* also has a relatively small range in California. Projected impacts from wildfire (which may increase with future climate change) coupled with the above impacts justify a Priority 2 status.

Management Recommendations

Conservation of remaining habitat is essential for the long-term protection of this species. Habitat protection efforts should focus on maintaining large, unfragmented blocks, and this species should be included in large-scale planning efforts like Natural Community Conservation Planning where the process permits. Establishing the minimum size of habitat blocks is a critical research need. Until these data become available, additional fragmentation and degradation should be prevented in habitat patches that currently support this taxon, and corridors of suitable habitat that connect occupied patches should be identified, protected, and/or restored as necessary.

Monitoring, Research, and Survey Needs

Additional data on this taxon's home range size, habitat requirements, and movement ecology are required to determine the minimum patch sizes and maximum amount of fragmentation that can support viable populations. As for many active, wide-ranging species, the effects of road traffic on mortality would be valuable information for future management efforts. Abundance surveys should be conducted in remaining populations of *Aspidoscelis tigris stejnegeri*. Information on abundance should be correlated with the local habitat patch size to better understand the minimum patch size required for population persistence. Further research should examine the effect of moderate habitat fragmentation on existing populations if habitat corridors between patches can be maintained. Given the patchy nature of the species, a landscape genetic approach that quantified both connectivity and effective population sizes of remaining populations would be valuable.

SAN DIEGO BANDED GECKO

Coleonyx variegatus abbotti Klauber 1945

Status Summary

Coleonyx variegatus abbotti is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 54% (59/110). It was not considered for Species of Special Concern status during the previous evaluation (Jennings and Hayes 1994a).

Identification

Coleonyx variegatus abbotti is a small (maximum 5.8 cm SVL) lizard with slender padless toes, moveable eyelids, vertical pupils, and soft skin covered in fine granular scales (Klauber 1945, Grismer 2002, Stebbins 2003). The dorsal ground coloration is variable and ranges from pale yellow to grayish pink. A series of contrasting darker-brown or tan lateral crossbands extend down the length of the body and are approximately the same width or narrower than the intervening areas of ground coloration. Areas between bands occasionally contain spots of the darker coloration. The head is dark and usually unmarked or only lightly mottled with a narrow light nuchal crescent extending backward from the eyes (Klauber 1945). The ventral surface is semi-transluscent and immaculate white to faint pink. The juvenile pattern is similar to that of adults, but the

San Diego Banded Gecko: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	3
v. Ecological tolerance (10)	7
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	59
Total Possible	110
Total Score/Total Possible	0.54

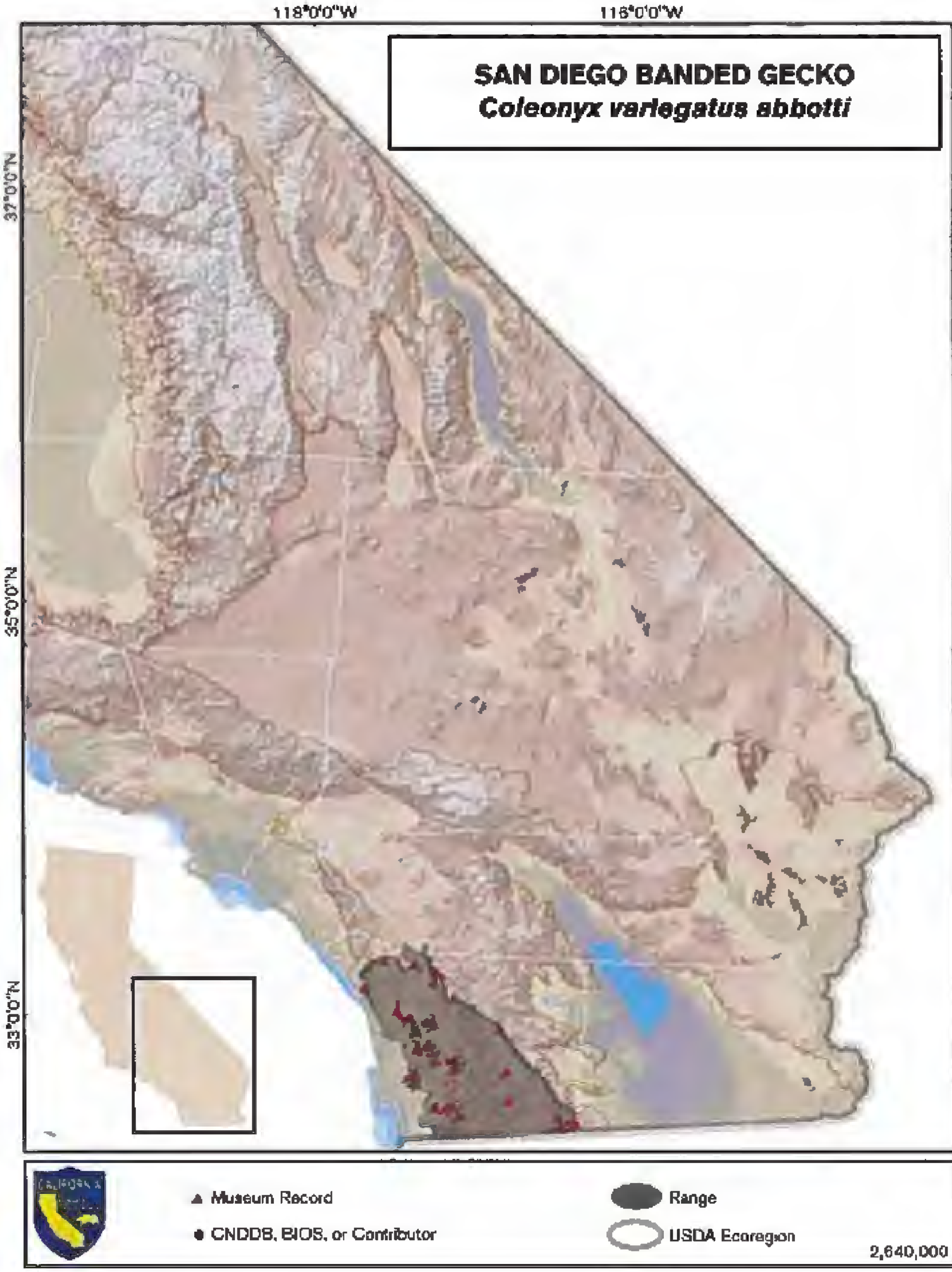


PHOTO ON PREVIOUS PAGE: San Diego banded gecko, San Diego County, California. Courtesy of Jeff Lemm.

coloration is often more pronounced and contrasting.

In California, this subspecies is only likely to be confused with other geckos that occur nearby. The closely related desert banded gecko (*C. v. variegatus*) is parapatric with *C. v. abbotti* along the Peninsular Ranges of Southern California and adjacent Baja California, Mexico. The two taxa are best distinguished based on color pattern, locality, and size. Although color and pattern in both subspecies are variable, *C. v. variegatus* generally lacks the nuchal collar, often has wider and less well-defined dark bands, has extensive spotting on the head, and attains larger overall body sizes (up to 7.1 cm) (Klauber 1945, Grismer 2002, Stebbins 2003, Lemm 2006). The two subspecies intergrade across narrow contact zones in Baja California and probably also in southern California, with *C. v. abbotti* occurring on the coastal side of the Peninsular Range mountains and *C. v. variegatus* on the inland side (Klauber 1945; D. Leavitt, pers. comm.). In some areas, animals that are morphologically referable to *C. v. abbotti* are genetically more similar to *C. v. variegatus* (D. Leavitt, unpublished data; see the "Taxonomic Relationships" and "Distribution" sections). The barefoot banded gecko (*C. switaki*) also has a superficially similar appearance but is more rarely encountered. In California, it has only been found in a narrow area of the Peuinsular Range. Other geckos in southern California have expanded toe pads and immovable eyelids and are often extreme habitat specialists (Stebbins 2003).

Taxonomic Relationships

Coleonyx variegatus abbotti is a close relative of *C. v. variegatus*. Its initial description was based primarily on coloration, pattern, and scalation (Klauber 1945). Sequence data from seven nuclear DNA markers confirm the distinctiveness of *C. v. abbotti* but also restrict the known range (see the "Distribution" section). The two taxa are not genetically isolated but have an abrupt genetic and morphological contact zone in Baja California and possibly also in Southern California (D. Leavitt, unpublished data).

Life History

The life history of *Coleonyx variegatus abbotti* is poorly studied, although it is likely similar to that of the better-studied *C. v. variegatus* in many respects. *Coleonyx variegatus abbotti* is active from March until September or October (Lemm 2006). It is nocturnal, emerging from rock crevices and burrows usually within 2 hours following sunset. Like other geckos, it is a predator, presumably taking a variety of small invertebrates, although the diet has not been studied in detail (Kingsbury 1989, Grismer 2002).

Reproduction takes place in late spring. Females lay one or two eggs at a time (Lemm 2006). Other subspecies of *C. variegatus* are known to lay up to three clutches per year between May and September (Stebbins 2003), and this may also occur in *C. v. abbotti*. Juveniles have been found as late as September (Lemm 2006).

Habitat Requirements

Coleonyx variegatus abbotti is restricted to rocky coastal sage and chaparral habitat, usually in areas between 150 and 900 m in elevation (Lemm 2006). Klauber (1945) noted that the subspecies seems to prefer areas with granite outcrops, though it is not restricted to them and has been found in dry rocky riverbeds. Most specimens have been found under cover objects or on roads at night. It is more frequently found under large cap rocks than under the small rock flakes favored by other small lizard species such as the granite night lizard (*Xantusia henshawi*) (Klauber 1945). Extensive pitfall trapping data indicate that *C. v. abbotti* is absent from areas with a high intensity of artificial night lighting (Perry and Fisher 2006; R. Fisher, unpublished data).

Distribution (Past and Present)

Ongoing genetic analyses of the *Coleonyx variegatus* complex are revising our understanding of *C. v. abbotti*'s distribution, and thus our current concept of its range may change as these studies are completed. Historically, all *Coleonyx* ranging from the United States–Mexico border north along coastal and cismontane Southern California were considered *C. v. abbotti*. However, genetic data indicate that the range is more limited and primarily restricted to San Diego County, with populations farther north belonging to *C. v. variegatus* (D. Leavitt, unpublished data). The extent of the potential intergrade zone between the two subspecies is not yet well understood. In Mexico, *C. v. abbotti* ranges from the border south along coastal and cismontane Baja California to the vicinity of Cataviña, then extends east across the peninsula and south, eventually intergrading with the Peninsular banded gecko (*C. v. peninsularis*) in the Vizcaíno mid-peninsula region.

Geckos have disappeared from much of coastal San Diego County, primarily in areas with high-intensity artificial night lighting (Perry and Fisher 2006). Whether night lighting itself or other habitat changes associated with artificial night lighting drove the declines is not well studied. Development and agricultural impacts have also extirpated geckos from some areas (R. Fisher, pers. comm.).

Trends in Abundance

Few quantitative data on historical or current abundances are available, although *Coleonyx variegatus abbotti* is less frequently encountered than *C. v. variegatus* farther east (Lemm 2006). This was apparently also the case historically. Klauber (1945) specifically noted that *C. v. abbotti* was less common throughout its range than *C. v. variegatus*. Bogert (1930) also reported that the geckos were rare in Los Angeles County, although genetic data suggest these might actually have been *C. v. variegatus*. Pitfall surveys indicate that the subspecies is found at a small number of sites within southern California (7 out of 21 survey areas, 15 individuals in total) compared to lizard species occupying similar habitats (Case and Fisher 2001). However, these surveys were not designed to specifically target *Coleonyx*, and no historical baseline data exist with which to compare current abundances.

Nature and Degree of Threat

The primary threat facing *Coleonyx variegatus abbotti* is apparently habitat loss due to agricultural and urban development, including deaths from automobile traffic. Some data further suggest that artificial night lighting is correlated with declines, although no causal link has been established. Climate change within its limited range is expected to increase the frequency and intensity of wildfires, which could degrade some currently suitable habitat. Finally, *C. v. abbotti* is encountered relatively rarely even in suitable habitat, which poses significant challenges in monitoring population trends and the impacts of habitat disturbance.

Status Determination

Coleonyx variegatus abbotti has a restricted range in California that falls within an area that is currently experiencing a large amount of development. Some data suggest that the subspecies has disappeared along the coast in a substantial fraction of its range. This, coupled with the ongoing habitat loss due to development and wildfire, could reduce the current distribution further and justifies a Priority 3 Species of Special Concern designation.

Management Recommendations

The most important management priority for *Coleonyx variegatus abbotti* is to protect remaining habitat. Our current understanding of habitat requirements and this taxon's sensitivity to habitat degradation is unfortunately weak, and there is a strong need for additional study before a thorough and informed management strategy can be developed.

Monitoring, Research, and Survey Needs

The relative rarity with which this subspecies is encountered makes the detection of past and ongoing declines difficult. A comparison of survey protocols for this subspecies, including time-constrained searches and pitfall trapping should be initiated. A goal of this comparison should be to develop a survey protocol that is capable of detecting changing abundances. A mark–recapture study would help determine whether the apparently low population densities currently observed reflect detectability or true population numbers. This should include a power analysis to clarify the trapping intensity needed in order to detect changes of varying magnitude. Surveys should include relatively pristine sites, moderately disturbed habitats, and those with varying degrees of artificial night lighting. Survey data should also be utilized to inform our understanding of habitat preferences, seasonality, and life history in this taxon.

Additional genetic surveys should also be undertaken to further clarify the range limits and genetic differentiation among members of the *Coleonyx variegatus* complex. In particular, contact zones between different subspecies should be further studied in order to develop a clear understanding of the range for both taxa in southern California. Landscape genetic studies would help to inform management in terms of connectivity of remaining populations and potentially help identify habitat corridors. Information from genetics, morphology, and survey data should be integrated to develop a more comprehensive understanding of differentiation between this subspecies and other members of the *C. variegatus* complex.

PANAMINT ALLIGATOR LIZARD

Elgaria panamintina (Stebbins 1958)

Status Summary

Elgaria panamintina is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 44% (48/110). During the previous evaluation, it was also designated as a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Elgaria panamintina is a large (9.2–15.2 cm SVL), slender, elongate lizard with a light yellow-brown or beige dorsum and a series of contrasting brown crossbands extending from the neck down the length of the body and tail (Stebbins 1958, Banta et al. 1996, Stebbins 2003). The ventral surface is light gray or cream, with small dusky markings forming continuous or broken longitudinal lines that run down the center of each scale row (Jennings and Hayes 1994a, Stebbins 2003). The iris is pale yellow (Stebbins 2003). The contrast between the dark crossbands and lighter dorsal coloration is usually more pronounced in juveniles than in adults. The tail, when intact, is up to twice the length of the body, although shorter broken/regenerated tails are common (Stebbins 2003).

This lizard is unlikely to be confused with any other species within its range. However,

Panamint Alligator Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	0
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	10
vi. Population trend (25)	5
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	10
Total Score	48
Total Possible	110
Total Score/Total Possible	0.44

PHOTO ON PREVIOUS PAGE. Panamint alligator lizard, Inyo County, California. Courtesy of Adam Clause.

two similar congeners occur in much of California: the northern alligator lizard (*E. coerulea*) and the southern alligator lizard (*E. multicarinata*). Neither of these species has the pattern of broad strongly contrasting crossbands down the length of the body. The crossbands are usually interrupted by a longitudinal, middorsal stripe in *E. coerulea* and are much narrower in *E. multicarinata* (Stebbins 2003).

Taxonomic Relationships

Different studies have recovered discordant phylogenetic placements of *Elgaria panamintina*. Good (1988) recovered a sister relationship between *E. panamintina* and the Madrean alligator lizard (*E. kingii*) from Arizona, using a dataset composed of 34 allozyme loci. More recent studies find that *E. panamintina* is nested within *E. multicarinata*, a placement that was supported by both mitochondrial sequence data (Feldman and Spicer 2006) and nuclear sequence data (D. Leavitt et al., unpublished data).

Leavitt et al. (unpublished data) found low levels of variation among populations of *E. panamintina* and no evidence for recent or ongoing gene flow between this species and other *Elgaria* in western North America. The discordance of the allozyme and nuclear sequence data, and therefore the monophyly of *E. multicarinata* with respect to *E. panamintina*, awaits further investigation.

Life History

The life history of *E. panamintina* is poorly understood. The species spends a large amount of time in rock piles and deep vegetation or brush, so it is not commonly observed (Stebbins 1958, Macey and Papenfuss 1991b). We presume that many aspects of *E. panamintina*'s life history are similar to that found in the better-studied *E. multicarinata*, particularly given the recent molecular evidence of their very close relationship.

Elgaria panamintina emerges from hibernation in late winter or spring, with higher-elevation populations becoming active later in the year. The species is generally diurnal in the spring through midsummer, when it may switch to nocturnal activity or aestivation, presumably as a response to increasing daytime temperatures (Stebbins 1958, Banta 1963, Dixon 1975, Stebbins 2003). Reproduction has not been documented in the wild, although captive animals have been observed copulating in mid-May (Banta and Leviton 1961). *Elgaria multicarinata* enters reproductive condition at this time of year as well (Goldberg 1972), so we assume that reproduction occurs in mid-spring, although the precise timing likely depends on elevation. Goldberg and Beaman (2003) examined sperm formation in museum specimens and concluded that reproduction takes place during the spring. Like *E. multicarinata* (and unlike *E. coerulea*), *E. panamintina* is oviparous. *Elgaria multicarinata* typically lays eggs in early summer, and we assume that *E. panamintina* does as well (Goldberg 1972). The timing of reproductive events in *E. multicarinata* varies among areas, with some populations producing only one clutch a year and others up to three (Burrage 1965, Goldberg 1972). No data on the number of clutches produced per year or incubation times exist for *E. panamintina*, although Goldberg and Beaman (2003) report a clutch size of four eggs from a single museum specimen.

Dietary data are lacking. We presume that *E. panamintina* is likely a generalist predator like *E. multicarinata*. The latter feeds on a wide variety of insects and other small arthropods, including spiders, centipedes, and scorpions, as well as on small vertebrates, including mice, birds, and lizards (including conspecifics) (Cunningham 1956). Observations in captivity found no obvious differences in feeding behavior between *E. panamintina*, *E. multicarinata*, and *E. kingii*, and we tentatively assume that feeding behavior is also similar in the wild (Stebbins 1958).

Elgaria species have a lower thermal tolerance than most sympatric lizards, which may allow them to maintain higher activity levels in the shaded moist habitats in which they are most commonly found (Cunningham 1956, Stebbins 1958). Predation on *E. panamintina* has not been recorded, though we assume that they are preyed upon by co-distributed lizard-eating snakes (e.g., coachwhips [*Masticophis*] and patch-nosed snakes [*Salvadora*]) and birds (e.g., raptors and roadrunners [*Geococcyx*]).

Habitat Requirements

Elgaria panamintina are most frequently found in rocky canyons in the immediate vicinity of permanent springs and seeps that are patchily distributed across their limited range (Stebbins 1958, Macey and Papenfuss 1991b). The species usually occurs in or adjacent to narrow strips of riparian vegetation immediately below springs and in deep leaf litter and rock piles along the margins of riparian habitat (Stebbins 1958, Macey and Papenfuss 1991b, Jennings and Hayes 1994a). *Elgaria panamintina* was initially thought to be restricted to these areas, but pitfall trapping surveys have documented their presence in arid areas well away from water (Banta 1963). Few quantitative data are available on the relative frequency of arid versus mesic habitat use, and it seems likely that populations require permanent water for persistence.

Distribution (Past and Present)

Elgaria panamintina occurs in relatively remote regions of the Great Basin in California. Given the difficulty of accessing much of its potential habitat and the limited work on the species to date, it may occur more widely than has so far been recorded. The known range encompasses many of the desert mountain ranges of Inyo and southern Mono Counties, including the Panamint, Inyo, Nelson, Argus, and Coso Mountains, as well as the western slopes of the White Mountains (Macey and Papenfuss 1991b, Banta et al. 1996, La Berteaux and Garlinger 1998). The known elevational range extends from 760 to 2290 m (Dixon 1975, Macey and Papenfuss 1991b, Stebbins 2003).

The species' present-day distribution is likely relictual, resulting from gradual drying of the Great Basin throughout the Pliocene and Pleistocene. This general drying has presumably isolated the remaining populations around the few remaining water sources (Stebbins 1958, Good 1988).

Trends in Abundance

No data are available regarding current or historical abundance, although habitat degradation due to mining, livestock grazing, and off-highway vehicle use has likely resulted in population declines (Jennings and Hayes 1994a). Given the very sensitive nature of the remaining islands of mesic habitat in the region, surveys of both population size and connectivity via arid habitat occupancy are needed to provide baseline information on current status.

Nature and Degree of Threat

The primary threat to this species is habitat loss or alteration in its already small range. Many of the known localities occur on private land and are vulnerable to mining, livestock grazing, off-highway vehicle use, and/or diversion of the water sources. Climate change could potentially impact this species if changes in hydrology cause springs to dry up or become less regular in their flow regimes.

Status Determination

Elgaria panamintina is a California endemic with a very small range. It primarily occurs in, and is likely dependent upon, uncommon, small patches of mesic habitat that are scattered widely throughout its range. Each habitat patch is sensitive to several potential disturbances, and if local extirpations occur, natural recolonization seems unlikely. Nearly all known localities occur on unprotected land and are subject to further alteration (Jennings and Hayes 1994a). These factors all contribute to a Priority 3 designation.

Management Recommendations

Terrestrial habitat surrounding permanent springs and seeps should be protected from water diversion and destruction or alteration of riparian vegetation. There may well be conflicts with livestock and large feral mammals since these animals may trample or otherwise disturb the vegetation and leaf litter surrounding desert springs. *Elgaria panamintina* may also occur at additional springs outside of its currently known range; therefore, riparian areas throughout the area should be preserved to the extent possible, even if *E. panamintina* has not yet specifically been documented at them.

Monitoring, Research, and Survey Needs

Surveys should be conducted at additional springs surrounding the known distribution of *Elgaria pananmintina*. These surveys should involve pitfall trapping and/or drift fence arrays, in order to increase detection probabilities. A thorough understanding of *E. panamintina*'s habitat requirements would be invaluable in determining what habitat modifications can be made to riparian areas without negatively impacting the species, as well as identifying suitable areas to focus survey efforts to look for new populations. A key question is the extent to which the species uses arid habitat away from springs, both as corridors for dispersal among springs and as upland habitat. Both drift fence surveys of this habitat and landscape genetic analyses of known spring populations may contribute to greater understanding of habitat use in this species. The lack of basic life history information on *E. panamintina* also needs to be addressed. Mark–recapture surveys would yield important information about population sizes and the extent of migration between springs. This basic information is crucial for any kind of active management and is largely lacking at the present time.

COPE'S LEOPARD LIZARD

Gambelia copeii (Yarrow 1882)

Status Summary

Gambelia copeii is designated as a Species of Special Concern, although we refrain from assigning a priority score due to a paucity of information. This taxon received a Total Score/Total Possible of 45% (38/85). It was not designated as a Species of Special Concern during the previous evaluation (Jennings and Hayes 1994a).

Identification

Gambelia copeii is a large (maximum 14 cm SVL) lizard, with a robust head and limbs, granular body scales, and a long cylindrical tail (Grismer 2002, Stebbins 2003, Lemm 2006, Mahrdt et al. 2010). The dorsal coloration is variable across the range, changing from dark brown in the north to light golden brown or tan in the south (Grismer 2002, Mahrdt et al. 2010). California populations of *G. copeii* form the northern edge of the species' overall range and are dark above with pairs of large, dark paravertebral spots on the dorsal surface that usually fade anteriorly, are almost always absent from the head, and broaden to form transverse bands on the tail (McGuire 1996, Stebbins 2003, Mahrdt et al. 2010). In many individuals, a lighter cream-colored transverse bar separates each pair of these spots along the trunk

Cope's Leopard Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	0
v. Ecological tolerance (10)	0
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	10
Total Score	38
Total Possible	85
Total Score/Total Possible	0.45

PHOTO ON PREVIOUS PAGE: Cope's leopard lizard, San Diego County, California. Courtesy of Rob Schell Photography.

(Mahrdt et al. 2010). Flecking is generally present on the sides, and females in breeding condition develop bright orange or red spots on the sides and underside of the tail (Stebbins 2003). In addition, there is pronounced sexual size dimorphism, with females averaging 6.5 mm larger in SVL and 1.3 mm in head length than males (Lappin and Swinney 1999, Goldberg et al. 2010).

In California, *G. copeii* is unlikely to be confused with other lizards within its range. However, it is found immediately adjacent to the range of the more widely distributed long-nosed leopard lizard (*G. wislizenii*), within which *G. copeii* appears to be phylogenetically nested (McGuire et al. 2007). *Gambelia wislizenii* populations that are adjacent to *G. copeii* are generally paler, with dorsal coloration ranging from off-white to tan and many moderately sized spots asymmetrically scattered along the dorsal surface (McGuire 1996, Grismer 2002). The spotting in *G. wislizenii* does not fade anteriorly, and small spots generally occur on the head (McGuire 1996, Stebbins 2003, Mahrdt et al. 2010).

Taxonomic Relationships

Though it was described over a century ago, *Gambelia copeii* was not widely recognized as a distinct species until recently. Morphologically and genetically, *G. copeii* is similar to *G. wislizenii*, which led many authors either to consider the two as conspecifics or to recognize them at the subspecific level. McGuire (1996) provided a comprehensive systematic analysis of the Crotaphytidae (the family in which *Gambelia* is included) and argued for the recognition of *G. copeii* as a distinct species, based in large part on the presence of a narrow zone of sympatry between the two species in Baja California, Mexico. Following McGuire's monographic review, the species became widely accepted.

Phylogenetically, *G. copeii* appears to form a monophyletic group that is nested within *G. wislizenii* (McGuire et al. 2007), although this result is based on an analysis of mitochondrial data alone and requires further verification. Rates of potential gene flow and/or hybridization within the zone of sympatry have not been measured.

Life History

Little is known about the natural history of *Gambelia copeii*, and the limited information that is available comes from populations that occur farther south in Baja California, Mexico. We assume that the California populations are similar in most aspects of their life history to populations from the northern regions of Baja California.

Gambelia copeii emerges from hibernation as early as mid-March in northern Baja California, with adults remaining active at least until September (Grismer 2002). The breeding season begins in March or April and lasts at least until July (Fitch 1970, McGuire 1996, Grismer 2002, Goldberg et al. 2010). Grismer (2002) reported a single female in breeding coloration in August at the southern end of the species' range near Todos Santos, Baja California Sur, suggesting that the breeding season could extend much later in the north. Gravid females have been documented in both March and June, providing some evidence that *G. copeii* may produce multiple clutches in optimal years (Fitch 1970, Goldberg et al. 2010). In a sample of 10 museum specimens, the mean clutch size was 5 and did not appear to depend on female body size (Goldberg et al. 2010).

Gambelia copeii is primarily an ambush predator that preys upon other lizards, including whiptail lizards (*Aspidoscelis*), zebra-tailed lizards (*Callisaurus*), and side-blotched lizards (*Uta*), as well as arthropods (McGuire 1996, Grismer 2002).

Habitat Requirements

Gambelia copeii occurs across a wide latitudinal gradient and tolerates a variety of ecological conditions throughout its range. Little published information exists for California populations, although the species appears to prefer open habitat in mixed chaparral and sage scrub (R. Fisher, pers. comm., C. Mahrdt, pers.

comm.). In Baja California, the species occurs across a wider variety of habitat types, although this likely reflects habitat availability throughout the Baja California peninsula rather than specialization of California populations.

In northern Baja California, *G. copeii* occurs on mesas and foothills in scattered patches of chaparral and inland sage scrub with coarse sandy soils (C. Mahrdt, pers. comm.) and in an increasingly wide variety of habitat types farther south in Baja California (Grismer 2002). *Gambelia copeii* apparently prefers relatively open habitat throughout the diversity of plant communities in which it is found.

Distribution (Past and Present)

In California, *Gambelia copeii* is restricted to an approximately 70 km^2 area centered around Campo and Potrero Valleys in extreme southern San Diego County (Mahrdt et al. 2010; C. Mahrdt, pers. comm.). However, recent field surveys have failed to reconfirm this species at several sites in both Potrero and Campo Valleys, and the species may be locally extirpated at some of these sites particularly along the western edge of its range (R. Fisher, pers. comm.).

Outside of California, *G. copeii* occurs from the California border throughout much of the Baja California peninsula south at least as far as Todos Santos (Grismer 2002). Few data exist on changes in distribution, although agricultural expansion and development in northern Baja California are likely to cause declines (R. Fisher, pers. comm.).

Trends in Abundance

Few data exist regarding historical or present abundance in California. Unpublished pitfall trapping data collected over a 2-year period indicate that the species occurs at very low densities. Between March 1970 and December 1971, pitfall trapping at a 60 × 60 m study site 2.7 km northeast of Cameron Corners, San Diego County, California, yielded many captures of other lizard species in the area but only a single capture of *Gambelia* (C. Mahrdt, unpublished data). A second individual was captured near this site 3 years later (C. Mahrdt, unpublished data).

Nature and Degree of Threat

The principal threat facing *Gambelia copeii* is habitat loss due to development. The species is able to persist in a wide variety of habitats farther south, so long as the habitat remains relatively open and, presumably, abundant prey (primarily arthropods and smaller lizards) remains available. However, the species occurs at the extreme northern limit of its range in California, so even minor changes in environmental conditions could have large impacts here. Development, including habitat degradation and fragmentation, and climate-change-associated increases in wildfire frequency and intensity have the potential to cause these changes. Invasion of exotic grasses may also lead to further habitat degradation by reducing the availability of open habitat that this species prefers.

Status Determination

Gambelia copeii has an extremely small range in California, which makes it inherently sensitive to any declines. Ongoing habitat loss and potential impacts from climate change may negatively impact the species, but we have relatively few data to assess risk beyond these broad measures of sensitivity, so we refrain from assigning a priority score at this time.

Management Recommendations

Within its very limited California range, remaining large blocks of habitat require protection from further development to prevent future declines. In the absence of information to the contrary, we assume that grazing, wood clearing, and activities that might negatively impact the density of prey (including the presence of feral or pet cats) are all threats to *Gambelia copeii*. Frequent high-intensity wildfire should also be prevented, to the extent possible, within the species range.

Monitoring, Research, and Survey Needs

As no population density data are available, presence/absence surveys followed by mark–recapture monitoring programs should be undertaken throughout the species' range in California to establish baseline information. Loss of habitat across the United States–Mexico border has the potential to isolate the California populations. To begin studying the potential for this to occur, field studies of migration rates and patterns through disturbed and fragmented habitats should be conducted with the aim of identifying and protecting remaining habitat corridors, as well as characterizing this taxon's sensitivity to various sources of habitat disturbance. Such information will also be useful for developing models of the effects of future climate change scenarios on *Gambelia copeii*. Additional genetic data from nuclear markers should help confirm the species status of this taxon as well as quantify whether, and to what extent, hybridization occurs between it and *G. wislizenii*.

GILA MONSTER

Heloderma suspectum Cope 1869

PHOTOS: (top) Gila monster documented 29 May 1993 in the Kingston Mountains, San Bernardino County, California. Courtesy of Beth Behm. (bottom) Gila monster documented 7 May 2015 in the Mesquite Mountains, San Bernardino County, California. Courtesy of Barrett Scurlock.


118°0'0"W
116°0'0"W
GILA MONSTER
Heloderma suspectum
37°0'0"N
35°0'0"N
33°0'0"N
Museum Record
CNDDB, BIOS, or Contributor
Range
USDA Ecoregion
2,640,000

Status Summary

Heloderma suspectum is a Species of Special Concern, though we refrain from assigning it a priority status due to lack of information. The species received a Total Score/Total Possible of 60% (30/50) and was data deficient for several metrics. During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Among California lizards, *Heloderma suspectum* is virtually unmistakable. *Heloderma suspectum* is a large (22.8–35.5 cm SVL) stocky lizard with a dark ground color and distinctive pinkish, orange, or yellow patterning over the trunk and tail that forms bands or a reticulating network. This species possesses distinctive bead-like scales and large, strongly curved claws (Bogert and Martín del Campo 1956, Beck 2005). The ventral coloration is similar to the rest of the body, with alternating black and yellowish or pinkish bands that may form a reticulated pattern (Bogert and Martín del Campo 1956). Within its range, this species could only possibly be confused with the chuckwalla (*Sauromalus ater*), which sometimes develops a pinkish or yellowish coloration on top of a dark ground color but lacks the banded or reticulate patterning and does not have large, bead-like scales.

Gila Monster, Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	Data deficient
iii. Population concentration/migration (10)	0
iv. Endemism (10)	0
v. Ecological tolerance (10)	10
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	Data deficient
Total Score	30
Total Possible	50
Total Score/Total Possible	0.60

Taxonomic Relationships

Heloderma suspectum is one of two extant members of the family Helodermatidae. It is a close relative of the Mexican beaded lizard (*H. horridum*). The description of this species is generally attributed to Cope (1869), although it was actually depicted in print earlier by Baird (1859) using the name *H. horridum*. Cope's (1869) description is a one-paragraph secondhand summary; a far more complete description of the taxon is given by Bogert and Martín del Campo (1956) in their monographic treatment of the family Helodermatidae. The recognition of two species in the genus has not been questioned since the initial description. More recent molecular results confirm the distinctiveness of the two taxa (Douglas et al. 2010).

Two subspecies of *H. suspectum* have been described based on the pattern of reticulation (or lack thereof) in coloration. *Heloderma suspectum suspectum* has a reticulated color pattern, whereas *H. s. cinctum* has a banded pattern that largely lacks reticulations among the bands. A recent genetic survey of intraspecific variation found little evidence supporting these groupings. Additional data are needed to more fully examine intraspecific variation within this species (Douglas et al. 2010). All specimens known from California match the *H. s. cinctum* color pattern, with the single exception of an individual photographed near Piute Springs, San Bernardino County (see the "Distribution" section) (Lovich and Beaman 2007).

Life History

The life history of *Heloderma suspectum* has not been studied in California. Here we use data from other parts of the range (primarily Utah) and cautiously assume that the life history in California is similar.

Heloderma suspectum overwinters in burrows on rocky slopes adjacent to lower-elevation arroyos and bajadas (Beck 1990, Beck 2005). In California, it likely emerges in April or early May. The species spends nearly all of its time in underground burrows (>95% in Utah), emerging rarely to forage for food and to locate mates (Beck 1990). This species is a strict nest predator, preying on the nests of mammals, ground-nesting birds, and reptiles (Hensley 1949, Jones 1983, Beck 1990, reviewed by Beck 2005). *Heloderma suspectum* is venomous, although it is not known to use venom in subduing prey (Beck 2005). Rather the venom probably serves as a predator avoidance mechanism (Beck 2005).

In California, the daily activity pattern is not well characterized. Nocturnal activity has not been reported, although data are lacking. Reproduction likely occurs in April and May, with oviposition occurring shortly thereafter. Elsewhere in the range (Arizona), males leave their burrows and undertake relatively long (~1.6 km) walks to visit other burrows in search of females (Beck 2005). When males encounter each other during this period of activity, prolonged male–male combat may ensue. This behavior entails males entwining one another and attempting to pin one another to the ground (Beck 2005). The time required for eggs to hatch is poorly characterized, although young appear in the spring, which suggests that they overwinter in the burrow before dispersing. Elsewhere in the range, sexual maturity develops in 2–3 years, and adults are probably long-lived (>20 years) (Jennings 1984, Beck 2005). This species appears to be highly susceptible to water loss, which partially explains its relatively sedentary activity patterns (Beck 2005).

Habitat Requirements

Heloderma suspectum occupies a relatively wide variety of desert habitats throughout its range. In California, it is known primarily from a few desert mountain ranges in the eastern Mojave Desert. It inhabits rocky slopes, arroyos, bajadas, and washes, and is presumably limited on a larger scale by the availability of summer rainfall in the California deserts. Areas that are known to support this species receive a moderate amount of their total annual rainfall during the summer months (24% of the total), which is similar to the pattern in adjacent areas of Arizona that also support this species (39% of total; Lovich and Beaman 2007). On a more local scale, distribution may be controlled by the availability of relatively deep burrows, the presence of food, and availability of riparian or xeroriparian habitat (Lovich and Beaman 2007). Preferences for certain burrow conditions apparently exist but are poorly understood (Beck 2005). Individuals frequently return to specific burrows while leaving others, apparently suitable ones, unoccupied (Beck 2005). Adult Gila monsters are known to return to the same burrows year after year, showing remarkable homing ability and apparent knowledge of the location of many different burrows within their home range (Beck 2005). Too few records exist from the California portion of the range to form a thorough understanding of habitat requirements, although many records are associated with large and relative high mountain ranges as well as with riparian areas (Lovich and Beaman 2007)

Distribution (Past and Present)

Heloderma suspectum ranges from extreme southwestern Utah, through southern Nevada, southwestern Arizona, and south to Sinaloa, Mexico. In California, the species is known from 30 records in the Kingston, Providence, Clark, Piute, and Chocolate Mountain ranges (Bradley and Deacon 1966, De Lisle 1979, Ford 1981, Bicket 1982, De Lisle 1983, Ford 1983, Lovich and Beaman 2007, Ruppert 2010a, Ruppert 2010b, Lovich and Haxel 2011). Lovich and Beaman (2007) reviewed 26 records in California. Four additional records are now known. On 29 May 1993, a single adult *H. suspectum* was photographed on Smith Talc/Kingston Mountain road in the Kingston Mountains, Inyo County, California, approximately 24 km east of Tecopa (B. Behm, pers. comm.). The photographs show an animal with the banded pattern typical of other animals found in California (we include the clearest photograph here). An additional record comes from Vulcan Mine Road on the western side of the Providence Mountains on 2 May 2009. A natural history class from Cuesta College observed and photographed a single adult moving along the road (Sneed 2009, Ruppert 2010a, Ruppert 2010b). The most recent record that we are aware of from California was documented on 7 May 2015 in the Mesquite Mountains of California. A single adult animal was found resting under the partial shade of a cat's claw plant in a wash running parallel to Kingston Road (B. Scurlock, pers. comm). Lovich and Haxel (2011) report an additional credible sighting from Black Mountain in the southern Chocolate Mountains that occurred on 30 April 1974 as well as a second record from the same vicinity that is less well substantiated but may be credible. In addition, old records from the vicinity of Blythe, the Lower Colorado River in Imperial County, Chuckwalla Valley, and the Mojave River are in the literature but are less well substantiated than the more recent records (Woodson 1949, Funk 1966, Tinkham 1971, Lovich and Beaman 2007). The species may also occur in a few additional desert mountain ranges in California where records have not yet been recorded. In particular, the New York Mountains are a likely candidate for future records. These mountains lie between the Providence and Piute Mountains, both of which have records and contain what appears to be suitable *Heloderma* habitat. Other large and potentially suitable mountain ranges in the area include the Whipple Mountains, Turtle Mountains, Chemehuevi Mountains, and the Chuckwalla Mountains (Brown and Carmony 1991, Lovich and Beaman 2007).

Trends in Abundance

No data exist on the current or historical abundance of this taxon in California. Elsewhere in the range, the species exists in low densities (maximum recorded is ~10 individuals/km^2) (Beck 1985). Given the paucity of records in California, the species is likely more rare here than in the rest of the range.

Nature and Degree of Threat

The principal threats facing *Heloderma suspectum* in California are its small and extremely patchy distribution, coupled with the probable marginal habitat found in the state and presumed sensitivity to the effects of climate change. Further, we know virtually nothing about the ecology or population status of this species in California, so declines may occur that go undetected.

Status Determination

The almost complete lack of information on this taxon in the state, coupled with a life history that is potentially sensitive to changing climate, justifies designating this taxon as a Species of Special Concern. Because we have virtually no information about the magnitude of threat in this species, we refrain from assigning it a priority at this time.

Management Recommendations

Management recommendations are extremely difficult to formulate other than to protect habitat known to support this species from modification. Activities that might collapse or otherwise destroy burrows, including intense livestock grazing and mining activities, should be avoided in areas suspected of harboring *Heloderma suspectum* populations. Sightings of this infrequently encountered species should be submitted to the California Natural Diversity Database or other natural history databases (e.g., the LACM RASCals project, http://www.nhm.org/site/activities-programs/citizen-science/rascals).

Monitoring, Research, and Survey Needs

It may be impossible to study this species in the field in California because it is so rarely encountered. However, opportunities to do so should be pursued. Telemetric data, in particular, would be difficult to gather because this species is encountered so infrequently, but would also be an important step in enabling the collection of additional information about California populations. We recommend modeling the climate envelope capable of supporting *Heloderma suspectum* to help focus efforts for future surveys. After potential habitat patches have been identified, dawn and dusk surveys during the spring and following summer rain events probably have the best chance at identifying additional populations. A key priority for future sightings of this species is to collect nonlethal genetic samples that can then be compared to those collected from elsewhere in the range. These tissues will help to clarify intraspecific variation in the species and, if enough samples can eventually be collected, have the potential to supply information about distinctiveness and isolation of populations inhabiting different mountain ranges in the state.

COAST HORNED LIZARD

Phrynosoma blainvillii Gray 1839

Status Summary

Phrynosoma blainvillii is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 49% (54/110). During the previous evaluation, it was also considered a Species of Special Concern under the name *P. coronatum* (see the "Taxonomic Relationships" section) (Jennings and Hayes 1994a).

Identification

Phrynosoma blainvillii has the typical oval, flattened body form of a horned lizard and reaches a maximum SVL of 11.4 cm (Stebbins 2003). It has a row of large horns behind the head, with the two central horns usually longer than the rest and separated at their base. Two rows of large pointed fringe scales run down each side of the body. Large pointed scales also occur on the throat in two or three rows on each side. The dorsum of the body and tail have randomly scattered large, pointed, keeled scales. The general dorsal coloration is tan, yellowish, brown, reddish, or gray, with large dark blotches. Coloration can vary within and between populations and with respect to substrate color. Ventral coloration is cream, beige, or yellow, with dusky spotting (Stebbins 2003).

Coast Horned Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	0
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	7
v. Ecological tolerance (10)	7
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	7
Total Score	54
Total Possible	110
Total Score/Total Possible	0.49

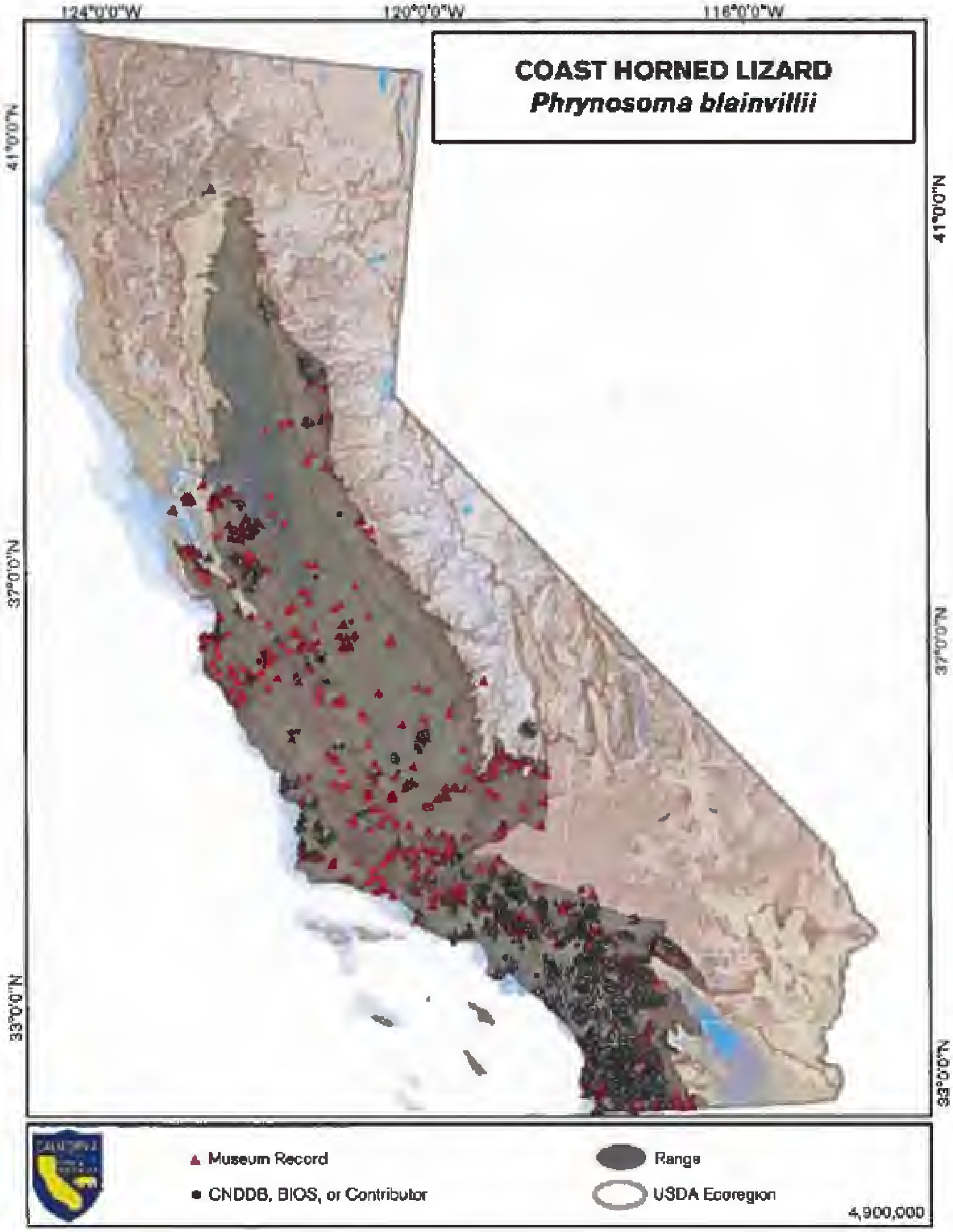


PHOTO ON PREVIOUS PAGE: Coast horned lizard, Kern County, California. Courtesy of Nicholas Hess.

Phrynosoma blainvillii may be confused with the desert horned lizard (*P. platyrhinos*) where the ranges of the two species meet in a small region of the southern and eastern part of the range of *P. blainvillii* in California. *Phrynosoma platyrhinos* is easily distinguishable based on a single row of fringe scales down each side of the body, a single row of pointed scales on either side of the throat, and smaller keeled scales on the dorsum.

Taxonomic Relationships

Phrynosoma blainvillii is a member of a species complex that has had a tumultuous taxonomic history, with several species and subspecies recognized by different researchers over time (Klauber 1936, Reeve 1952, Brattstrom 1997). During the previous Species of Special Concern evaluation (Jennings and Hayes 1994a), a single species, *P. coronatum*, was recognized, and California populations were considered as two subspecies: the California coast horned lizard (*P. c. frontale*) and the San Diego coast horned lizard (*P. c. blainvillii*). Recent studies on morphological, ecological, and genetic variation among populations support the recognition of only a single taxon in California, *P. blainvillii*, leading to a revised species-level taxonomy that restricts the species name *P. coronatum* to populations in Baja California Sur, Mexico (Montanucci 2004, Leaché et al. 2009). Three clades have been identified in California based on mitochondrial DNA: northern Baja California, southern California, and northern California (Leaché et al. 2009; see "Distribution" trend). However, two nuclear loci did not distinguish among the clades in California, and ecological and morphological data show substantial overlap among the clades (Montanucci 2004, Leaché et al. 2009). Therefore, we do not recognize any of these clades as conservation units at this time.

Life History

Phrynosoma blainvillii adults are typically active in California from February to November, with peak activity between April and July (Banta and Morafka 1968, Hager and Brattstrom 1997, Fisher et al. 2002, Alberts et al. 2004, Gerson 2011). Hatchlings are active from mid to late summer into November (Banta and Morafka 1968, Hager 1996, Hager and Brattstrom 1997, Fisher et al. 2002, Alberts et al. 2004). Diurnal activity switches from midday peaks in the spring to more crepuscular activity in summer and early fall (Heath 1965, Hager and Brattstrom 1997).

Most information on reproduction has been collected in the southern part of the range in California. Goldberg (1983) looked at reproductive condition in 164 specimens collected mostly from March to September in Los Angeles, Riverside, San Bernardino, San Diego, Ventura, and Riverside Counties. Reproductive activity occurred from March to June, with females commonly ovipositing in May. Clutch sizes usually average around 11–12 eggs (Stebbins 1954, Howard 1974, Pianka and Parker 1975, Goldberg 1983). Goldberg (1983) reported that a single female appeared to be yolking a second clutch, suggesting the possibility for multiple clutches per year in this species, though how common this may be is unknown. In northern Baja California and southern California, males have spermatozoa present from April until early June (Howard 1974), and oviposition occurs from late May to July with an incubation period of about 60 days (Howard 1974, Pianka and Parker 1975). Montanucci (1968) observed mating in the field as late as May in Merced County. Howard (1974) observed 25 mm SVL hatchlings in late July and early August in northern Baja California. These animals had attained sizes averaging 42 mm SVL by October. First-year males emerged from winter dormancy at ~51 mm SVL. Animals in this population were sexually mature around 75 mm SVL (Howard 1974). Pianka and Parker (1975) reported minimum female SVL at maturity as 73 mm in Baja California and southern California. Goldberg (1983) reported that the smallest mature males were 62 mm SVL, and the smallest females were 73 mm SVL in southern California.

Annual adult survival estimates from radio-tracked animals in Riverside County were roughly twice as high for males as females: males 62% (95%, CI 42–81%) and females 34% (95%, CI 15–53%) (estimates assume animals of unknown fate are dead; Alberts et al. 2004). Most deaths were due to predation (31% birds, 23% snakes), followed by road mortality (15%), with the rest due to unknown causes (Alberts et al. 2004). Average home range size varied from 1.9 to 4.0 ha across habitat types, with smaller ranges and lower activity levels observed during a drought year (Alberts et al. 2004).

Surface activity is determined partly by temperature. Adults in a Riverside County population had field active body temperatures ranging from 13.3°C to 39.4°C (mean 34.5°C), and hatchlings had a narrower range of temperatures ranging from 21.1°C to 41.1°C (mean 34.4°C) (Alberts et al. 2004). Animals were not active when ground surface temperatures were below 19.4°C or above 57.3°C (Alberts et al. 2004). Gerson (2011) reported capturing lizards when surface temperatures were up to 63°C in a Merced County population. Pianka and Parker (1975) reported a mean field active body temperature for 15 animals of 36.7°C. The critical thermal minima and maxima are −3°C and 46.7°C, respectively (Brattstrom 1965).

Ants can make up 90% of prey items and 45% of prey volume in stomach contents (n = 214; Pianka and Parker 1975), although many other insect prey are also consumed depending on availability (Stebbins 1954, Miller and Stebbins 1964, Alberts et al. 2004). About half of the prey found in scat was *Pogonomyrmex* ants (*P. rugosus* and *P. californicus*) (Riverside County; Alberts et al. 2004). Other ant prey and non-ant insects were taken as well. In Merced County, every scat examined contained beetles, but not every scat contained ants, suggesting less reliance on ant prey in this area (M. Gerson, unpublished data). See the "Nature and Degree of Threat" section for effects of nonnative ants.

Habitat Requirements

Phrynosoma blainvillii is found in a variety of habitat types, including sage scrub, dunes, alluvial scrub, annual grassland, chaparral, oak woodland, riparian woodland, Joshua tree woodland, coniferous forest, and saltbush scrub (Grinnell and Grinnell 1907, Klauber 1939, Stebbins 1954, Banta and Morafka 1968, Montanucci 1968, Tollestrup 1981, Hager and Brattstrom 1997). However, microhabitat preferences are much narrower. *Phrynosoma blainvillii* needs loose, fine soils for burrowing, open areas for thermoregulation, and shrub cover for refugia (Jennings and Hayes 1994a). In undisturbed sage scrub habitat in Riverside County, animals preferred leafy plant species with relatively dense foliage for cover, overwintering, and aestivation (Alberts et al. 2004). In the absence of shrubs, *P. blainvillii* may rely instead upon California kangaroo rat (*Dipodomys californicus*) burrows for refugia (Shedd et al. 2011). In a mark–recapture study in San Bernardino and Riverside Counties, Hager and Brattstrom (1997) observed *P. blainvillii* in the open 64% of the time, in the shade of vegetation 14% of the time, next to vegetation 7% of the time, and in rodent burrows 5% of the time.

Pitfall trapping at 21 sites in 4 counties in southern California revealed that within sites, *P. blainvillii* abundance was positively correlated with the presence of organic soils and chaparral vegetation and negatively associated with nonnative Argentine ant (*Linepithema humile*) presence (Fisher et al. 2002). At a larger scale, the abundance of *P. blainvillii* between sites was positively associated with the presence of native ants and chaparral vegetation and negatively associated with canopy height. Similar to patterns in abundance, *P. blainvillii* presence was positively associated with sandy soils and chaparral vegetation and negatively associated with Argentine ant presence.

Distribution (Past and Present)

Phrynosoma blainvillii occurs from northern Baja California north along the coast, continuing into the Central Valley and Coast Range, and east to the Sierra Nevada foothills and the western edge of the Mojave Desert (Leaché et al. 2009). The southern and northern California clades (see the "Taxonomic Relationships" section) roughly correspond in range to the previously recognized subspecies *Phrynosoma coronatum blainvillii* and *P. c. frontale*, respectively. The northern Baja California clade extends from Ensenada, Mexico, north into San Diego County. The southern California clade slightly overlaps with the northern Baja California clade in San Diego County and continues north to the Los Angeles Basin and east to the San Gabriel Mountains and the edge of the Mojave Desert. A third group, the northern California clade, comprises the rest of the range in California, from the Los Angeles basin north through the Central Valley and Coast Ranges.

Historically, this species occurred in California from an isolated record in Shasta County in the north, south along the edges of the Sacramento Valley, through much of the south Coast Ranges, the San Joaquin Valley, the Sierra Nevada foothills, south along the coast to the Mexican border, and throughout the Transverse and Peninsular Ranges, ending along the western edge of the desert slope (Jennings 1988c). Recent field observations in the NAFHA database document this species at Kennedy Meadows in Tulare County; further information about the status here is needed. Jennings and Hayes (1994a) estimated that *P. blainvillii* has disappeared from 35% of its historical range in northern California and from 45% of its historical range in southern California. Remaining populations in the northern end of its range in the Coast Range and in the Sierra Nevada foothills from Butte County to Fresno County are highly disjunct (Jennings and Hayes 1994a; J. Shedd, pers. comm.).

Trends in Abundance

Declines in the early decades of the twentieth century were partly due to collecting for the curio trade in the Los Angeles basin. Jennings (1987) estimated that at least 115,000 *Phrynosoma blainvillii* were harvested over a 45-year period, with substantial collecting ending around the 1930s. Due to collecting, lizards were noted as being scarce or absent in many areas where they had formerly been abundant (Grinnell and Grinnell 1907, Bryant 1911, Van Denburgh 1922). Agriculture and development has led to declines in more recent decades (see the "Nature and Degree of Threat" section).

It is very difficult to estimate population sizes for horned lizards because their cryptic coloration and behavior make them difficult to detect. In sage scrub habitat in Riverside County, *P. blainvillii* density was estimated as 3–4 adults/km of road transect traveled and 1.1–4.2 adults/ha, with a total of 402 lizards (adults and juveniles) captured over 5 years (Alberts et al. 2004). In Merced County, Gerson (2011) captured 145 individuals (adults and juveniles) on 2.4 ha of transect over an 8-month period, roughly 60 lizards/ha. Lizards were patchily distributed at this site, and transects were purposefully placed in areas with high lizard abundance (M. Gerson, pers. comm.). The sites in both studies experienced controlled burns and grazing and supported a mix of native and introduced plants (Alberts et al. 2004, Gerson 2011).

Nature and Degree of Threat

Major threats to *Phrynosoma blainvillii* include urbanization, agriculture, off-highway vehicles, flood control structures, energy development, and nonnative Argentine ants (Grinnell and Grinnell 1907, Montanucci 1968, Jennings 1987, Jennings and Hayes 1994a; J. Shedd, pers. comm.). These threats may be more pronounced in the southern part of the range (S. Sweet, pers. comm.). Leatherman (1996) observed a single *P. blainvillii* that had apparently died from getting its horns stuck in an erosion control blanket. Introduced Argentine ants have displaced native ant prey over parts of central and southern California and appear to be spreading largely as a commensal with human development (Ward 1987, Holway 1995, Holway 1998). In choice tests, lizards preferred native ants to Argentine ants, and Argentine ants were not detected in field-collected scat, suggesting that they are not commonly taken as prey (Suarez et al. 2000). In the laboratory, growth rates were lower for animals raised experimentally on Argentine ant diets relative to native diets (Suarez and Case 2002). However, lizards will shift their diets to include more non-ant prey in Argentine ant-invaded areas (Suarez et al. 2000).

The effects of wildfire on *P. blainvillii* are complex and only beginning to be studied. In southern California, capture rates increased by about 30% in chaparral habitat a few years post-fire compared to unburned reference plots (Rochester et al. 2010). No changes were detected in coastal sage scrub habitat, though both habitat types lost substantial vegetative cover. The positive response to fire in chaparral was likely due to the creation of open habitat and the fact that ant prey communities appeared to be unaffected (Rochester et al. 2010). However, the proportion of plots occupied in chaparral habitat decreased in response to fire, possibly due to direct mortality effects of fire. Population increases in burned areas were hypothesized to be due to recolonization from unburned refugia. If so, then the timing and distribution of fire across the landscape would affect how lizards are able to respond and whether the net effect of fire on populations is positive or negative. Additionally, monitoring for this study detected very few *P. blainvillii* in grassland habitats. Because repeated or high-intensity fires can lead to conversion of shrubland to grassland, this also represents a potential threat.

Under climate change, the probability of large (>200 ha) fires and area burned is expected to increase in the northern coastal part of the range and the Sierran foothills, and be largely unchanged in the Central Valley (Fried et al. 2004, Lenihan et al. 2008, Westerling and Bryant 2008). In the southern part of the range where wildfire is common, there is little consensus on future fire dynamics because of the difficulty in modeling Santa Ana weather events (Westerling et al. 2004, Westerling and Bryant 2008). Land use in the Central Valley is predominantly agricultural; thus, habitat availability is likely to remain low in this area. Elsewhere in the range, large decreases are expected in shrubland with concomitant increases in grassland (Lenihan et al. 2008, PRBO 2011).

Status Determination

Documented extirpations and declines in this species, coupled with a moderate ecological sensitivity, justify a Priority 2 Species of Special Concern status.

Management Recommendations

Protecting remaining populations from further habitat loss and disturbance is the most important management strategy for this species. The presence/absence and abundance of *Phrynosoma blainvillii* appears to be determined by local, rather than regional-scale factors, so management strategies should focus on protecting local populations (Fisher et al. 2002). Because they tend to rely on crypsis rather than speed for protection, they may be particularly sensitive to land uses that increase the likelihood of animals being crushed or killed, including off-highway vehicle use and grazing. Preventing the spread of Argentine ants into *P. blainvillii* habitat is difficult but also important for the persistence of the species. Given that Argentine ants prefer moist microhabitats, xeric landscaping and reducing artificial surface water may be beneficial for native ants and horned lizards in developed areas.

Monitoring, Research, and Survey Needs

Existing populations should be monitored to determine trends in population abundance. An important research question is the extent to which small habitat fragments, on the order of a few hectares or less, can support viable populations of this lizard. Given the high human population density in much of its range, the effects of human commensal predators, including raccoons, skunks, ravens, and domestic cats should be studied, with control measures implemented as feasible. Continued work on the effects of Argentine ants, including follow-up studies on shifts in lizard diet after their long-term establishment, would provide valuable information on whether *Phrynosoma blainvillii* can adjust to this widespread invasive ant. More research is needed on the effects of introduced plants, which may increase cover, affect native ant prey, and influence thermoregulation and locomotion (Germano et al. 2001, Alberts et al. 2004, Newbold 2005, Rieder et al. 2010). Grazing and fire can have positive effects by maintaining open habitat and negative effects by facilitating the spread of invasives or through direct mortality (Kimball and Schiffman 2003, Alberts et al. 2004, HilleRisLambers et al. 2010). The effects of cattle grazing on *P. blainvillii* need more study. Cattle and other grazers may help maintain open habitats that are favorable to *P. blainvillii* but also may degrade habitat through soil compaction. The net effect of grazing and fire as management strategies requires more study and likely needs to be determined at the site scale. The effects of wildfire on *P. blainvillii* should continue to be studied, particularly given the uncertainty concerning future fire dynamics in the southern part of the range.

FLAT-TAILED HORNED LIZARD

Phrynosoma mcallii (Hallowell 1852)

Status Summary

Phrynosoma mcallii is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 57% (63/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Like other horned lizards, *Phrynosoma mcallii* has a round body and is dorsoventrally flattened. It is readily distinguished from other horned lizards by a dark middorsal stripe (Smith 1946). The two largest horns behind the head are long and thin, the tail is broad and flat, and two rows of lateral spines run down each side of the body. The limbs are long and thin relative to other horned lizards. The dorsum is cryptically colored, ranging from pale cream to a light rusty brown, and the ventral surface is white and unmarked. Adults can be as large as 8.7 cm SVL (Boundy and Balgooyen 1988, McGrann et al. 2006), but 6.5–8.0 cm SVL is more typical.

Phrynosoma mcallii co-occurs in narrow sympatry with the desert horned lizard (*P. platyrhinos*) along the Salton Trough in California (Stebbins 2003). The two species are easily distinguished because *P. platyrhinos* has a single

Flat-Tailed Horned Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	3
v. Ecological tolerance (10)	10
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	7
Total Score	63
Total Possible	110
Total Score/Total Possible	0.57

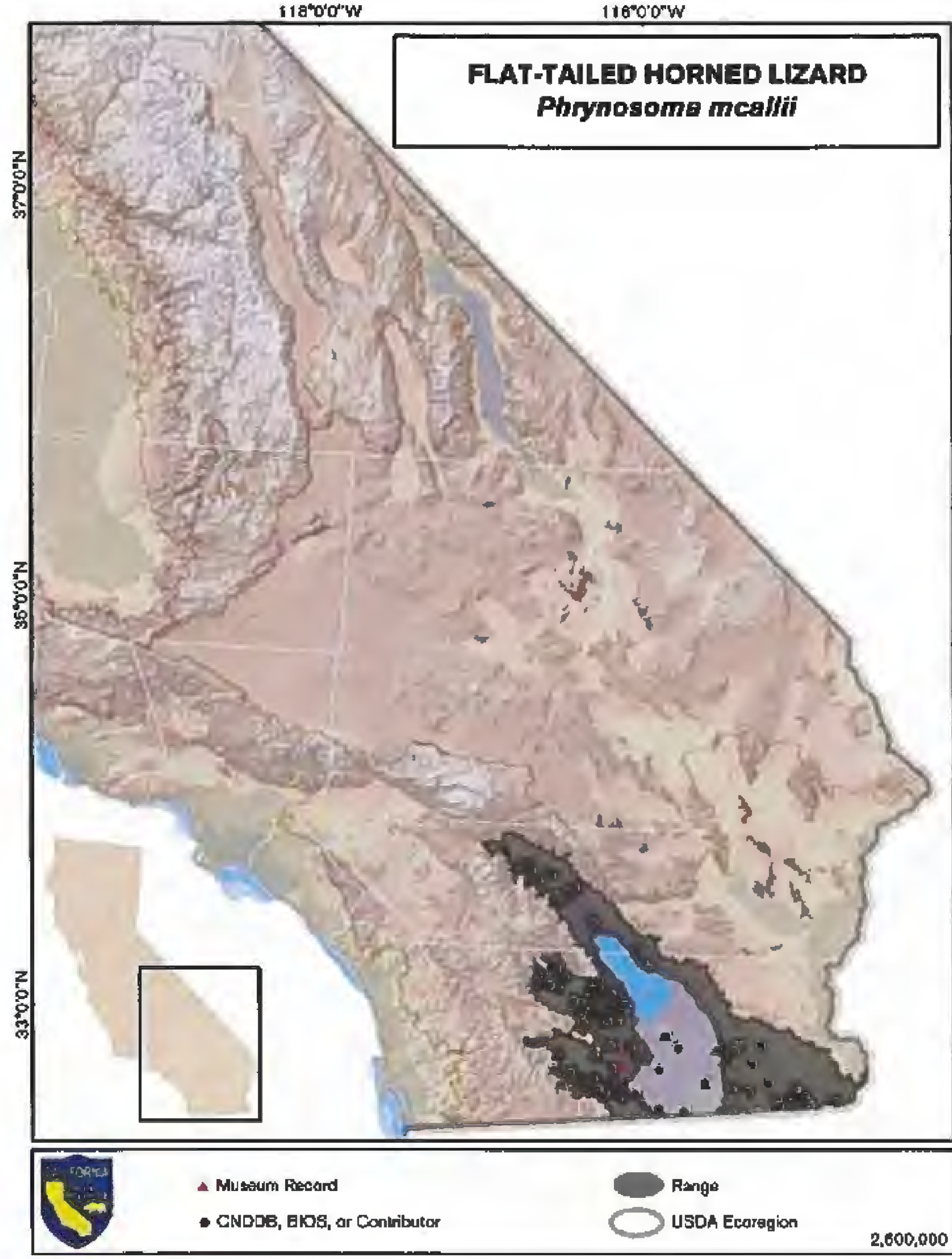


PHOTO ON PREVIOUS PAGE: Flat-tailed horned lizard, Sonora, Mexico. Courtesy of Rob Lovich.

row of lateral spines, shorter horns on the head, and lacks a dark middorsal stripe (Smith 1946, Stebbins 2003). Morphologically intermediate animals thought to be hybrids have been observed near Ocotillo, California (Stebbins 2003), and near Yuma, Arizona (Young 2010).

Taxonomic Relationships

Mulcahy et al. (2006) conducted a phylogeographic study of *Phrynosoma mcallii* and *P. platyrhinos*. They identified management units for *P. mcallii* on either side of the Colorado River. Populations west of the Imperial Valley were historically connected but are now fragmented by human development. The Coachella Valley population, in particular, appears to be highly isolated (Mulcahy et al. 2006).

Life History

Phrynosoma mcallii is generally most active in the summer and inactive during the winter, although there is some flexibility in their winter dormancy behavior. Adult activity in the Coachella Valley in Riverside County peaked from June to August, with little or no activity observed from November to February (Barrows and Allen 2009). At sites in San Diego and Imperial Counties, adults entered hibernation burrows from early October to late December, and smaller animals entered dormancy later than larger animals (Grant and Doherty 2006). The average onset of winter dormancy occurred in mid-November in Imperial County and lasted for an average of 89 days (range 14–138 days), with most animals emerging in mid-February (Muth and Fisher 1992). Radiotelemetry studies have shown that not all individuals enter this distinct period of dormancy (Muth and Fisher 1992, Wone and Beauchamp 2003, Grant and Doherty 2006). Juveniles have been observed surface-active on warm days in December, suggesting that winter dormancy behavior may be more flexible in juveniles compared to adults (Grant and Doherty 2006). Burrows at sites in Imperial and San Diego Counties were 6 cm deep on average (range 2–17; Muth and Fisher 1992, Grant and Doherty 2006). Summer burrows in Yuma, Arizona, were 25–30 cm deep and 70–80 cm long (Young and Young 2000).

Daily activity patterns shift seasonally (Mayhew 1968, Wone and Beauchamp 2003). At Ocotillo Wells State Vehicular Recreation Area (Imperial and San Diego Counties), *P. mcallii* was active throughout the day in spring and fall but showed a bimodal daily activity pattern in the summer (Wone and Beauchamp 2003). *Phrynosoma mcallii* was out in the open during the early morning but retreated under shrub cover by 10:00 a.m. (Wone and Beauchamp 2003). When substrate temperatures exceeded 49°C, lizards entered burrows and reemerged in the evening when substrate temperatures dropped below 47°C (Wone and Beauchamp 2003). Norris (1949) also reported animals retreating between 10:00 and 11:00 a.m. in Riverside County during July. In outdoor enclosures, Heath (1965) observed shade-seeking behavior when body temperatures averaged 40°C and emergence from shade when mean body temperatures were 34.9°C. Brattstrom (1965) recorded lizards at temperatures ranging from 29.3°C to 41.0°C.

Breeding activity has been observed in the field from early May through the end of August (Setser 2004, Barrows and Allen 2009, Young 2010). Adults emerge from winter dormancy in reproductive condition, with testes at maximum size in males and enlarged yolked follicles present in females (Howard 1974). Eggs are laid in burrows dug by the lizards (Setser 2004) and can be deposited from 14 to 90 cm deep, depending on soil moisture (Setser 2004, Young 2010). Clutch sizes range from 2 to 10 eggs, with the average typically around 5 (Norris 1949, Stebbins 1954, Howard 1974, Pianka and Parker 1975, Setser 2004, Young 2010).

Under good conditions, *P. mcallii* can breed early in the season, young can attain adult size rapidly and breed in their first year, and two clutches per season are possible (Howard 1974, Turner and Medica 1982, Muth and Fisher 1992, Barrows and Allen 2009, Young 2010). In multiple-clutch years, the first cohort emerges in late July or early August at 35–37 mm SVL (Howard 1974, Turner and Medica 1982, Muth and Fisher 1992). These hatchlings may be capable of reproducing in their first spring because they can reach near adult sizes before entering winter dormancy (Howard 1974, Muth and Fisher 1992). The second cohort emerges in late August or early September (Howard 1974, Turner and Medica 1982). However, these animals are only ~38 mm SVL in October and may not reach sexual maturity until another season of growth has occurred (Howard 1974, Muth and Fisher 1992). Working in Yuma, Arizona, Young (2010) observed that hatchlings and yearlings did not attain adult size by the following summer under drought conditions, but in wet years animals attained adult size within six months.

Males usually have larger home ranges than females, and home ranges tend to be larger in wet compared to dry years (Wone and Beauchamp 2003, Setser 2004, Young 2010). Radiotelemetry studies at the Ocotillo Wells State Vehicular Recreation Area found average male home range sizes of 1.8–2.4 ha and female home ranges of 0.9–1.3 ha (Wone and Beauchamp 2003, Setser 2004). Setser (2004) observed high site fidelity, with few lizards shifting their range centers outside of the home range used in the previous year. Near Yuma, Arizona, average male home range size varied from 2.5 ha (males) and 1.3 ha (females) in a very dry year to 10.5 ha (males) and 1.9 ha (females) in a very wet year (Young and Young 2000). In wet years at the Yuma, Arizona, site, maximum mean daily movements were 200–700 m, compared to only 50–100 m in drier years (Young 2010).

Survivorship has been measured in a few populations using radiotelemetry and mark–recapture methods. At Ocotillo Wells State Vehicular Recreation Area, adult yearly survivorship was estimated as approximately 50% over a 2-year study period (Setser 2004). This high survivorship rate was attributed to the scarcity of ground squirrel predators, with only 5–8% of radio-tagged lizards lost to predation (Setser 2004). Similarly, adult survivorship over 2 years in Imperial County was approximately 50%, with half of known mortalities due to ground squirrel predation (Muth and Fisher 1992). In contrast, 39% (21/54) of radio-tagged *P. mcallii* succumbed to predation at the Yuma, Arizona, site in 1 year, with most deaths attributable to ground squirrels (Young 2010). Predation rates were only 10% in another year at this site (Young 2010). Survivorship of hatchlings over their first year was greater than 50% across multiple cohorts in Coachella Valley (Barrows and Allen 2009). Survivorship declined in subsequent years to less than 20% for 2-year-olds and less than 5% for 3-year-olds (Barrows and Allen 2009). Such low survivorship beyond the first year suggests that early maturity and multiple clutches may be key to positive growth of populations in the Coachella Valley (Barrows and Allen 2009).

Phrynosoma mcallii is a dietary specialist on ants, particularly native harvester ant species. Ants typically make up over 90% of prey items in stomach content and scat analyses (Pianka and Parker 1975, Turner and Medica 1982, Young 2010). In 106 specimens examined by Pianka and Parker (1975), 97% of prey items were ants. While at least 11 species of ants have been identified from scats, ants from the genera *Pogonomyrmex* and *Messor* are most commonly taken (Turner and Medica 1982). Near Yuma, Arizona, ants (mostly genus *Pogonomyrmex*) constituted 99% of prey items, with a few beetles taken as well (Young 2010).

Habitat Requirements

In California, *Phrynosoma mcallii* occurs in several Sonoran Desert habitat types, including sandy areas (flats, hills, and valleys), salt flats, badlands, and gravelly areas (Stebbins 2003, Turner and Medica 1982). While they may prefer areas with a layer of fine, wind-blown sand, *P. mcallii* also occur on substrates ranging from hard-packed soils to sand dunes and mud hills (e.g., Beauchamp et al. 1998, Muth and Fisher 1992). For example, at Ocotillo Wells State Vehicular Recreation Area, a site where sandy habitats are highly disturbed by off-highway vehicle use, *P. mcallii* abundance was highest in sparsely vegetated gravel and mud hills in less-disturbed areas (Beauchamp et al. 1998).

In the Coachella Valley, *P. mcallii* were 2–6 times more abundant on stabilized sand fields than on active dunes (Barrows and Allen 2009) and were not observed in ephemeral sand fields or stable dune habitats (Barrows and Allen 2010). A reduction in windblown sand over the last few decades due to climatic factors and disturbance is thought to be responsible for the apparent absence of *P. mcallii* from ephemeral sand fields, habitats that still support *P. platyrhinos* populations (Barrows and Allen 2010). Lizards selected moderately compacted sands in both stabilized sand fields and active dunes, and this habitat feature may be important for maintaining the integrity of burrows while still being loose enough for digging (Barrows and Allen 2009).

Distribution (Past and Present)

Phrynosoma mcallii is a desert animal with the smallest range of any *Phrynosoma* species that occurs in the United States (Stebbins 2003). It is found from the Coachella Valley in Riverside County south into extreme northeast Baja California and northwest Sonora, Mexico, and east to the extreme southwest corner of Arizona (Stebbins 2003). The species typically occurs below 230 m elevation, but has been found as high as 520 m (FTHL ICC 2003, Rorabaugh and Young 2009, Turner et al. 1980). The currently occupied range is patchily distributed within the historical range. In California, these areas are the Coachella Valley, west of the Salton Sea and the Imperial Valley, and east of the Salton Sea and the Imperial Valley on the west side of the Colorado River (Mulcahy et al. 2006).

The Flat-tailed Horned Lizard Interagency Coordinating Committee estimated that nearly half of the entire range of *P. mcallii* has been altered by human activities, with 39–43% of the historical habitat in the United States converted to agriculture, urban areas, or other uses (reviewed in FTHL ICC 2003). The historical range of *P. mcallii* in California has been estimated at 700,000–900,000 ha, mostly in Imperial County but including parts of eastern San Diego and central Riverside Counties (reviewed in FTHL ICC 2003). Of this historical range, the Flat-tailed Horned Lizard Interagency Coordinating Committee further estimated that 400,000 ha of habitat remain in California (FTHL ICC 2003).

Within the Coachella Valley, Barrows et al. (2008) used niche models to estimate that 83–92% of historically occupied habitat has been lost to development, agriculture, fragmentation, or disruption of windblown sand transport processes. Of the estimated 33,500 ha of historically available suitable habitat, 2600 ha of potential habitat remain in the valley, of which only 1400 ha is currently occupied (Barrows et al. 2008).

Trends in Abundance

Phrynosoma mcallii has long been regarded as a relatively rare species (e.g., Klauber 1939). Density is very difficult to estimate for this cryptic species, and earlier estimates were based on scat counting methods that are no longer thought to be reliable (see the "Monitoring, Research, and Survey Needs" section). Despite these difficulties, dramatic declines have been documented in some areas (Turner and Medica 1982).

Populations of *P. mcallii* appear to naturally fluctuate in abundance, and the drivers of these dynamics are beginning to be explored. In the Coachella Valley, the population declined by about 50% per year during 2002–2005, resulting in an overall decline of 90% (Barrows and Allen 2009). However, in the following 2 years, *P. mcallii* abundance rebounded to half of the 2002 levels (Barrows and Allen 2009). Unlike some other desert species, abundance was not correlated with year-to-year variation in rainfall (Barrows and Allen 2010). Instead, increased rainfall was negatively correlated with the abundance of ant prey and positively associated with increased soil compaction (Barrows and Allen 2009). Other studies have also found associations between *P. mcallii* abundance and ant abundance (e.g., Turner and Medica 1982, Rorabaugh et al. 1987).

Mark–recapture studies have generated minimum density estimates of approximately 1.1/ha at Ocotillo Wells State Vehicular Recreation Area (Setser 2004) to 6.1/ha in Imperial County (Turner and Medica 1982). Increasingly sophisticated statistical methods for estimating abundance have been employed to compensate for the low abundance and cryptic nature of *P. mcallii*. Grant and Doherty (2007) working in Imperial County estimated densities from 0.41 to 1.55 lizards/ha at different sites, using methods that explicitly account for detection probability (see the "Monitoring, Research, and Survey Needs" section).

Nature and Degree of Threat

Habitat loss and fragmentation due to urban development and agriculture have been the major threats faced by *Phrynosoma mcallii* populations in California, with future threats anticipated due to renewable energy development. *Phrynosoma mcallii* are particularly sensitive to such disturbances because they are ecological specialists, and their ability to recover from population declines through reproductive responses is highly dependent upon favorable environmental conditions.

Phrynosoma mcallii is negatively impacted by fragmentation, and edge effects can extend several hundred meters into undisturbed habitat (Young and Young 2005, Barrows et al. 2006). Based on surveys of lizard tracks, Barrows et al. (2006) found that *P. mcalllii* in the Thousand Palms Oasis Preserve in Coachella Valley, Riverside County, experienced negative edge effects along the desert/suburban boundary of the preserve. *Phrynosoma mcallii* were at low abundance within 150 m of the edge compared to farther into the preserve. The mechanism behind the negative effect was hypothesized to be mortality due to roads and subsidized predators such as shrikes and kestrels. Bird predators were positively associated with suburban edge habitats because of increased availability of trees and poles for perching compared to desert habitat. There was no edge effect on native harvester ant abundance and nonnative ants were not detected, so the edge effect is probably not due to impacts on prey availability (Barrows et al. 2006). In another study in Yuma, Arizona, 90% of shrike-killed *P. mcallii* were within 10 m of a road (Young 2010). Nonnative ants and plants may also pose a threat to *P. mcallii* (see the "Monitoring, Research, and Survey Needs" section). Wind and solar development may be of particular concern in western Imperial County and east of the Imperial Sand Dunes, while geothermal development may threaten populations inside of the Ocotillo Wells State Vehicular Recreation Area (J. Weigand, pers. comm.). Such development may lead to habitat degradation and loss, as well as increased fragmentation.

Off-highway vehicle use has long been suspected of negatively impacting *P. mcallii* populations through direct effects such as mortality and indirect effects on habitat quality. Within protected Management Areas (see the "Management Recommendations" section), off-highway vehicle use is restricted to designated areas (Grant and Doherty 2009). Outside of these protected areas, approximately 100,000 ha of remaining habitat may be subject to off-highway vehicle activity, an area encompassing more than a quarter of remaining habitat in California (Grant and Doherty 2009). Grant and Doherty (2009) experimentally tested the hypothesis that off-highway vehicles crush dormant *P. mcallii* by controlled rides over radio-tagged animals in burrows. None of the animals in their study died or were injured, suggesting that direct effects on animals in burrows may be weak. Direct mortality of surface-active *P. mcallii* due to off-highway vehicle activity has been reported anecdotally from some sites (e.g., Turner and Medica 1982, Muth and Fisher 1992). McGrann et al. (2006) found that lizard body mass, but not density, was higher on sites with low off-highway vehicle impact compared to high-impact areas. The density of ant mounds (i.e., prey) was also higher in low-impact sites, supporting the possibility of indirect effects of off-highway vehicle use on *P. mcallii* (McGrann et al. 2006).

Aside from increases in temperature, there is little consensus as to how climate change will affect the Sonoran Desert region of California where *P. mcallii* occurs. Mean annual temperatures are expected to increase, with 22 additional extremely hot days per year (where temperatures exceed the long-term 95th percentile) and 10 fewer days below 0°C predicted (Bell et al. 2004). High temperatures may limit surface activity, whereas warmer, shorter winters may increase opportunities for growth and reproduction. Estimates of changes in rainfall range from modest increases in mean annual rainfall up to 45% decreases (reviewed in PRBO 2011). This uncertainty in how precipitation will change makes it difficult to predict how *P. mcallii* will be affected. The effect of rainfall timing and magnitude on *P. mcallii* populations is likely complex, as drought reduces juvenile growth rate and adult movement, but wet years reduce prey abundance (see the "Life History" section). How fire dynamics will change in this area is also highly uncertain (Westerling and Bryant 2008). Little change is expected in vegetation communities (Lenihan et al. 2008, Stralberg et al. 2009).

Status Determination

The specialized diet of *Phrynosoma mcallii*, its low reproductive rates, and small geographic range in a highly fragmented region of California contribute to a Priority 2 Species of Special Concern status.

At the federal level, efforts to secure range-wide protection for *P. mcallii* have been underway for several years, with the species first identified as a candidate for listing under the federal Endangered Species Act in 1982 (reviewed in USFWS 2011a). In 1989, this lizard was rejected for listing under the California Endangered Species Act. Following these efforts, several state and federal agencies comprising the Flat-tailed Horned Lizard Interagency Coordinating Committee signed a voluntary conservation agreement, which resulted in the protection of management and research areas and a plan for monitoring the species (Foreman 1997). In 2003, the range-wide management strategy was updated, providing reviews of biology, threats, and management recommendations for *P. mcallii* (FTHL ICC 2003). In 2011, *P. mcallii* was again denied federal protection under the Endangered Species Act (USFWS 2011a). In broad terms, the USFWS concluded that the threats to *P. mcallii* that initiated consideration for listing have been largely addressed by management efforts (USFWS 2011a). However, *P. mcallii* populations continue to face a variety of threats throughout their range in California.

Management Recommendations

The main management actions that can support *Phrynosoma mcallii* populations are those that limit habitat disturbance and destruction. Development that leads to habitat conversion or fragmentation should be avoided or limited in *P. mcallii* habitat. Renewable energy projects should consider potential negative impacts on *P. mcallii*. Limiting off-highway vehicle use to the overwintering season when animals are less likely to be surface-active may help limit direct mortality impacts. Roadside barriers and crossing structures should be investigated to reduce road mortality in areas where roads may be barriers to population connectivity. The use of pesticides in or near *P. mcallii* habitat should consider potential negative impacts on native ant prey that are an important determinant of habitat quality for this species. Habitat corridors should be established or maintained to promote connectivity among remaining populations, particularly across the United States–Mexico border. Assisted migration may be important for ensuring gene flow across obstacles such as fences along the United States–Mexico border. Restoration of degraded habitats could include activities such as manipulating soil properties, removing or controlling nonnative plants, and replanting of native plant species that provide food for harvester ants and open habitat for *P. mcallii*.

The Flat-tailed Horned Lizard Interagency Coordinating Committee has implemented a management strategy for *P. mcallii* (FTHL ICC 2003). In California, this strategy includes the establishment of three Management Areas and one Research Area encompassing roughly 170,000 ha in regions of California deemed especially important to the species including the Borrego Badlands, West Mesa, East Mesa, and Ocotillo Wells. The conservation and management of these areas is described in the FHTL ICC (2003) document, and we refer the reader there for additional details. There is currently no management area in the northwestern portion of the range. However, the Coachella Valley Multiple Species Habitat Conservation Plan and Natural Communities Conservation Plan will protect approximately 44% of remaining habitat in Coachella Valley (FTHL ICC 2003).

Monitoring, Research, and Survey Needs

Monitoring *Phrynosoma mcallii* is difficult because this species is cryptic, population abundance fluctuates, and densities are often low. This results in low detection probabilities overall and a high degree of variation in detection probability with respect to different observers, habitats, substrates, and seasons. *Phrynosoma mcallii* is cryptically colored and also exhibits cryptic behavior, tending to freeze and/or bury itself in the sand instead of fleeing (Bryant 1911). Young (2010) observed that radio-tagged individuals in Arizona were almost always motionless when approached, but tracks showed that the animals ran 1–2 m to reach the cover of twigs or vegetation, then froze to avoid detection. Over 25% of the time, fleeing individuals also shuffled into the sand (Young 2010). Such crypsis results in a strong effect of observer experience on survey success (Grant and Doherty 2007).

To deal with these challenges, researchers have tried to use statistical methods to explicitly incorporate detection probability (the probability of seeing lizards if they are present) into mark–recapture estimates of population abundance (e.g., Grant and Doherty 2007, Royle and Young 2008). Detection probabilities ranged from 0.06 to 0.15 (Young 2010) to as high as 0.52 on sandy plots intensively searched by experienced observers (Young and Royle 2005). Range-wide monitoring by members of the FTHL ICC from 2005 to 2012 yielded detection probabilities ranging from 0.15 in the Borrego Badlands to 0.79 in the Yuma Desert (R. Lovich, pers. comm.), and these monitoring efforts are ongoing. Even with increasingly sophisticated mark–recapture analyses, data collection requires substantial effort, and abundance estimates will always be plagued by low detection probabilities. Because of these challenges, distinguishing population declines from natural fluctuations in abundance is difficult, unless declines are severe. As an alternative, Young (2010) recommended monitoring presence/absence over large areas using scat surveys.

Scat counts were commonly used into the 1990s to estimate abundance, but their reliability for measuring density has subsequently been questioned (e.g., Muth and Fisher 1992, Beauchamp et al. 1998). However, scats have been shown to be a good indicator of *P. mcallii* presence, at least in areas where congeners are absent (Young and Royle 2005). If scats are present on a 0.75 ha plot, there is a >99% probability of an observer detecting them within an hour (Young and Royle 2005). Young (2010) proposed that such scat surveys could be useful for delineating occupied habitat across large areas and that monitoring changes in site occupancy over time might be a more viable monitoring strategy than trying to estimate abundance.

In addition to improved monitoring strategies, other research needs include determining the effects of introduced species, the design and efficacy of road-crossing structures, and landscape genetic studies of population connectivity. An additional important research problem is to identify and monitor processes that reduce the abundance of ant prey and/or affect sand compaction (Barrows and Allen 2009). Monitoring for the spread of Argentine ants, which have been shown to negatively impact *P. blainvillii*, may also be warranted, especially along suburban–desert boundaries. Argentine ants have invaded the Coachella Valley but to date are not known to have moved into *P. mcallii* habitat (Barrows et al. 2006). Fire ants may also pose a threat to *P. mcallii* (J. Weigand, pers. comm.), and their spread and potential impacts should be studied. Nonnative plant species are suspected to negatively impact horned lizards by reducing the availability of open habitat and seed-producing plants and by impacting locomotion (Germano et al. 2001, Newbold 2005, Barrows et al. 2009, Rieder et al. 2010). Introduced plants such as tall-growing or Sahara mustard (*Brassica tournefortii*) now occur in *P. mcallii* habitat, and the effects of these species require further study (J. Shedd, pers. comm., Barrows 2012). Barrier fences that prevent lizard access to roads have been successful in Yuma, Arizona (e.g., Gardner et al. 2004), and may be beneficial in targeted areas in California. However, more research is needed into crossing structure design and siting to prevent further fragmentation of populations. Finally, a clearer understanding of the extent of habitat fragmentation (using both genetic and mark–recapture methods) and how it affects population viability is an important research need.

COLORADO DESERT FRINGE-TOED LIZARD

Uma notata Baird 1858

Status Summary

Uma notata is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 58% (64/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Uma notata is a medium-sized lizard (7.0–12.2 cm SVL) with a moderately flattened body, a countersunk lower jaw, keeled labial scales, projecting row of pointed scales on the toes, eyelids, and ear openings that form a fringe (Cope 1894, Heifetz 1941, Stebbins 1954, Stebbins 2003). The dorsal color pattern consists of light pale yellow to cream ocelli, with dark or reddish centers over a dark ground color (Van Denburgh 1922, Stebbins 1954, Stebbins 2003). These ocelli tend to form broken lengthwise lines at the shoulders (Heifetz 1941). The dark dorsal coloration fades to reddish brown on the head and legs (Van Denburgh 1922). The undersurface is white, with prominent dark ventrolateral spots or bars on the underside of the tail and narrow diagonal lines on the underside of the throat (Stebbins 2003). An orange or pinkish stripe occurs along the lower flanks and becomes more prominent during the breeding season (Stebbins 1954).

Colorado Desert Fringe-Toed Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	0
v. Ecological tolerance (10)	7
vi. Population trend (25)	15
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	64
Total Possible	110
Total Score/Total Possible	0.58

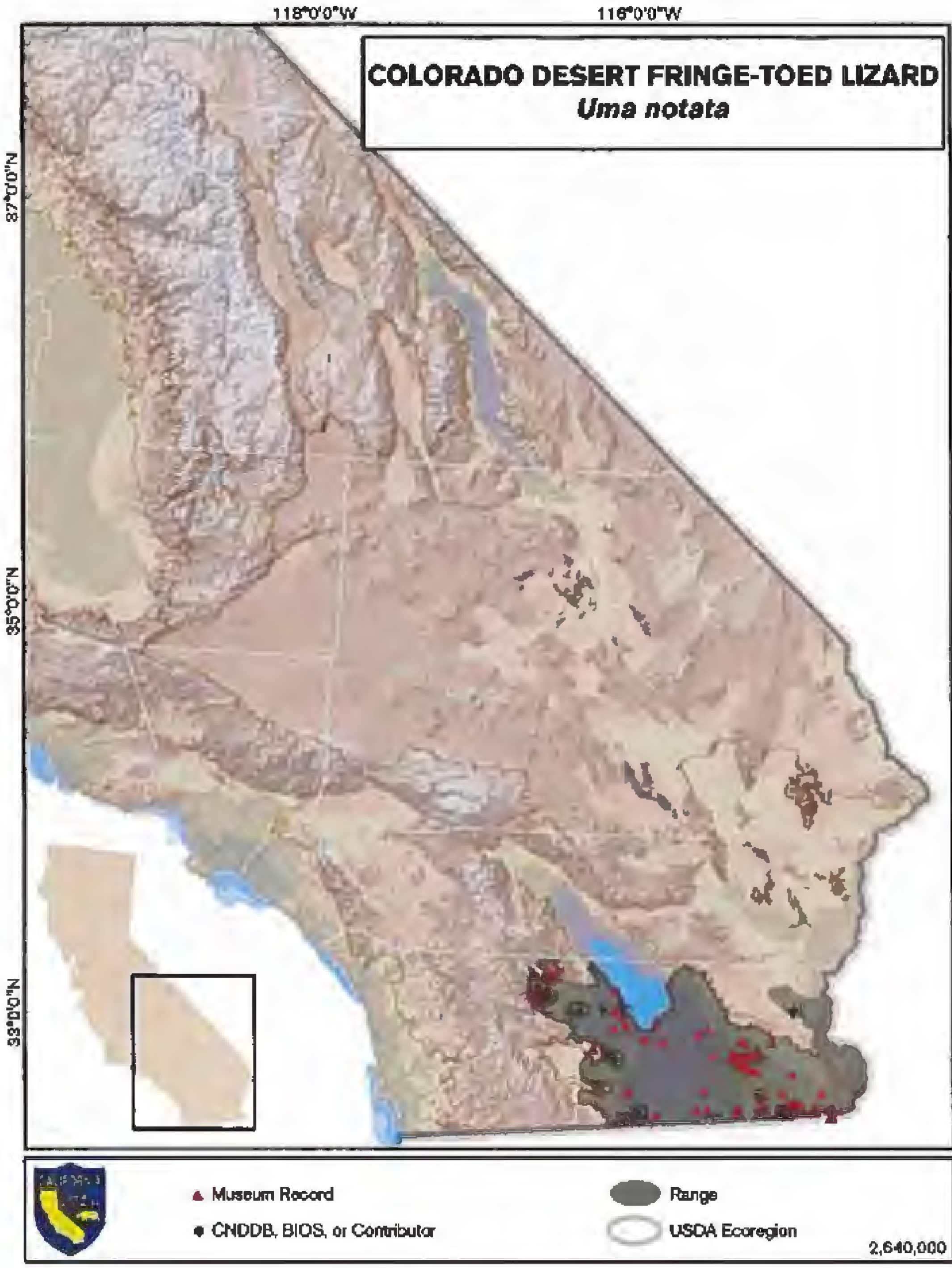


PHOTO ON PREVIOUS PAGE: Colorado Desert fringe-toed lizard, Imperial County, California. Courtesy of Adam Clause.

Orange coloration may also be present around the eye.

Uma notata can easily be confused with its congeners in California, the Coachella Valley fringe-toed lizard (*U. inornata*) and the Mojave fringe-toed lizard (*U. scoparia*), although none of these species have overlapping ranges. *Uma inornata* lacks the large and prominent blotches on the ventral surface, although small black spots may be present (Stebbins 2003). *Uma scoparia* usually has narrow lines on the throat that form chevrons and has dorsal ocelli that do not form broken lines on the shoulders (Stebbins 2003). The sympatric zebra-tailed lizard (*Callisaurus draconoides*) also has black bars on the tail, although these form bands that encircle the tail rather than being present only on the underside. *Callisaurus* also lacks fringes on both the toes and the ear openings and has an overall slimmer body shape (Stebbins 2003).

Taxonomic Relationships

The taxonomy of the fringe-toed lizards has been confusing since their original description and remains somewhat controversial. *Uma notata* was initially described from a single preserved juvenile specimen in poor condition (Baird 1858). The initial description of morphology was inadequate to diagnose the taxon and provided details on coloration specific to the poorly preserved specimen ("light pea green, spotted with darker green") and an inaccurate type locality ("Mojave Desert"). An expanded description was later provided by Cope (1894, 1895b), which helped clarify the distinctiveness of the taxon. Heifetz (1941) provided a thorough morphological analysis of the genus and concluded that *U. notata* should be treated as a species separate from the other two California species (*U. inornata* and *U. scoparia*). However, these three species are closely related and their treatment in the literature has shifted between subspecies (of *U. notata*) and full species (Stebbins 1954, Norris 1958, Mayhew 1964a, Mayhew 1964b, Adest 1977, Zalusky et al. 1980).

In addition, some authors recognize two subspecies within *U. notata*. *Uma notata rufopunctata* (Cope 1895b) ranges through Arizona and northwestern mainland Mexico, while *U. n. notata* is present only in California. Analyses of mitochondrial data suggest that these two subspecies do not form a monophyletic group. Rather, *U. n. notata* is sister to *U. inornata* to the exclusion of *U. n. rufopunctata* (Wilgenbusch and De Queiroz 2000, Trépanier and Murphy 2001). Trépanier and Murphy (2001) noted that the mitochondrial DNA implied either that *U. inornata* should be considered part of *U. notata* or that *U. n. notata* should be elevated to a full species and that *U. n. rufopunctata* contains two species (one of which is cryptic and had not previously been recognized). They preferred this latter arrangement, although this has not been formally presented to date. Here, we treat *U. notata* as a full species, separate from *U. n. rufopunctata*. Further genetic analyses using multiple independent sequence markers are needed to clarify these species boundaries, as well as the phylogenetic relationships among species and subspecies.

Life History

To the extent that it has been studied, the life history of *U. notata* is essentially identical to that of *U. scoparia*. This species specializes on fine windblown sand habitats and possesses several behavioral, morphological, and physiological adaptations allowing it to do so (see account for *U. scoparia*). This species has a yearly activity cycle that is similar to *U. scoparia*, becoming surface-active as early as February, breeding between April and July, with egg laying in May–July and young appearing in September (Stebbins 1954, Grismer 2002, Stebbins 2003). The two species also exhibit similar daily activity patterns and behavior. They are known to differ in the pattern and cadence of "pushups" used in territorial displays, which may have served as a behavioral isolating mechanism (Carpenter 1963). *Uma notata* has a generalized diet composed of leaves, flowers, seeds, and a variety of small arthropods that is similar to the diet of

U. scoparia (Stebbins 1944). See the account for *U. scoparia* for additional details.

Habitat Requirements

To the extent that they have been studied, habitat requirements are identical to those of *Uma scoparia* and are described in that species' account.

Distribution (Past and Present)

Uma notata ranges from the southeastern corner of California north and west to the Salton Sea and the northeastern corner of San Diego County. Outside of California, it ranges farther south into Baja California, Mexico, to a latitude roughly parallel with the mouth of the Colorado River (Jennings and Hayes 1994a, Grismer 2002). The species' known elevational range extends from 74 m below to 180 m above sea level (Jennings and Hayes 1994a, Stebbins 2003).

Few distributional declines have been documented, although we presume that they have occurred in some areas that have been heavily impacted by off-highway vehicular use, as well as in areas that have experienced heavy development (see the "Trends in Abundance" section). In particular, agricultural development has eliminated habitat in extensive areas around the Salton Sea (Jennings and Hayes 1994a).

Trends in Abundance

Few data regarding historical *Uma notata* population densities exist, although survey data strongly suggest that ongoing declines are occurring in areas that experience off-highway vehicle use. Luckenbach and Bury (1983) conducted surveys in paired plots at the Algodones Dunes (Imperial County, California) that had or had not experienced off-highway vehicle disturbance. *Uma notata* abundance on off-highway vehicle-impacted plots was significantly lower than nonimpacted areas.

Nature and Degree of Threat

Uma notata is experiencing many of the same threats as *U. scoparia*. Habitat loss due to off-highway vehicle damage and habitat destruction due to human activities is the greatest immediate concern. Luckenbach and Bury (1983) demonstrated major decreases in abundance from off-highway vehicle use due to direct mortality and decreasing vegetation density and quality. Off-highway vehicle use in *Uma* habitat also causes increased rates of tail loss and hearing loss, neither of which are fatal but both of which decrease individual fitness (Brattstrom and Bondello 1983, Luckenbach and Bury 1983). Climate change models for this region predict relatively sharp increases in mean temperature of up to 2°C. The impact of such increases on *U. notata* is not known but should be a high priority for future research. Other threats include increasing predation associated with human commensals and the more general problems associated with reduced population size and fragmentation. See the *U. scoparia* account for additional discussion.

Status Determination

Uma notata specializes on a habitat which is uncommon, patchy, and undergoing significant degradation, and this is the primary justification for this Priority 2 designation. Several populations of this species appear to be stable, and some of the habitat occurs on protected land; thus, a higher-priority designation is not currently justified.

Management Recommendations

The primary management need for *Uma notata* is habitat protection. Protecting sand dune habitat from the impact of off-highway vehicle use alone will significantly increase the probability of long-term survival of this species in California. Habitat conversion for housing, agriculture, and solar/wind energy may all have strongly detrimental effects on *U. notata*, and the limited distribution of the species requires that impacts be reviewed on a project-by-project basis. Over the longer term, increasing temperature and potentially decreased precipitation due to climate change (PRBO 2011) could also lead to habitat loss, which may require the development of additional management actions.

Given their strong association with windblown sand habitats, all species of *Uma* may be subject to local extirpations with limited opportunities for natural recolonization, and human-mediated gene flow may be necessary to maintain such populations.

Monitoring, Research, and Survey Needs

The monitoring needs for *Uma notata* are essentially identical to those of *U. scoparia*. Overall, less of *U. notata*'s range occurs on protected land, so these monitoring efforts (and accompanying habitat protection) are needed more urgently for this taxon than for *U. scoparia*. The impact that habitat modification may have on *U. notata* populations is an area in need of additional study. Two genetic needs are critical. First, the species boundaries of *Uma*, including the distinctiveness of the subspecies of *U. n. rufopunctata* and the resolution of the number and identity of species contained within the genus, require a multi-locus nuclear dataset to complement initial work using mitochondrial DNA (Trépanier and Murphy 2001). Second, landscape genetic analyses quantifying the extent of past and current gene flow among isolated or semi-isolated populations are needed to better understand how to manage landscapes and have the least possible impact on metapopulation dynamics and future population viability.

MOJAVE FRINGE-TOED LIZARD

Uma scoparia Cope 1894

Status Summary

Uma scoparia is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 55% (61/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Uma scoparia is a medium-sized lizard (7.0–11.4 cm SVL) with a moderately flattened body, a countersunk lower jaw, keeled labial scales, a projecting row of pointed scales on the toes, eyelids, and ear openings that form a fringe (Cope 1894, Stebbins 1954). The dorsal ground coloration is black and is heavily covered, with a pattern of white or tan ocelli with blackish to reddish centers that do not form lines over the shoulders (Cope 1894, Heifetz 1941, Jennings and Hayes 1994a, Stebbins 2003). This dark coloration fades to brown or tan on the head, limbs, and tail. The light dorsal coloration tends to vary among populations and usually matches the color of the sand in the vicinity (Miller and Stebbins 1964). The ventral surface is white, with two prominent black spots on either side of the body (some populations have an additional set of preanal spots) and black bars along the underside of the tail (Heifetz 1941). The

Mojave Fringe-Toed Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	10
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	7
v. Ecological tolerance (10)	7
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	61
Total Possible	110
Total Score/Total Possible	0.55

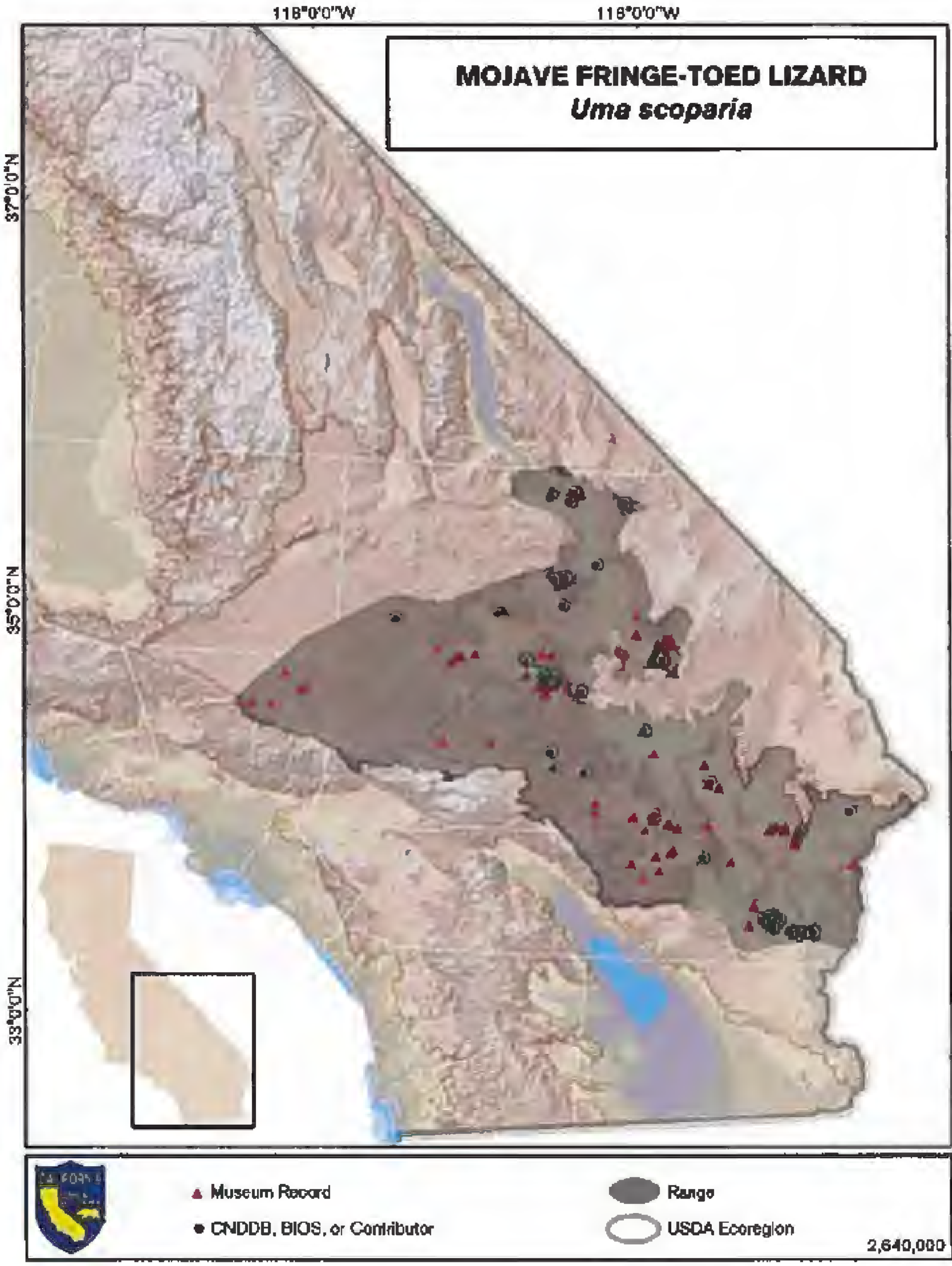


PHOTO ON PREVIOUS PAGE: Mojave fringe-toed lizard, San Bernardino County, California. Courtesy of Luke Mahler.

throat is marked with narrow crescent-shaped black bars (Cope 1895b, Heifetz 1941, Stebbins 2003). During the breeding season, a yellow-green wash may develop on the ventral surface and fade into pink on the sides (Stebbins 2003).

This species could be confused with its congeners, the Coachella Valley fringe-toed lizard (*U. inornata*) and the Colorado Desert fringe-toed lizard (*U. notata*). *Uma inornata* has greatly reduced, or lacks altogether, the conspicuous black spots on the sides of the belly and has ocelli that tend to form lines over the shoulders. *Uma notata* usually has diagonal lines on the throat rather than crescent-shaped lines and has ocelli that tend to form lines over the shoulders (Stebbins 2003). These three species do not overlap in range, although *U. scoparia* is broadly sympatric with the zebra-tailed lizard (*C. draconoides*), with which it also might be confused. *Callisaurus draconoides* lacks fringe scales on the ear openings and toes, has an overall slimmer body shape, and has black bands that form rings around the distal portion of the tail rather than only being on the tail underside (Stebbins 2003).

Taxonomic Relationships

Uma scoparia was initially described on the basis of femoral pore counts and several scalation characters (Cope 1894, Cope 1895b). It was later placed in synonymy with *U. notata* when several of Cope's diagnostic characters were reinterpreted as representing individual variation rather than species differences (Camp 1916b, Van Denburgh 1922). The taxon was later resurrected to full species status based on a larger series of specimens that identified diagnostic morphological differences among the taxa (Heifetz 1941). Several different authors have noted external morphological, osteological, and genetic similarity among members of the genus and have variously treated *U. scoparia* as a full species or subspecies of *U. notata* (Stebbins 1954, Norris 1958, Mayhew 1964a, Mayhew 1964b, Adest 1977, Zalusky et al. 1980). Carpenter (1963) showed that the pattern of push-up behavior used in territorial displays was distinct in *U. scoparia*, compared to *U. inornata* and *U. notata*, and suggested that this may serve as an isolating mechanism.

Phylogenetic analyses of mitochondrial data suggested that *U. scoparia* is monophyletic (Trépanier and Murphy 2001, Murphy et al. 2006) and forms a clade with the other Mojave and Sonoran Desert taxa (*U. inornata* and *U. notata*) (Wilgenbusch and De Queiroz 2000). Mitochondrial data also suggest that some haplotype diversity occurs within the *U. scoparia* (Murphy et al. 2006), although divergences are low and additional, multigene nuclear data are needed to clarify intraspecific variation. Populations occurring in the northern part of the range have been proposed as a distinct population segment based on mitochondrial phylogeography and presumed isolation (Murphy et al. 2006).

Life History

Uma scoparia is an active, wary, diurnal lizard that specializes on fine windblown sand habitat. It is extremely similar in most aspects of life history to other species in the genus (Stebbins 1944), and here we make use of life history information from these other species when it is not available for *U. scoparia*. Species in the genus *Uma* all possess a number of morphological, behavioral, and physiological adaptations that allow them to persist in arid habitats. Specifically, a countersunk lower jaw, nasal valves, and fringes on the eyes and ear openings allow *U. scoparia* to prevent sand from entering the body (Norris 1958). The nasal passages have a complex convoluted shape that reduces moisture loss and excludes sand from inhalation (Stebbins 1943, Stebbins 1948). Enlarged fringes on the toes have been experimentally shown to increase both maximum velocity and acceleration on fine sand, particularly on steeply sloped landscapes such as are often found in sand dunes (Carothers 1986). The flattened body form, wedge-shaped head, enlarged, keeled scales on the head, limbs and toes, and the smooth granular scales over the rest of the body aid in burrowing and "sand-swimming" behavior (Stebbins 1944). *Uma scoparia* employs this behavior both to escape from predators and to take refuge from extremely hot surface conditions (typically when surface temperature exceeds 43°C; Norris 1958). *Uma scoparia* possesses both acute vision and hearing, which aid in predator avoidance and prey capture (Stebbins 1944).

Adult *U. scoparia* overwinter in the sand between November and February, then become surface-active throughout the day as temperatures allow. The species maintains a mean body temperature of 36–37.5°C, often becoming inactive during the hottest part of the day during midsummer (Mayhew 1964b, Miller and Stebbins 1964). Breeding occurs throughout the spring and summer between April and July, and females lay clutches of 1–5 eggs (usually 2 or 3); more than one clutch may be produced in optimal years (Stebbins 1954, Mayhew 1966, Fromer et al. 1983, Stebbins 2003). Young begin to appear on the surface in September (Miller and Stebbins 1964).

Uma scoparia has a generalized diet that includes a variety of beetles, ants, wasps, flies, and other small arthropods, as well as plant leaves and seeds (Stebbins 1944). At Dale Dry Lake, San Bernardino County, the diet of adult *U. scoparia* consisted of approximately 60% plant material (mainly in the form of small seeds) and 40% small arthropods (Minnich and Shoemaker 1972). The juvenile diet, conversely, was composed of over 90% arthropods (Minnich and Shoemaker 1972). In low rainfall years, adults may be forced to switch to a diet composed mostly of arthropods due to lack of vegetation, and this may be suboptimal (Barrows 2006). The quality of available food is probably dependent on the local rainfall, which varies widely from year to year throughout the species' range. Barrows (2006) found that a regression model including rainfall and diet explained 92% of the variation in *U. inornata* density and that population sizes could approach zero during multiyear droughts and then quickly rebound when average rainfall resumed.

Habitat Requirements

Uma scoparia lives exclusively on fine wind-blown sand (Stebbins 1944). Habitat where lizards are found in the highest abundances generally consists of relatively sparse creosote scrub on loose sand dunes. The diameter of individual sand grains in these areas is usually <0.5 mm. Areas with large sand grains (>2 mm in diameter) appear to be avoided, presumably because this impedes sand swimming and burying behavior (Stebbins 1944, Norris 1958, Fromer et al. 1983). Within appropriate habitat, individuals select areas with the finest sand available (often the downwind side of vegetation and slopes) (Stebbins 1944, Norris 1958). Some vegetation is probably required for food and shade (Miller and Stebbins 1964). The species is not present in areas where the sand becomes too firmly packed to allow for sand swimming, and washes and desert flats are generally unsuitable (Miller and Stebbins 1964). No evidence exists that *Uma* will enter these areas to migrate between adjacent areas of suitable habitat, although additional study of this question would be valuable.

Uma scoparia may require relatively large habitat patches for long-term persistence. Population modeling in the ecologically similar *U. inornata* suggests that plot sizes smaller than 100–200 ha are unlikely to allow long-term persistence of isolated populations (Chen et al. 2006).

Distribution (Past and Present)

Uma scoparia is patchily distributed throughout much of the Mojave Desert in California. The range extends from near the southern end of Death Valley at the Inyo–San Bernadino County line south through San Bernardino and Riverside Counties, extending west narrowly into Los Angeles County (Van Denburgh 1922, Norris 1958, Miller and Stebbins 1964, Pough

1974, Jennings and Hayes 1994a, Stebbins 2003). Norris (1958) reports a record from Inyo County, which has often been repeated in the literature. However, the stated locality "one and one-half miles southeast of Saratoga Springs" places this record in San Bernardino County, and we know of no other confirmed records from Inyo County. This species is nearly endemic to California, extending into Arizona in one small area near Parker, Yuma County (Pough 1974). A single report of possible *Uma* tracks reported from the Eureka Sand Dunes, Inyo County, California, would extend the known range ~175 km to the northwest and requires verification (Bolster et al. 2000). The known elevational range extends from below sea level to nearly 1000 m (Jennings and Hayes 1994a).

Extirpations have been documented at El Mirage and Harper Dry Lakes, San Bernardino County, and at Lovejoy Buttes and Piute Butte, Los Angeles County (Murphy et al. 2006). Additional extirpations may have occurred at Rogers Dry Lake, Kern County, California, and Saddleback Butte, Los Angeles County, California (CBD 2006).

Trends in Abundance

No quantitative data are available regarding historical abundance, though the lizard was, and is, common at many isolated localities. Some data suggest that this species has become uncommon in areas where habitat degradation due to off-highway vehicle use has occurred (Bolster et al. 2000, CBD 2006).

Nature and Degree of Threat

The most important threats facing *Uma scoparia* are habitat loss and fragmentation due to human activities and off-highway vehicle use, which negatively impacts loose sand habitat. Other activities, including the development of renewable energy facilities, may also negatively impact the structure of essential windblown sand habitat patches. The species is only found in loose sand areas, and experimental work in the closely related and ecologically similar species *U. inornata* suggests that these lizards are highly sensitive to stabilization of their sand habitat (Turner et al. 1984). Habitat fragmentation is also an important threat. Even where patches of intact habitat remain, fragmentation and small patch sizes have been shown to be associated with declines and extirpations in *U. inornata* (Barrows and Allen 2007). In addition, surveys for the ecologically similar *U. notata* that compared lizard abundances in areas that experienced off-highway vehicle use to areas that do not, found much higher densities in the less-impacted habitat (Luckenbach and Bury 1983). Off-highway vehicles impact this species through direct mortality, destruction of vegetation (which is correlated with lizard abundance), and increased rates of tail loss (Luckenbach and Bury 1983, Ouren et al. 2007). Further, *U. scoparia* has sensitive hearing that is easily damaged by even moderate and short duration off-highway vehicle activity (Brattstrom and Bondello 1983). Hearing loss likely harms this lizard's efficiency at capturing prey and its ability to avoid predation (Brattstrom and Bondello 1983). Increasing predator densities (e.g., common ravens) in certain areas, often in association with human development and the presence of garbage dumps, may also be causing declines in lizard abundance in localized areas (Bolster et al. 2000).

Uma scoparia is likely sensitive to the effects of climate change. Climate change models for this region predict relatively sharp increases in mean temperature of up to 2°C (PRBO 2011). The impact of such increases on *U. scoparia* and on critical plant species is not known but could be large and should be a high priority for future research. The distribution of *U. inornata* is associated with an east-to-west drought gradient in the Coachella Valley (Barrows and Allen 2007). Like off-highway vehicle use, drought decreases the amount and quality of vegetation present, which limits both food and cover for this species (Barrows et al. 2010).

Because *Uma* specializes on relatively isolated patches of habitat, it is probably unable to track available habitat with changing climatic conditions. Climate change modeling studies on other *Uma* species (*U. inornata*; the Coahuila fringe-toed lizard, *U. exsul*; and the Chihuahuan fringe-toed lizard, *U. paraphygas*) predict significant habitat loss under a relatively wide range of climate change scenarios (Ballesteros-Barrera et al. 2007, Barrows et al. 2010) and these results are also likely to apply to *U. scoparia*.

Status Determination

Uma's specialized habitat is relatively uncommon and undergoing significant degradation, and this is the primary justification for Priority 3 designation. While some populations have been extirpated, several populations of this species are still common, and some habitat occurs on protected land that is not subject to off-highway vehicle use, precluding the need for a higher-priority designation.

On 10 April 2006, the Center for Biological Diversity and Sylvia Papadakos-Morafka petitioned the US Department of the Interior to list the northern population segment identified by Murphy et al. (2006) under the Federal Endangered Species Act (CBD 2006). The USFWS issued a 90-day finding that substantial evidence for listing need had been presented and initiated a 12-month status review for the taxon (USFWS 2008). This review concluded that the Amargosa River populations of *U. scoparia* do not constitute a distinct population segment and are therefore ineligible for listing under the US Endangered Species Act (USFWS 2011).

Management Recommendations

Effective management of this taxon over the short term can likely be accomplished by protecting habitat from development and degradation from off-highway vehicles and other human impacts. Over longer time periods, climate change could begin to have a larger impact, and this may require additional management efforts. Such efforts could range from human-assisted translocation to planting drought-resistant vegetation, depending on local conditions and the extent of temperature and precipitation changes. If restoration occurs in areas where extirpation has occurred or if development activities further isolate occupied habitat patches, human-assisted translocation, potentially in association with captive breeding programs, may be a key strategy for this species.

Monitoring, Research and Survey Needs

Two key research efforts for *Uma scoparia* should focus on the effects of human activities (including off-highway vehicles, solar and wind energy development, and roads) and the genetic effects of both natural and anthropogenic habitat fragmentation. The effects of off-highway vehicles are particularly important, and monitoring efforts should be initiated in areas that experience off-highway vehicle use compared to more pristine, adjacent areas. In particular, these efforts should focus on comparing the effect of varying intensity of anthropogenic disturbance on populations, with the aim of establishing what intensity of off-highway vehicle use can be tolerated. These efforts should also attempt to disentangle the effects of habitat destruction, noise pollution, and direct mortality on populations, since each can in principle be managed independently. For example, if off-highway vehicle use primarily affects these lizards through reductions in vegetation, habitat restoration coupled with restricting off-highway vehicles to certain trails or corridors could constitute a reasonable management strategy. Alternatively, noise pollution effects may require eliminating off-highway vehicle access in areas where the lizards are present. Because population sizes naturally fluctuate with rainfall in this species (Barrows 2006), and in some cases can approach zero before rebounding, monitoring this species is inherently difficult, and multiyear surveys spanning several drought and non-drought years are essential. The frequency of lizard detection and the accuracy of population size estimates can be increased with repeated sampling and specific detection methods (Turner et al. 1984, Bolster et al. 2000), and these should form the foundation of monitoring protocols.

Genetic studies are critical at two distinct levels of resolution that require different datasets and analytical approaches. Species boundaries across *Uma* and large-scale phylogeographic patterns within *U. scoparia* remain poorly resolved (see also the species account for *U. notata*), and both are critical for effective management. For species boundary work, the resolution of the number and identity of species contained within the genus requires a multilocus nuclear dataset to complement initial work using mitochondrial DNA (Trépanier and Murphy 2001). Within *U. scoparia*, phylogeographic studies using multiple nuclear markers are also needed in order to quantify the intraspecific diversity present within the species. At a finer scale, landscape and population genetic studies are also badly needed to establish natural levels of gene flow, including movement across seemingly inhospitable habitat patches, for this windblown sand habitat specialist. These data can advise and guide plans for habitat acquisition both now and in the face of climate change, and may be a critical element in establishing appropriate habitat corridors and supplementing ecological survey data to guide potential human-assisted translocation. Finally, these multi-locus microsatellite or SNP-based studies can help clarify the amount of migration (if any) between adjacent populations and effective population sizes of existing local populations.

SANDSTONE NIGHT LIZARD

Xantusia gracilis Grismer and Galvan 1986

Status Summary

Xantusia gracilis is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 38% (42/110). During the previous evaluation, it was also designated as a Species of Special Concern (as *Xantusia henshawi gracilis*; Jennings and Hayes 1994a).

Identification

Xantusia gracilis is a medium-sized (5.1–7 cm SVL) lizard with soft skin and granular scales on the dorsal surface, enlarged plates on the ventral surface, and a prominent gular fold (Grismer and Galvan 1986, Lovich and Grismer 2001, Stebbins 2003, Lovich 2009b). The dorsal coloration is pale tan/brown, with many round dark-brown spots, while the ventral surface is clean white or white, with a very small amount of black speckling on the front limbs and throat (Grismer and Galvan 1986). The head is flattened, and the eyes have vertically oriented pupils (Stebbins 2003). The overall body shape is relatively slender compared to its closest (and most similar) relative the granite night lizard (*X. henshawi*) (Grismer and Galvan 1986).

Within its range, *X. gracilis* is only likely to be confused with its sister species *X. henshawi*.

Sandstone Night Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	5
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	10
vi. Population trend (25)	0
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	0
Total Score	42
Total Possible	110
Total Score/Total Possible	0.38

PHOTO ON PREVIOUS PAGE: Sandstone night lizard. San Diego County, California. Courtesy of Jeff Lemm.

The two species do not overlap in range but occur within 32 km of each other. *Xantusia henshawi* has larger dark spots on the dorsal surface, more extensive speckling on the ventral surface, and an overall more robust body shape (Grismer and Galvan 1986). *Xantusia gracilis* also has an enlarged temporal scale (about half the size of the postparietal) compared to *X. henshawi* (typically less than one-quarter the size of the postparietal; Grismer and Galvan 1986). The peninsular leaf-toed gecko (*Phyllodactylus nocticolus*) also occurs in the vicinity of *X. gracilis*, but this lizard lacks the dark-brown dorsal spots and has prominent, expanded toe tips.

Taxonomic Relationships

Xantusia gracilis was initially described as a subspecies of *X. henshawi* on the basis of color, scalation, allozyme variation, and behavior (Grismer and Galvan 1986). The taxon was elevated to species status because it is diagnosable, geographically isolated, and forms a monophyletic clade nested within *X. henshawi* for a single mitochondrial locus (Lovich 2001). This arrangement is now widely accepted.

Life History

The life history of *Xantusia gracilis* is poorly studied, particularly so in wild populations. Given the species' overall similarity in most respects to *X. henshawi*, we expect that life history information from *X. henshawi* is a reasonably good predictor for *X. gracilis* (Lee 1975). However, the two taxa live in distinct habitats and show some behavioral differences in captivity, so some life history differences probably exist in the wild. *Xantusia gracilis* is likely active from spring through fall (Lemm 2006). In captivity, it has been shown to be more strongly nocturnal than *X. henshawi*, more frequently found on the sandy substrate on the bottom of the enclosure and does not seem to be limited to rock faces (Lee 1975, Grismer and Galvan 1986). Based on what is known about *X. henshawi*, we expect that *X. gracilis* has a low metabolic rate and is quite sedentary, feeding primarily ants, beetles, and spiders (Brattstrom 1952, Lee 1975, Mautz 1979). In captivity, *X. gracilis* are also known to feed on the eggs of *Phyllodactylus nocticolus*, a behavior that captive *X. henshawi* in the same enclosure did not exhibit (Grismer and Galvan 1986). In *X. henshawi*, mating occurs in June and July, with one or two live young born in September or October (Brattstrom 1951, Lee 1975), and this may also be the case for *X. gracilis*. Individuals probably do not become reproductively mature until 2.5–3.5 years of age and are likely long-lived, although field data are lacking (Lee 1975).

Habitat Requirements

Xantusia gracilis lives in eroding sandstone and mudstone habitat where it utilizes crevices, rodent burrows, and the undersides of exfoliating rock flakes as shelter (Grismer and Galvan 1986). At night, it emerges from its shelters and can be found moving about on the surface (Grismer and Galvan 1986). This species is less dependent on exfoliating rock habitat than *X. henshawi* (Grismer and Galvan 1986).

Distribution (Past and Present)

Xantusia gracilis is restricted to one small area, approximately 3.9 km^2 in total area, on the southeastern flank of the Santa Rosa Mountains, entirely within Anza Borrego Desert State Park (Grismer and Galvan 1986). The known elevational range extends from approximately 240 to 305 m. Within this small region the species is patchily distributed, common in some areas and apparently absent in others (Grismer and Galvan 1986). *Xantusia henshawi* occurs approximately 32 km to the north and west, and no xantusiid lizards are known from the intervening area. No historical distribution data are available for this taxon, although we have no reason to think that the distribution has declined recently.

Trends in Abundance

No data on historical or current abundance have been published, although some have suggested that habitat quality has declined due to collection activity (R. Lovich, pers. comm.).

The extent and severity of such impact has not been quantified (R. Fisher, pers. comm.). Some amount of illegal collection occurs for this species, which may be driving small declines (M. Jorgensen, pers. comm.).

Nature and Degree of Threat

Xantusia gracilis lives in a fragile habitat in an extremely localized area. Damage to this small patch of habitat, be it from habitat destruction, invasive species, collecting, or climate change, is the largest risk facing the species (Lovich 2009b). It is also likely long-lived and late maturing with a low reproductive potential, and populations are likely to be slow to recover from declines. Some amount of illegal collecting occurs, which could be contributing to such declines, particularly in areas that are most easily accessible by road.

Status Determination

The extremely localized range and relative fragility of *Xantusia gracilis*' habitat are significant risk factors. The species' life history also predisposes it to decline in the face of any increased adult mortality. Although data are almost entirely lacking, *X. gracilis* appears to be relatively stable at the present time; thus, we designate it as a Priority 3 Species of Special Concern.

Management Recommendations

Limiting access and minimizing disturbance to *Xantusia gracilis*' habitat is currently the most important component of effective conservation. This management strategy should be reviewed as needed depending on the results of the surveys outlined below. All collecting should be restricted or eliminated unless it is absolutely necessary for scientific purposes that further conservation of this species.

Monitoring, Research, and Survey Needs

As published historical or current abundances of *Xantusia gracilis* are lacking, publication of any existing data is a priority. Formal monitoring should be initiated to establish and publish baseline population data. These surveys should be performed at night, and it is essential not to disturb the fragile microhabitat (e.g., moving rocks or rock flakes, excavating rodent burrows). Aside from estimating population size, these surveys should also quantify and document any observed habitat disturbance. Year-to-year fluctuations in population size occur in other xantusiid lizards (Lee 1975) and are to be expected in *X. gracilis* as well. Establishing a long-term monitoring program is a critical objective. Additional surveys to establish the precise limits of the range of *X. gracilis* will help determine best practices for managing its fragile habitat in the heavily used Anza-Borrego Desert State Park.

The life history of this species has not been studied and an autecological study is badly needed to provide basic information on habitat suitability and reproduction. These data will be urgently needed should more extensive management efforts become necessary.

Finally, multi-locus microsatellite or SNP data should be collected to provide genetic estimates of effective population size, and potentially levels of gene flow, even for this restricted species. A key issue for this species is to sample individuals without invasive tissue-removal techniques, and it would probably be best to work out such protocols on *X. henshawi* before applying them to *X. gracilis*.

SIERRA NIGHT LIZARD

Xantusia vigilis sierrae Bezy 1967

Status Summary

Xantusia vigilis sierrae is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 47% (52/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Xantusia vigilis sierrae is a small (4–5.1 cm SVL), somewhat flattened lizard with granular dorsal scales, enlarged square ventral scales, soft skin, and a prominent gular fold (Bezy 1967, Stebbins 2003). The head is covered with enlarged plates, the eyes are lidless, and the pupils are vertical (Stebbins 2003). Most specimens are olive or grayish brown above, with a pattern of interconnected dark markings that form a network, which may give the animal a mottled appearance (Bezy 1967, Stebbins 2003). The ventral surface is light bluish pink and generally unmarked (Bezy 1967). A prominent light stripe extends from the rear of the eye posteriorly to the neck or just beyond the neck (Bezy 1967).

Within its range, *X. v. sierrae* is unlikely to be confused with other species, although it is similar in appearance to the Yucca night lizard

Sierra Night Lizard: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	0
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	10
vi. Population trend (25)	5
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	52
Total Possible	110
Total Score/Total Possible	0.47

PHOTO ON PREVIOUS PAGE: Sierra night lizard, Kern County, California. Courtesy of Jackson Shedd.

(*X. v. vigilis*), which occurs nearby. *Xantusia vigilis vigilis* has fairly distinct dark spots on the dorsal surface that do not form a network, though they form narrow longitudinal stripes in some populations (Stebbins 2003). Several aspects of the scalation also differentiate these two subspecies (Bezy 1967).

Taxonomic Relationships

Xantusia vigilis sierrae is a member of the *X. vigilis* species complex. It was initially recognized on the basis of habitat type, coloration, scalation, and femoral pore count (Bezy 1967). Since its initial recognition, genetic analyses have shown that *X. v. sierrae* forms a monophyletic group embedded within *X. vigilis* for mitochondrial and nuclear DNA sequence data (Sinclair et al. 2004, Leavitt et al. 2007). Allozyme data also suggest that it is distinct, but a close relative of *X. v. vigilis* (Bezy and Sites 1987). One population of *X. v. vigilis* that occurs within 60 km of *X. v. sierrae* is suspected to contain intergrades based on femoral pore counts (Bezy 1967), although geographically more proximate populations (~20 km apart) show no evidence of this intermediate condition (Leavitt et al. 2007). Sinclair et al. (2004) considered *X. v. sierrae* a "candidate species" whose status required further testing with additional data. Some recent taxonomic lists have elevated it to species status without additional justification (de Queiroz and Reeder 2008, Collins and Taggart 2009). The weight of current evidence suggests that species status is probably warranted, and a population genetic analysis of *X. v. sierrae* and nearby *X. v. vigilis* populations is needed help clarify this issue. In particular, such a study could quantify whether, and to what extent, migration and intergradation occur along the eastern edge of the taxon's range.

Life History

The life history of *Xantusia vigilis sierrae* has not been studied. However, among xantusiid species that have been examined, life history features are largely conserved across southwestern United States, and we assume that the life history of *X. v. vigilis* may be a good predictor for *X. v. sierrae* in many respects. *Xantusia vigilis sierrae* is primarily a rock-dwelling species, whereas *X. v. vigilis* is more of a habitat generalist with some preference for fallen vegetation. Some aspects of the life history may therefore be more similar to other rock-specialist night lizards (e.g., *X. henshawi* or *X. gracilis*).

Based on information from other species, *X. v. sierrae* is probably a generalist predator that consumes a variety of small invertebrate prey (Brattstrom 1952, Stebbins 2003, Bezy 2009). Its diet is probably dominated by ants and other insects that occur within crevices (Brattstrom 1952, Bezy 2009). *Xantusia vigilis sierrae* is probably long-lived and takes 2.5–3.5 years to reach sexual maturity (Lee 1975), eventually producing 1 or 2 live young/year (Brattstrom 1951). This species likely has a low metabolic rate relative to other lizards and grows slowly (Mautz 1979). Daily activity cycles are unknown. Some rock-dwelling night lizards are largely diurnal and/or crepuscular (*X. henshawi*; Mautz and Case 1974), while others appear to be nocturnal (*X. gracilis*; Grismer and Galavan 1986).

Habitat Requirements

Xantusia vigilis sierrae is known primarily from exfoliating granite outcrops (Bezy 1967), though it can also be found under tree bark that has fallen on the ground or is loosely attached to trees (D. Leavitt, pers. comm.). Within its rocky habitat type, this species is more frequently found under large horizontal cap rocks than the more numerous, vertically oriented smaller flakes (Bezy 1967). *Xantusia vigilis sierrae* is also more frequently found in small clusters of one or a few boulders than in larger rock piles on rocky slopes and canyons (Bezy 1967). Some authors have speculated that this may reflect varying abundances associated with differences in predator access

(Jennings and Hayes 1994a), although it is also possible that it reflects differences in detectability. The dominant vegetation of its preferred habitat is foothill grassland with interspersed shrubs and woody vegetation (Bezy 1967).

Distribution (Past and Present)

Xantusia vigilis sierrae is known only from rocky hillsides on the western edge of the Greenhorn Mountains near Granite Station, Kern County, California (Bezy 1967, Stebbins 2003). The known elevational range extends from 450 to 500 m (Bezy 1967). No significant changes in distribution are known, although the development of small ranches may impact populations in the area.

Trends in Abundance

No historical or current abundance data are available for this taxon, although these lizards do not currently appear to be rare (D. Leavitt, pers. comm.). Moderate habitat degradation from previous collecting efforts as well as moderate amounts of landscape modification may be causing declines (R. Fisher, pers. comm.), although this has not been confirmed.

Nature and Degree of Threat

The primary threat facing *Xantusia vigilis sierrae* is its exceedingly small range that occurs on unprotected land. Development in the region is taking place and could have catastrophic effects on the existing populations, as could any fragmentation of the habitat that isolates granite outcrops in which this lizard lives. The rock cap and crevice habitat that this species prefers is also susceptible to degradation by humans (Jennings and Hayes 1994a, Stebbins 2003; D. Leavitt, pers. comm.).

Status Determination

Xantusia vigilis sierrae is a narrowly distributed habitat specialist that is endemic to a small region of the Sierra Nevada. However, no distributional declines have been documented, and only small declines in abundance are suspected, resulting in a Priority 3 designation.

Management Recommendations

To protect this species, habitat loss and degradation need to be avoided. Effective protection of this species can likely be accomplished by protecting rocky habitats from most human interference, including intensive collecting efforts and protecting the surrounding area from development. Housing development in the form of ranchettes and other rural development projects should be closely managed to avoid impacting *Xantusia vigilis sierrae* populations, including provisions for habitat corridors to prevent fragmentation. It is unknown whether grazing adversely affects the species.

Monitoring, Research, and Survey Needs

Given the almost complete dearth of ecological work on this species, several research and monitoring needs are required for its future management and protection. Until recently, this species was known only to inhabit exfoliating granite, although its actual habitat utilization now seems to be somewhat broader than this. Further study of habitat use and preferences in *Xantusia vigilis sierrae* is essential to establish an effective management program.

A long-term population monitoring program needs to be initiated for this species, ideally across all utilized habitat types. These monitoring programs need not be extensive, but at minimum should document population size in disturbed and pristine habitats at regular intervals. Such monitoring can provide both critical data on natural population fluctuations and an early warning of declines in their initial stages.

Finally, genetic analyses using multiple nuclear markers are needed to address two important conservation issues. First, additional work at the phylogeographic/species boundary level is needed to determine whether *X. v. sierrae* is best considered a species or subspecies within the *X. vigilis* complex. An important aspect of this work should be to examine populations in close proximity to *X. v. vigilis* to determine the degree and extent of admixture between these taxa. Second, landscape genetic work across its limited range is needed to quantify the degree of population isolation and substructure among habitat patches, migration corridors that are most heavily used by the lizards, and effective population sizes of populations in ecologically diverse habitat patches. Ideally, tissue samples in the form of small tail clips should be collected each year from study populations to allow for genetic as well as demographic estimation of population size fluctuations over time.

CALIFORNIA GLOSSY SNAKE

Arizona elegans occidentalis Blanchard 1924

Status Summary

Arizona elegans occidentalis is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 67% (74/110). It was not on the list of candidates considered for Species of Special Concern designation during the previous evaluation (Jennings and Hayes 1994a).

Identification

Arizona elegans occidentalis is a medium-sized colubrid (64–99 cm SVL) with tan or brown dorsal coloration. It has dark-brown blotches edged in black running down the back and a series of similar, though smaller, blotches running down the sides (Klauber 1946, Stebbins 2003, Lemm 2006). The dorsal coloration is often lighter middorsally and darkens to a deeper brown on the sides. The lateral blotching sometimes touches the edges of the ventral belly scales, but otherwise the underside is unmarked (Klauber 1946). Scales are unkeeled, smooth and glossy, and only one pair of prefrontals are present (Stebbins 2003). A dark stripe runs from the corner of the mouth to the eye on each side of the face, and a third stripe connects the eyes across the posterior edge of the prefrontals (Blanchard 1924). An additional

California Glossy Snake: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	25
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	3
v. Ecological tolerance (10)	3
vi. Population trend (25)	25
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	10
Total Score	74
Total Possible	110
Total Score/Total Possible	0.67

PHOTO ON PREVIOUS PAGE: California glossy snake, San Diego County, California. Courtesy of Jeff Lemm.

dark spot is usually present below each eye (Klauber 1946).

In California, this taxon could be confused with other subspecies of *A. elegans*, with the gopher snake (*Pituophis catenifer*), or the night snakes (*Hypsiglena* spp.). This subspecies is generally darker than other subspecies of *A. elegans* in California, though intergrades are common along the desert slopes of the coastal mountains (Klauber 1946). Generally, *A. e. occidentalis* is best distinguished from other subspecies based on range. *Pituophis catenifer* has keeled scales and (usually) two pairs of prefrontals, while *Hypsiglena* is smaller (up to 66 cm), has strongly elliptical pupils, and an extensive dark blotch on the neck (Stebbins 2003).

Taxonomic Relationships

Arizona elegans occidentalis was initially described on the basis of scale counts and dorsal blotching and included all snakes in this genus ranging from California through southeastern Arizona (Blanchard 1924). Klauber (1946) later restricted this taxon and described two new subspecies occurring in eastern California (the Mojave glossy snake, *A. e. candida*, and the desert glossy snake, *A. e. eburnata*), which differ from *A. e. occidentalis* primarily in body color. Intraspecific (or intrageneric) variation has not yet been assessed genetically, although at the generic level, *Arizona* appears to be a relatively distant sister taxon to the long-nosed snake (*Rhinocheilus lecontei*) (Pyron and Burbrink 2009).

Life History

Arizona elegans is a nocturnal snake that is generally active from late February until November, depending on local weather conditions (Klauber 1946, Grismer 2002). In California, *A. e. occidentalis* reaches peak activity during May (Klauber 1946; S. Sweet, pers. comm.), with few specimens being collected throughout the remainder of the summer (Klauber 1939, Goldberg 2000). The species feeds primarily on diurnal lizards, which it captures while they sleep, and small nocturnal mammals, which it ambushes (Klauber 1946, Rodríguez-Robles et al. 1999a). In a sample of 107 prey specimens, 50% were lizards (primarily *Sceloporus* and *Uta*) and 44% were mammals (primarily small rodents). Larger specimens are also known to take small birds and other snakes (Rodríguez-Robles et al. 1999a).

Arizona elegans retreats to burrows during the day, using either existing mammal burrows, excavations under rocks, or creating burrows for itself (Klauber 1946, Degenhardt et al. 1996). This species can be nocturnally active at relatively low temperatures (as low as 14°C, though typically 19–20°C; Cowles and Bogert 1944).

Reproduction is poorly studied in the wild, but museum specimens indicate that ovulation begins in June, and spermiogenesis occurs in late summer (Goldberg 2000). In *A. elegans* from New Mexico, ovulation also begins in June with oviposition occurring in July (Aldridge 1979). Clutch size is poorly documented in this subspecies, though two individuals contained three and seven eggs, respectively (Reynolds 1943, Klauber 1946). Across *A. elegans*, clutch size varies widely from 3 to 23 eggs, with a mean of 8.5 (Fitch 1970). Recent hatchlings are typically found in September (S. Sweet, pers. comm.).

Habitat Requirements

Arizona elegans is found in a wide variety of habitat types, including open desert, grasslands, shrublands, chaparral, and woodlands. However, only a subset of these habitat types occurs within *A. e. occidentalis*' range, primarily grasslands, fields, coastal sage scrub, and chaparral (Klauber 1946). No studies of habitat requirements exist, although this subspecies appears to prefer open microhabitats. The majority of records occur in relatively open patches in a surrounding matrix of denser vegetation (Klauber 1946). This subspecies can be patchy within its range, with certain areas consistently producing more records than others that have seemingly identical habitat (Klauber

1946). *Arizona elegans* appears to prefer areas where the soil is loose, which allows for burrowing (Grismer 2002, Stebbins 2003). Unpublished survey data indicate that *A. e. occidentalis* may prefer sandy soil habitats such as coastal sand dunes, alluvial creek beds, and ancient dunes on the marine terraces (R. Fisher, pers. comm.).

Distribution (Past and Present)

Range-wide, *Arizona elegans* occurs throughout much of southwestern North America, extending east as far as central Texas, Oklahoma, and Kansas, and south to central Mexico. Klauber (1946) restricted *A. e. occidentalis* to the central San Joaquin Valley south to the Tehachapi Mountains and along the base of the Coast Range mountains farther south to San Quintin, Baja California. This subspecies is known to occur from sea level to ~1800 m (Lemm 2006).

Arizona elegans occidentalis has apparently declined throughout much of its range. In San Diego County, survey data are available for Torrey Pines State Reserve, Point Loma, and the Tijuana Estuary. The subspecies was formerly present in these areas but now appears to be extirpated (Wells 1998, Case and Fisher 2001, Fisher 2004). Extensive agricultural development and habitat modification throughout the San Joaquin Valley and urban development within the Los Angeles basin have likely led to declines and/or extirpations in these areas as well (Stebbins 2003; R. Fisher, pers. comm.).

Trends in Abundance

Few abundance data exist for this subspecies. However, extensive early surveys of snakes in San Diego County failed to find the species, suggesting that they were uncommon (Klauber 1924). Bogert (1930) was aware of only two records for Los Angeles County. Klauber (1946) observed that *Arizona elegans occidentalis* existed in lower densities, relative to the total snake population, than either *A. e. candida* or *A. e. eburnata*, and that *A. e. occidentalis* was patchily distributed. Pitfall trapping data collected by the US Geological Survey (USGS) over 17 years in San Diego, Orange, and Los Angeles Counties have resulted in only a single capture of this taxon (C. Rochester, pers. comm.). Presently, the subspecies is found less commonly than it once was throughout the San Diego region (Case and Fisher 2001, Lemm 2006). Both low densities and patchiness could make this taxon particularly susceptible to declines and may explain why the species has seemingly disappeared from some areas, while several other colubrid snakes remain present. Development continues within the species' range and thus ongoing declines in abundance are likely.

Nature and Degree of Threat

The greatest threat to this subspecies is habitat modification due to agricultural, commercial, and residential development. However, the specific mechanisms that cause declines are not well understood. Abundant prey and small habitat blocks that appear suitable remain in some developed areas, although the species may be sensitive to the light pollution arising from this development (Perry and Fisher 2006, Perry et al. 2008). This species' response to wildfire is not well understood, but increasing frequency and intensity of wildfires due to climate change may plausibly lead to habitat modification that impacts this taxon. The projected changes in wildfire regime in this area are mixed (PRBO 2011), so the degree of this threat is still unknown. Wildfires that are small in scale and intensity may have a beneficial impact by temporarily clearing patches of chaparral habitat, which then recover over a period of a few years, creating the patchwork of open and densely vegetated habitat that this species appear to prefer. Large and intense wildfires, conversely, kill chaparral and convert large habitat patches to grassland for longer periods of time. This process would likely have a detrimental impact on this species.

Status Determination

A moderately small range and moderate degree of ecological specialization and endemism, coupled with documented declines within this species range and projected impacts from ongoing development, contribute to a Priority 1 designation for this subspecies.

Management Recommendations

Habitat protection is currently the most important management priority for *Arizona elegans occidentalis*. The studies outlined below will help to characterize habitat usage, home range size, distribution, and abundance. Once these data become available, a more specific management program can be developed that targets specific remaining populations and protects appropriately sized habitat blocks for the species' home range size and movement patterns.

Monitoring, Research, and Survey Needs

This is a poorly studied component of California's herpetofauna. Two immediate research priorities exist for this taxon. First, ecological studies need to be initiated to enhance our currently poor understanding of the life history and existing population sizes in this subspecies. Without this basic information, designing a coherent management strategy is impossible. These studies should take place in concert with survey efforts to more precisely quantify the subspecies' present distribution. These surveys should employ a variety of techniques, likely including night driving, snake trapping, and artificial cover object transects in order to increase capture success. If reasonably high capture rates can be obtained, individually marking snakes for mark–recapture population size estimates should also be performed. Radiotelemetry studies may be a fruitful means for determining home range size and more thoroughly characterizing habitat usage, particularly given the indications that this species might have specific microhabitat preferences. Second, a species-wide phylogeographic study should be performed in order to elucidate intraspecific variation and identify appropriate units for conservation. Phylogenetic and phylogeographic studies of other wide-ranging snakes have frequently led to changes in the understanding of species boundaries and diversity, including the genetic diversity that exists within a species and its concordance with morphological subspecies boundaries. Finer-scale landscape ecological studies, particularly in concert with radiotelemetry on the same landscapes, would also provide important information for conservation strategies. These important data are entirely lacking for this taxon at present.

RED DIAMOND RATTLESNAKE

Crotalus ruber Cope 1892

Status Summary

Crotalus ruber is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 44% (48/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Crotalus ruber is a large (165 cm TL), heavy-bodied, tan, brick-red, reddish- or pinkish-brown rattlesnake (Stebbins 2003). As is typical of pit vipers, *C. ruber* has a large triangular head, a thin neck, and a heat-sensing pit on each side of the head between the eyes and nostrils. An average of 35 light-edged or indistinct diamonds run down the back (Ernst and Ernst 2003). The tail is ringed with alternating bands of black and white or gray, ending in a rattle. Two light stripes occur on the sides of the head, and the venter is light colored and unmarked (Ernst and Ernst 2003). The dorsal body scales are keeled.

Neonates of *C. ruber* are similar in appearance to the western diamond-backed rattlesnake (*C. atrox*). Adults can be distinguished by coloration and behavior, with *C. ruber* much redder and less aggressive than *C. atrox*. In California, the ranges of these two species barely meet (Stebbins 2003).

Red Diamond Rattlesnake: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	0
v. Ecological tolerance (10)	3
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	7
Total Score	48
Total Possible	110
Total Score/Total Possible	0.44

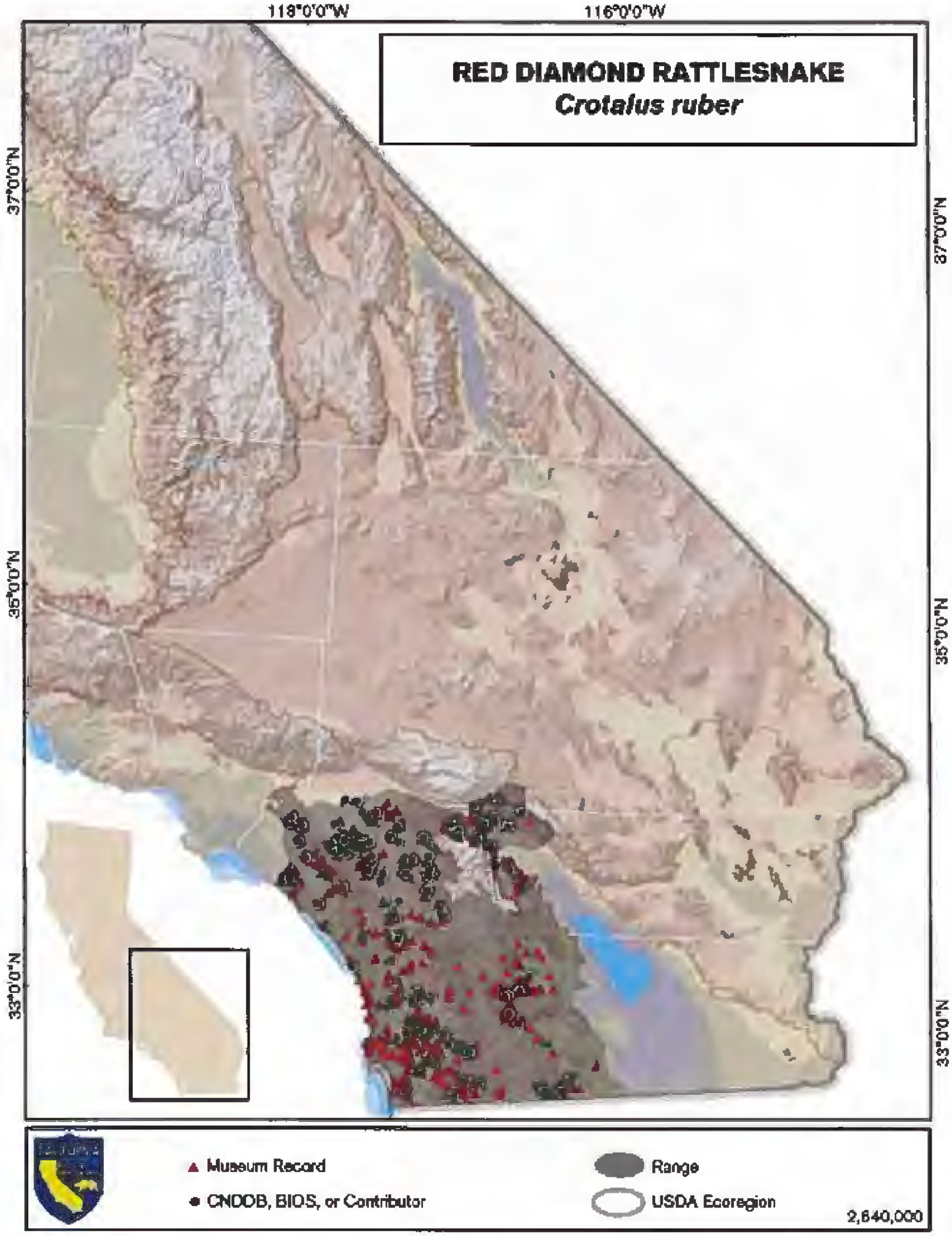


PHOTO ON PREVIOUS PAGE: Red diamond rattlesnake, San Diego County, California. Courtesy of Jeff Lemm.

Taxonomic Relationships

Based on analyses of morphology and mitochondrial DNA, Murphy et al. (1995) proposed synonymizing *Crotalus ruber* with an island species, the Cedros Island diamond rattlesnake (*C. exsul* Garman 1884). Because *C. exsul* was named first, Murphy et al. (1995) suggested changing the name of *C. ruber*. However, this has been opposed in favor of stability of the nomenclature (Smith et al. 1998, ICZN 2000). Here, we use *C. ruber* to refer to all California animals.

Life History

Crotalus ruber is generally most active between March and June (Ernst and Ernst 2003). In one study from San Diego County, snakes typically emerged from overwintering locations in late February, but some individuals were inactive until mid-April (Brown et al. 2008). Most movement occurred in late spring and summer, dens were populated in November, and no movement was recorded in December or January (Brown et al. 2008). In Riverside County, desert animals were active from early March to late November (Greenberg 2002).

During the cold winter months, *C. ruber* spends most of its time underground in dens located in rock crevices, animal burrows, or under shrubs or cacti. Several individuals may aggregate in these dens, but denning behavior is variable across sites (Klauber 1956, Ernst and Ernst 2003). In one study in San Diego County, 7 out of 11 radio-tracked snakes overwintered in communal dens located in rock crevices of granite boulders with up to 7 other individuals (Brown et al. 2008). Most snakes reused den sites over multiple years and moved ~300 m away from den sites during the active period the following year (Brown et al. 2008). In contrast, in sites where large rocks were rare, animals were observed to overwinter singly under prickly pears (*Opuntia* sp.), did not show consistent site fidelity to overwintering sites, and moved farther from overwintering sites after emergence (Greenberg 2002, Dugan et al. 2008).

Home range area is also variable in this species, and male home ranges are larger than those of females (Tracey 2000, Greenberg 2002, Brown et al. 2008). The few available radiotelemetry studies suggest that home ranges may be larger in the desert than in coastal habitats. In a reserve in San Diego County, average home range sizes were 2.8 ha for males ($n = 5$) and 0.9 ha for females ($n = 6$; Brown et al. 2008). At another relatively coastal site in Chino Hills State Park in southwestern San Bernardino County, Dugan et al. (2008) found that male home range size varied from 0.3 to 4.5 ha ($n = 7$). In contrast, average home range sizes for desert animals from Riverside County were 25.7 ha for males ($n = 5$) and 5.9 ha for females ($n = 4$; Greenberg and McClintock 2008).

Courtship and mating have been observed in the field in California from February to May (Brown et al. 2008, Dugan et al. 2008). In San Diego County, Brown et al. (2008) witnessed females mating from April to May (sometimes with den mates), and births occurred in September. Goldberg (1999) examined the reproductive condition of 43 specimens, 41 of which were from desert habitat in Riverside County and 2 from coastal Orange County. Reproductively active males were observed in August (Goldberg 1999). Although specimens were unavailable from later in the year, Goldberg (1999) speculated that sperm production continued through the early fall. Sperm was found in the vas deferens for all animals (collected February through August), suggesting the use of sperm stored overwinter for spring mating (Goldberg 1999). Females contained enlarged ovarian follicles (>10 mm) from March through September. Females may reproduce every other year, given that only 7 of 15 females showed evidence of reproductive activity (Goldberg 1999).

An average of eight young (range 3–20, $n = 40$; Klauber 1956) are live-born after a gestation period of 141–173 days ($n = 3$, data from captive animals; Klauber 1956). Goldberg (1999) estimated similar average litter sizes from counts of enlarged ovarian follicles (range 4–8, mean

6.3, $n = 7$). Klauber (1956) examined 249 specimens from San Diego County to estimate growth curves and found that young are 30 cm TL at birth on average and roughly double in length during their first year. Estimates for size at reproductive maturity range from 60 to 75 cm TL (Klauber 1956, Wright and Wright 1957, Goldberg 1999).

Crotalus ruber mostly feeds on small mammals but will also eat lizards, birds, and other snakes (Tevis 1943, Klauber 1956, Cunningham 1959b, Patten and Banta 1980). Dugan and Hayes (2012) compiled range-wide dietary data from museum specimens, live animals, road kills, existing literature, and other observations. Roughly 92% of all prey items were mammals, with lizards (8%) and birds (1%) taken less frequently. Prey items were found in snakes collected year-round, suggesting that *C. ruber* occasionally feeds during the winter (Dugan and Hayes 2012).

Habitat Requirements

Crotalus ruber occurs in several habitat types, including coastal sage scrub, chamise chaparral, redshank, desert slope scrub, desert washes, grassy fields, orchards, cactus patches, and rocky areas (Klauber 1956, Jennings and Hayes 1994a, Tracey 2000, Dugan et al. 2008). Klauber (1956) noted that 44% (30/68) of animals were found near heavy shrub and chaparral, and 21% (14/68) were found near rocks and boulders in road surveys. On a reserve in San Diego County, snakes were found in association with rock outcrops 57% of the time and in shrubby vegetated habitats without rocks 28% of the time (Brown et al. 2008). There are several accounts of *C. ruber* climbing in bushes and trees up to 2 m off the ground (Klauber 1956 and pers. comm. therein) and *C. ruber* has also been observed swimming in reservoirs (Klauber 1956).

In one radio-tracking study from San Diego County, habitat use was nonrandom with respect to available vegetation. Snakes preferred scrub vegetation less than 1.5 m tall and avoided human development (Tracey 2000). For animals that were radio-tracked in fragmented habitats, none were observed to cross a developed edge or road over a 2-year period. For example, one adult male in a naturally vegetated fragment actively avoided a road edge, and turning movements away from this edge were detectable up to 50 m from the road (Tracey et al. 2005).

Dugan et al. (2008) radio-tracked adult males at a site that lacked large rocks but had cactus, coastal sage scrub, nonnative grassland, riparian areas, and oak woodland habitats. The preferred habitat was cactus patches of prickly pear (*Opuntia* sp.) followed by chaparral, and none of the tracked snakes used oak woodland. Several individuals spent most of their time within a single cactus patch during the year (Dugan et al. 2008).

Distribution (Past and Present)

Crotalus ruber has a small range in California, occupying the southwestern corner of the state. It occurs in southeastern Los Angeles and Orange Counties, the Morongo area of southwestern San Bernardino County, western Riverside County, San Diego County, and extreme southwestern Imperial County (Klauber 1956). *Crotalus ruber* occurs in areas with rainfall ranging from 8 to 80 cm/year, usually in areas below 1200 m in elevation (Klauber 1956). The geographic range of *C. ruber* extends out onto the desert floor from the eastern slope of the Peninsular Ranges (Klauber 1956). Outside of California its range extends south through Baja California and several nearshore islands (Klauber 1956).

Much of the range in California is in close proximity to areas of high human density. Jennings and Hayes (1994a) estimated that *C. ruber* was extirpated from roughly 20% of historical sites and attributed extirpations to habitat loss from urbanization and agriculture. Coastal populations are the most reduced, particularly in southern San Diego County (S. Barry, pers. comm.). Case and Fisher (2001) conducted pitfall trapping surveys in southern California and did not capture or observe animals at several localities where Klauber (1939 and unpublished data) had previously noted them as common. Halama et al. (2008) noted that many native habitat localities where snakes were collected in the 1990s in western Riverside County have now been developed.

Trends in Abundance

While population estimates are not available, population declines are suspected due to habitat loss and fragmentation. Current declines of existing populations may be occurring particularly in the Morongo Valley in the northern end of the range due to development (S. Barry, pers. comm.). In one San Diego County site, minimum density was estimated as 0.63 *Crotalus ruber* per hectare, although the actual density was likely higher (41 individuals observed haphazardly in a 65 ha area over ~5 years; Brown et al. 2008).

Nature and Degree of Threat

Crotalus ruber is mainly threatened by development, which causes habitat loss and fragmentation. This species may also suffer from persecution and road mortality. Climate change may affect *C. ruber* through changes in fire regime and vegetation shifts. However, both increases and decreases in fire have been predicted, and there is little consensus because of the difficulty in modeling Santa Ana weather events in southern California (Westerling et al. 2004, Westerling and Bryant 2008). How *C. ruber* may respond to changes in fire regime is unknown. Climate change is predicted to decrease the availability of chaparral and shrubland by up to 44%, while grassland is predicted to increase by up to 390% in southern California (Lenihan et al. 2008, PRBO 2011). Though *C. ruber* has been documented in grassy areas, large losses in shrub habitat may negatively affect this species.

Status Determination

Crotalus ruber has a small range in California that includes areas of high human population density and development, resulting in a Priority 3 Species of Special Concern designation.

Management Recommendations

Remaining populations of *Crotalus ruber* in California often occur in habitats that are fragmented by roads and urban development. Existing large habitat fragments should be identified and protected. For example, a proposed Habitat Conservation Plan for western Riverside County represents a 3.5-fold increase in the amount of snake habitat protected (Halama et al. 2008). However, Halama et al. (2008) estimated from habitat suitability models that roughly 100,000 ha of predicted highly suitable habitat in the area would still be unprotected and at risk of development. It may be possible to reduce road mortality with wildlife tunnels and associated drift fences installed beneath high-traffic roads.

Monitoring, Research, and Survey Needs

Additional research into *Crotalus ruber* ecology and population dynamics in developed and fragmented landscapes would be useful for developing management strategies, particularly with regard to maintaining connectivity among populations. Creating habitat buffers around large remaining fragments and habitat corridors between fragments may help populations persist in these landscapes, but more research on habitat use and corridor placement is needed. Radiotelemetry data to date suggest high site fidelity among adults, at least in some coastal populations within years. Juveniles may show different dispersal behavior and benefit more from management strategies like habitat corridors (Tracey 2000). Current snake telemetry techniques that rely on surgically implanted transmitters have a lower size limit (e.g., animals needed to be >500 g in one study; Brown et al. 2008), making it difficult to study movement in small individuals. In these cases, landscape genetic data could provide important data to complement more detailed telemetry studies. The role of hibernacula in population viability and movement patterns is also an important research need, particularly for juveniles.

Monitoring is needed to estimate abundances in addition to ongoing work on presence/absence to document local extirpations. Pitfall trapping has been used to successfully document presence and absence of this species in southern California (e.g., Case and Fisher 2001), and pitfall arrays that specifically compare habitats with different levels of human disturbance would provide valuable monitoring information.

REGAL RING-NECKED SNAKE

Diadophis punctatus regalis Baird and Girard 1853a

Status Summary

Diadophis punctatus regalis is a Species of Special Concern, although we refrain from assigning it a priority status at this time due to limited information. This taxon received a Total Score/Total Possible of 68% (27/40) and was not previously considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Diadophis punctatus regalis is the largest of the ring-necked snakes, reaching up to 85.7 cm TL, while most subspecies are less than 50 cm TL (Ernst and Ernst 2003). A slender snake with smooth scales, *D. p. regalis* is light gray, olive gray, or olive above with orange or red ventral coloration. The venter is speckled with irregular black spots. An orange or red neckband is generally present behind the head, though it can be faint or absent in some populations of this subspecies, particularly in New Mexico and Utah (Ernst and Ernst 2003, Stebbins 2003). Recent specimens from California and Nevada have lacked neck rings (Emmerich and Cunningham 2003, Wood and Richmond 2003).

Regal Ring-Necked Snake: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	Data deficient
iii. Population concentration/ migration (10)	Data deficient
iv. Endemism (10)	0
v. Ecological tolerance (10)	10
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	Data deficient
Total Score	27
Total Possible	40
Total Score/Total Possible	0.68

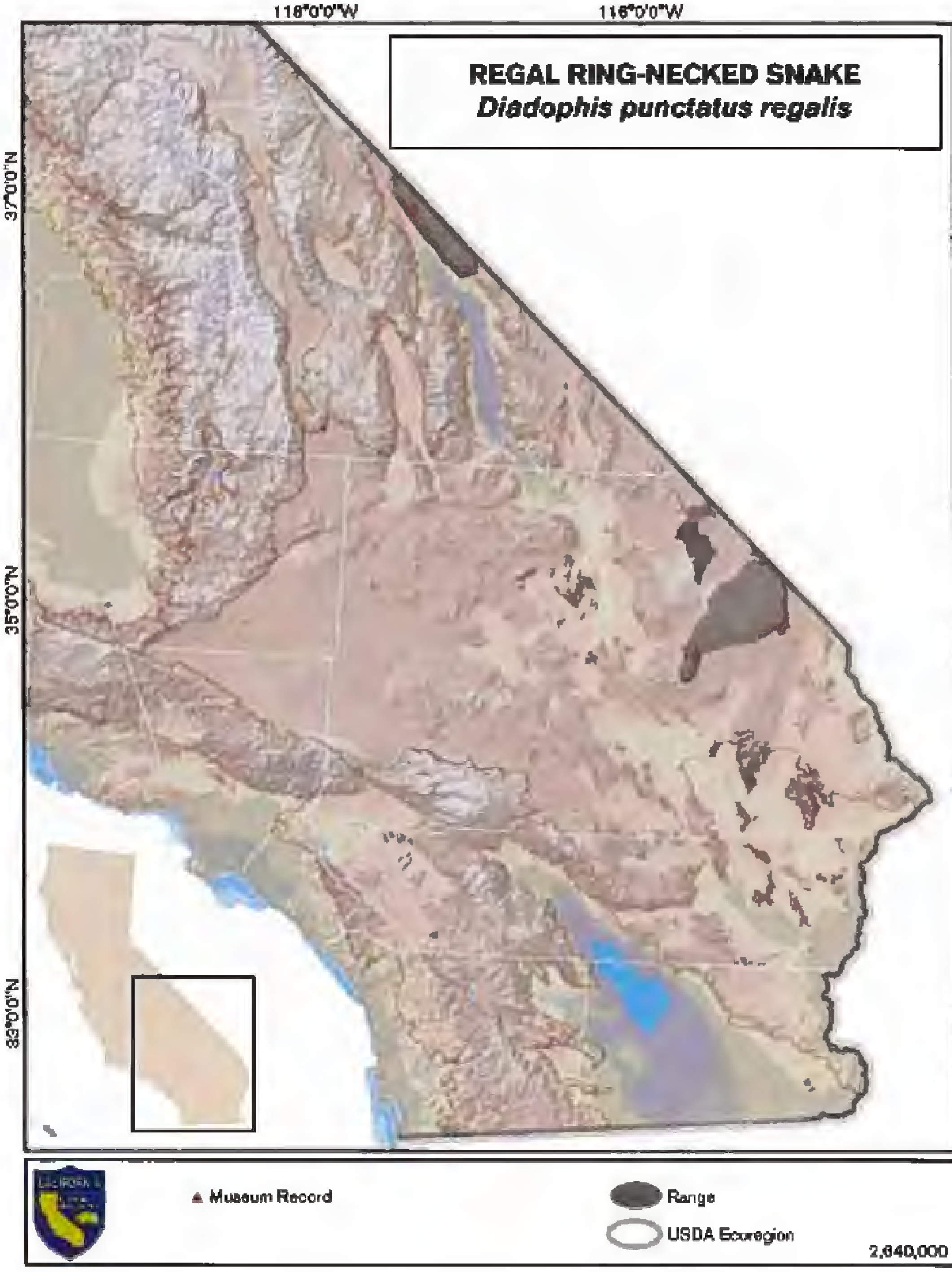


PHOTO ON PREVIOUS PAGE: Regal ring-necked snake, Santa Cruz County, Arizona. Courtesy of Jackson Shedd.

Taxonomic Relationships

Twelve subspecies of *D. punctatus* have traditionally been recognized, largely on the basis of morphology (Ernst and Ernst 2003). *Diadophis punctatus regalis* is one of seven subspecies that occur in California (Ernst and Ernst 2003). Recent molecular work has called this traditional view of the subspecies into question (Feldman and Spicer 2006, Fontanella et al. 2008), and a taxonomic revision is likely in the near future. Feldman and Spicer (2006) sampled mitochondrial DNA from 39 animals throughout the range of *D. punctatus* in California. *Diadophis punctatus regalis* was recovered as sister to a clade containing all other California samples, although only two *D. p. regalis* individuals were included in the analysis (one from California and one from Arizona). Fontanella et al. (2008) conducted a more comprehensive phylogeographic analysis of *D. punctatus*, sampling across the known range of the species in the United States. The previously recognized seven subspecies in California were found to fall into four lineages, with *D. p. regalis* as a part of a Great Basin clade. Fontanella et al. (2008) concluded that species-level diversity is currently underestimated, warranting a full taxonomic review requiring further sampling (particularly throughout Mexico) and the addition of nuclear markers.

Life History

Very little natural history information is available for *Diadophis punctatus regalis*, especially for California populations. Being such a widespread species, life history characteristics vary greatly across the species' range. It is reasonable to presume that *D. p. regalis* are ecologically distinct from other California *D. punctatus* populations based on their much larger size and unique restriction to desert spring habitats. Unless stated explicitly, life history information here is from other subspecies of *D. punctatus* and caution should be used in generalizing to *D. p. regalis*.

Diadophis punctatus is most active in the spring and early fall, and is primarily nocturnal (Ernst and Ernst 2003). Males aggregate for mating in the spring and fall (Noble and Clausen 1936, Dundee and Miller 1968). Females are thought to reproduce annually and may produce more than one clutch per year (Ernst and Ernst 2003). Oviposition occurs from May to September but is concentrated in June and July, and hatching occurs from July to September (Ernst and Ernst 2003). Clutches from multiple females may be laid together in communal nest sites (Blanchard 1942, Gilhen 1970). *Diadophis punctatus* eggs are 16–44 mm long (mean 25 mm, $n = 108$) and hatchlings are 7.6–18.8 cm TL (mean 12.4 cm, $n = 120$; Ernst and Ernst 2003). *Diadophis punctatus regalis* eggs and hatchlings are likely at the larger end of the spectrum. A field-collected 60 cm SVL female *D. p. regalis* from Arizona contained three large eggs (mean length 44 mm, mean width 11.3 mm) that hatched after 52 days of incubation (Vitt 1975). The neonates were 16.9–18.8 cm long (mean 18 cm). Gehlbach (1965) reported one female *D. p. regalis* carrying 18 eggs. Estimates for size at maturity for *D. punctatus* range from 17.8 to 18 cm (Wright and Wright 1957, Myers 1965), but given that hatchling *D. p. regalis* can be this large, they likely mature at a larger size. Development times to maturity in *D. punctatus* can take 1–3 years depending on locality (Fitch 1975, Degenhardt et al. 1996).

Diadophis punctatus can often be found in aggregations under cover objects (Ernst and Ernst 2003), and some populations make spring and fall migrations to and from hibernacula. *Diadophis punctatus regalis* from the Rocky Mountains in Utah at 1580 m elevation showed communal denning and repeated use of the same hibernacula in multiple years (Parker and Brown 1974). It is unknown whether California populations of *D. p. regalis* also show this behavior. Field-active body temperatures across several populations of *D. punctatus* range from 2.0°C to 34.4°C (Clarke 1958, Brattstrom 1965, Fitch 1975, Mitchell 1994).

Diadophis punctatus regalis is a mildly venomous rear-fanged colubrid snake, using enlarged posterior teeth to deliver venom to prey such as snakes and lizards (Gehlbach 1974, Anton 1994, Hill and Mackessy 2000, O'Donnell et al. 2007). In addition to subduing prey, copious salivation has been observed as a defensive response in *D. p. regalis* (Blanchard 1942). While *D. punctatus* is a generalized predator, southwestern populations, including *D. p. regalis,* have a diet composed of proportionately more reptiles (Gehlbach 1974) than other populations, which tend to consume a greater fraction of amphibians and earthworms (Ernst and Ernst 2003).

Habitat Requirements

In California, *Diadophis punctatus regalis* appears to be restricted to riparian areas surrounding desert springs. Snakes have been found in Death Valley in Inyo County in heavy riparian vegetation within 5 m of surface water (Emmerich and Cunningham 2003) and at Pachalka Spring, Clark Mountain, San Bernardino County, near the spring head (Wood and Richmond 2003). Outside of California, *D. p. regalis* have been found in evergreen woodland, deciduous woodland, desert grassland, oak-juniper, and succulent desert habitats such as sotol-agave and juniper-agave (Gehlbach 1974).

Distribution (Past and Present)

In California, the documented range of *Diadophis punctatus regalis* is extremely small. It is known from only a few isolated populations in the Clark, Grapevine, Mute, and Providence Mountains in the Mojave Desert. However, there is a strong possibility that undetected populations exist, particularly at additional springs in the mountain ranges where this taxon occurs. Outside of California, the subspecies occurs in parts of Idaho, Utah, Nevada, Arizona, New Mexico, Texas, and Mexico (Ernst and Ernst 2003). The Great Basin lineage defined by Fontanella et al. (2008) ranges from southern New Mexico north to southern Idaho and between roughly the Sierra Nevada Mountains in the west and the Guadalupe Mountains in the east. Much of the Great Basin clade is restricted to patches of suitable mesic environments surrounded by less hospitable xeric habitats (Fontanella et al. 2008).

Trends in Abundance

No population estimates are available for California populations. Declines may have occurred near Fort Piute in the Mute Mountain Range (R. Fisher, pers. comm.).

Nature and Degree of Threat

The small and patchy distribution of *Diadophis punctatus regalis* in California makes it at risk of extirpation. Its dependence on rare desert spring habitats is an additional risk factor, particularly because these spring habitats are threatened by overexploitation of groundwater resources. Because of their reliance on mesic habitats in an arid matrix, *D. p. regalis* may also be sensitive to climate changes that affect the timing and amount of precipitation. While there is a large degree of uncertainty in how rainfall patterns will change within its range, most studies predict decreases in mean annual rainfall of up to 40% (reviewed in PRBO 2011). In addition to decreases in mean annual rainfall, the number of extremely hot days where temperatures exceed the long-term 95th percentile is expected to increase by roughly 30 days a year (Bell et al. 2004). If conditions become warmer and drier, this could negatively impact *D. p. regalis* habitat.

Status Determination

Diadophis punctatus regalis has an extremely small range in California and is dependent upon a rare habitat type that is sensitive to human use of groundwater in the desert. However, extirpations have not been well documented, and we have virtually no information about the number or status of populations occurring in California. Because of this, we choose not to define a priority at this time.

Management Recommendations

Protecting desert springs and associated mesic habitat patches is a key requirement for the continued existence of this species. Minimizing use of water from desert spring sites will help maintain habitat for *Diadophis punctatus regalis*. It is difficult to make further management recommendations given the lack of information on this taxon's ecology in California or similar habitats.

Monitoring, Research, and Survey Needs

Given the inhospitable nature of habitat between occupied patches, long-distance dispersal events probably do not occur, and it is unlikely that populations in California are demographically connected. The extremely isolated nature of most of their desert habitat and their relatively short surface activity period also make increased surveys an important priority for this taxon. Increased genetic sampling would help determine patterns of connectivity between fragmented southwestern populations, and we strongly encourage all field surveys to take nonlethal tissue samples of any specimens that are encountered. Given that no population study has been conducted within California and the unique habitat requirements of the taxon, additional work quantifying the basic ecology and life history of *Diadophis punctatus regalis* is badly needed.

SAN JOAQUIN COACHWHIP

Masticophis flagellum ruddocki Brattstrom and Warren 1953

Status Summary

Masticophis flagellum ruddocki is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 53% (58/110). It was previously considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Masticophis flagellum is a large (91–260 cm TL) slender colubrid snake with smooth scales and a large head and eyes (Stebbins 2003). The species is distributed across the southern portion of the United States from Florida to California, with western subspecies tending to be smaller than eastern animals. Brattstrom and Warren (1953) reported that their largest specimen of *M. f. ruddocki* was 170 cm TL. Coloration is highly variable within *M. flagellum*. The subspecies *M. f. ruddocki* has a tan, olive-brown, or yellowish-brown dorsal color and lacks the dark head and neckbands characteristic of other subspecies. The ventral coloration is light tan or yellow, with a pink or orange cast under the tail (Jennings and Hayes 1994a, Stebbins 2003). The scales on the tail are often described as having a "braided" appearance (Stebbins 2003).

San Joaquin Coachwhip: Risk Factors

Ranking Metric (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	10
v. Ecological tolerance (10)	3
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	7
Total Score	58
Total Possible	110
Total Score/Total Possible	0.53

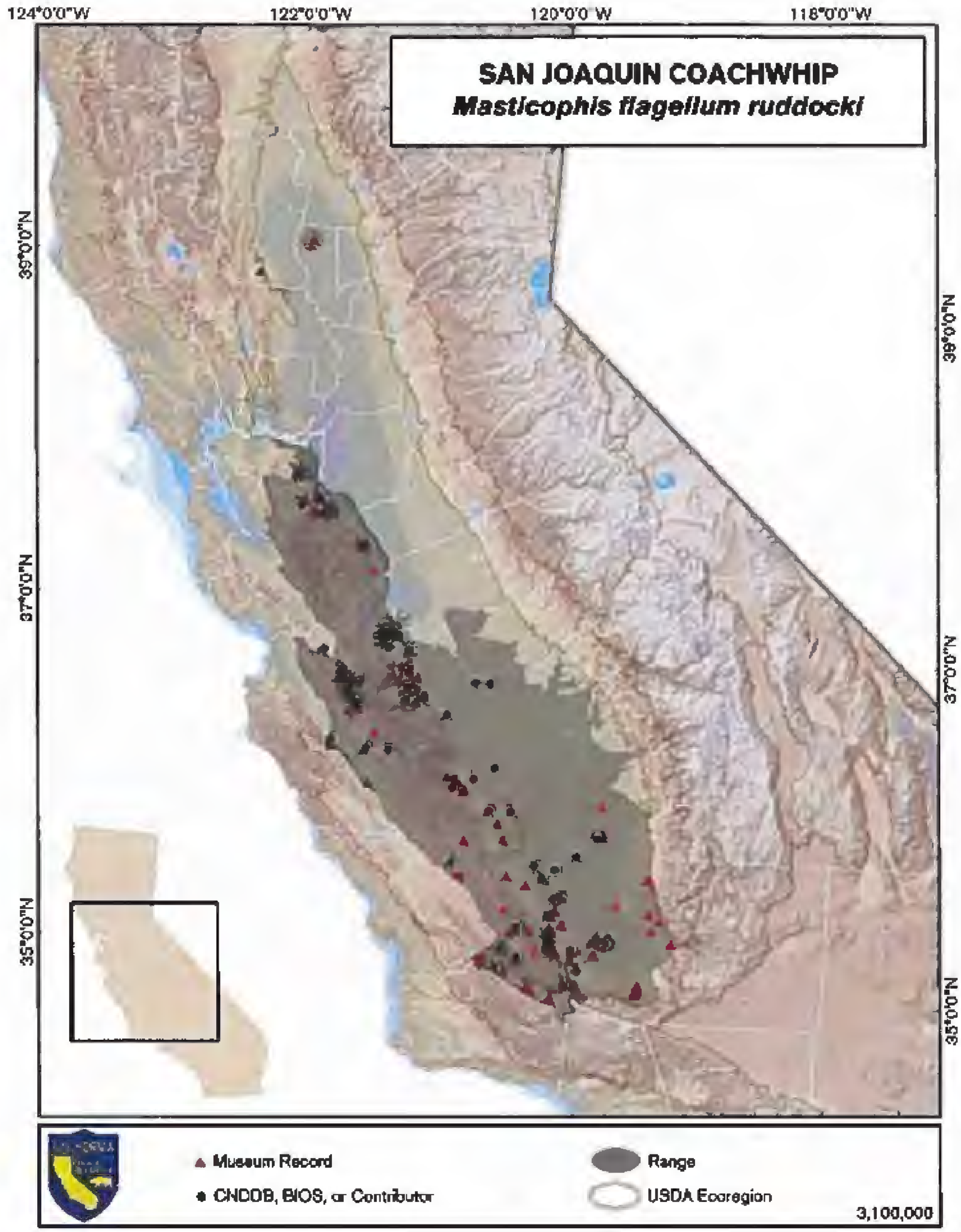


PHOTO ON PREVIOUS PAGE: San Joaquin coachwhip, Kern County, California. Courtesy of Jeff Lemm.

Taxonomic Relationships

Brattstrom and Warren (1953) described *Masticophis flagellum ruddocki* as a subspecies based on morphological characters including dorsal coloration, the lack of dark neckbands, and a relatively low number of subcaudal scales. Analysis of mitochondrial DNA from California populations supported the uniqueness of *M. f. ruddocki*, corroborating the morphological data (Mitrovich 2006).

Life History

Very little is known about the life history of *Masticophis flagellum ruddocki*. In general, *M. flagellum* is an extremely active diurnal snake that prefers warm temperatures (Brattstrom 1965, Hammerson 1977). Home ranges are suspected to be large, but no movement data are available for this subspecies (R. Hansen, pers. comm. in Jennings and Hayes 1994a). Preference for warm temperatures results in late-season emergence (April May), and daily surface activity corresponds to the warmest parts of the day (Hammerson 1977). Data from red coachwhips (*M. f. piceus*) in the Mojave Desert found body temperatures as low as 13.9°C when inactive in burrows to a high of 40.8°C while actively moving (Secor 1995). Mating is thought to take place in May, with oviposition occurring in June or July (Jennings and Hayes 1994a). Adults may cease surface activity and retreat to mammal burrows as early as August (pers. obs. in Jennings and Hayes 1994a).

Like other members of the *M. flagellum* complex, the diet of *M. f. ruddocki* is presumably generalized on vertebrates, including large prey like antelope squirrels (*Ammospermophilus nelsoni*), blunt-nosed leopard lizards (*Gambelia sila*), and whiptails (*Aspidoscelis tigris*) (Montanucci 1965, Tollestrup 1979; S. Barry, pers. comm.; R. Hansen, pers. comm.; S. Sweet, pers. comm. in Jennings and Hayes 1994a).

Habitat Requirements

Masticophis flagellum ruddocki occurs in open, dry areas with little or no tree cover (Morafka and Banta 1976). Valley grassland and saltbush scrub habitats are used in the western San Joaquin Valley (Montanucci 1965, Banta and Morafka 1968, Tollestrup 1979, Sullivan 1981; pers. obs. in Jennings and Hayes 1994a). Spring road cruising surveys from 1972 to 1979 in eastern Alameda and western San Joaquin Counties found *M. f. ruddocki* in grassland and transitional habitat but not in mixed oak chaparral woodland (Sullivan 1981). *Masticophis flagellum ruddocki* will climb into bushes, apparently to scan for predators and prey or to seek cover (Cunningham 1955, Stebbins 2003). Mammal burrows are used for overwintering and possibly also for oviposition (Jennings and Hayes 1994a).

Distribution (Past and Present)

Masticophis flagellum ruddocki is endemic to California, with a small range extending from Arbuckle, Colusa County, in the Sacramento Valley south to the Kern County portion of the San Joaquin Valley and west into the inner South Coast Ranges (Brattstrom and Warren 1953, Jennings and Hayes 1994a). A disjunct population occurs in the Sutter Buttes (Hayes and Cliff 1982).

Much of this subspecies' historic range has undergone dramatic land use changes from grassland to intensive agriculture in the Central Valley. *Masticophis flagellum ruddocki* is thought to be sensitive to disturbance and does not persist in cultivated areas (Ernst and Ernst 2003; S. Barry, pers. comm.). It has therefore suffered a severe range contraction in its Central Valley range.

Trends in Abundance

Though neither historical nor current abundance estimates are available, we suspect that the conversion of historical habitat to row crop agriculture and urban development has resulted in lower abundances than in preagricultural times.

Nature and Degree of Threat

Habitat loss and fragmentation due to agriculture and urbanization are the major threats to

Masticophis flagellum ruddocki. As with other diurnally active, highly mobile snakes, road mortality is probably a significant source of mortality, although its overall impact requires more study. The greatest potential threats from climate change are due to changes in fire regime. In the more coastal parts of the range, the area burned is expected to increase by up to 50% (Fried et al. 2004, Lenihan et al. 2008), and the probability of large (>200 ha) fires is predicted to increase (Westerling and Bryant 2008). Modest decreases in the probability of large wildfires are expected in the San Joaquin Valley. How *M. f. ruddocki* may respond to increased fire needs more study. Fire may have direct mortality effects on snakes and negative effects on prey populations but may also benefit *M. f. ruddocki* by increasing or maintaining the availability of open habitat. Under climate change projections, grassland habitat is expected to increase by up to 140% in the coastal part of the range, with little change in vegetation expected in the Central Valley (Lenihan et al. 2008, PRBO 2011). These vegetation shifts may result in additional potential habitat for *M. f. ruddocki*.

Status Determination

Masticophis flagellum ruddocki is a California endemic with a small range and is restricted to a heavily disturbed part of the state, resulting in a Priority 2 designation.

Management Recommendations

The lack of basic ecological information on this subspecies needs to be addressed before any meaningful management can be accomplished. At a minimum, remaining large habitat fragments and connectivity among fragments must be protected if the species is to persist.

Monitoring, Research, and Survey Needs

Although additional work on all aspects of its ecology, demography, and population genetic differentiation would be useful, information on reproductive biology, movement ecology, population sizes, and fragmentation is key priority for future work. Some large habitat fragments are currently protected from some kinds of human disturbance (e.g., the Carrizo Plain National Monument) and provide suitable areas to begin studying basic ecology and habitat requirements in this taxon.

BAJA CALIFORNIA COACHWHIP

Masticophis fuliginosus (Cope 1895a)

Status Summary

Masticophis fuliginosus is a Priority 3 Species of Special Concern, receiving a Total Score/Total Possible of 45% (50/110). This species has not previously been considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Masticophis fuliginosus is a large slender colubrid snake with smooth scales, reaching up to 170 cm in TL (Grismer 2002). Two color phases exist in the species, a light morph and a dark morph. The light morph is yellowish, tan, or gray above with dark zigzagging bands on the body and dark neckbands. The dark morph has a dark gray brown, golden brown, or black dorsal ground color, and sometimes has distinguishable dark neckbands (Wilson 1971, Grismer 2002). Dark morph animals can be uniformly dark above, or the scales on the sides of the body can have pale edges, giving the appearance of narrow lines (Wilson 1971). The venter is light colored with brown spots (Wilson 1971, Grismer 2002).

Masticophis fuliginosus can be distinguished from congeneric southern California snakes by geographic range, as there is little overlap with

Baja California Coachwhip: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	0
v. Ecological tolerance (10)	0
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	7
Total Score	50
Total Possible	110
Total Score/Total Possible	0.45

PHOTO ON PREVIOUS PAGE: Baja California coachwhip, Baja California, Mexico. Courtesy of Brad Shaffer.

other species, and by color. *Masticophis flagellum piceus* is reddish to pinkish above, with dark bands at the neck. California whipsnakes (*M. lateralis*) have a conspicuous light stripe on either side. Racers (*Coluber constrictor*) are pale green or dark above (brown, olive, or bluish) but have unmarked white to yellow ventral surfaces (Stebbins 2003).

Taxonomic Relationships

We follow Grismer's (2002) proposal that *Masticophis fuliginosus* is a full species, rather than a subspecies of *M. flagellum*. This arrangement is based on a lack of intergradation with neighboring *M. f. piceus* (Wilson 1971, Grismer 1994). Analysis of a single mitochondrial DNA gene from 229 *M. flagellum* individuals (including 30 *M. fuliginosus*) from 30 localities in southern California supported the genetic distinctiveness of *M. fuliginosus* (Mitrovich 2006). However, 4 out of 30 snakes identified in the field as *M. fuliginosus* had mitochondrial DNA sequences that were most closely related to *M. f. piceus* (Mitrovich 2006). This could be due to hybridization or incorrect identification in the field, as the study was conducted on tissues without voucher specimens. Further resolution of this problem with multiple nuclear DNA markers would likely help to clarify the taxonomic status of this species.

Life History

Very little is known about the life history of *Masticophis fuliginosus* in California. In general, *Masticophis* are extremely active diurnal snakes that prefer warm temperatures (Brattstrom 1965, Hammerson 1977). In southern Baja California, *M. fuliginosus* can be active year-round, but in the northern part of the range, they tend to be inactive in winter and emerge in mid-March (Grismer 2002). Activity in San Diego was observed to be greatest in spring and summer and greatly reduced in the fall (Mitrovich et al. 2009). Mating has been observed in northern Baja California in late April, and hatchlings have been seen in early August (Grismer 2002).

Radiotelemetry of 24 snakes in two reserves in San Diego County found large variation in home range size, from roughly 11 to 130 ha (Mitrovich et al. 2009). Variation in home range size was largely due to habitat availability, with smaller home ranges in smaller habitat fragments. No differences in potential prey were detected among sites where snakes had different home range sizes. The diet of *M. fuliginosus*, like its close relative *M. flagellum*, is broad and includes a variety of vertebrate prey such as lizards, snakes, birds, and mammals (Cliff 1954, Grismer 2002).

Habitat Requirements

Masticophis fuliginosus is a habitat generalist throughout Baja California, Mexico, and is common in marshlands, coastal sand dunes, rocky arroyos and hillsides, thorn forests, sandy flats, and scrub vegetation (Linsdale 1932, Cliff 1954, Leviton and Banta 1964, Bostic 1971, Welsh 1988, Grismer 2002). In California, *M. fuliginosus* occurs mainly in coastal sand dunes, shrubland, and grassland, and is most commonly observed foraging in bushes and shrubs (Linsdale 1932, Bostic 1971, Welsh 1988, Grismer 2002). Hollow stumps of plants such as agave and yucca are used as retreats (Bostic 1971, Grismer 2002).

Distribution (Past and Present)

Masticophis fuliginosus has a very small geographic range in California, occurring in a small area of San Diego County near the United States–Mexico border (Wilson 1973). The range of the species extends over most of the Baja California peninsula, including some small offshore islands (Wilson 1973, Grismer 2002).

A resurvey of Klauber's (1939) sites in southern California found that *M. fuliginosus* was absent from some previously occupied sites (Fisher and Case 2000, Case and Fisher 2001; R. Fisher, pers. comm.), suggesting that the species has declined in the last seven decades. However, the full extent of local extirpations is unknown.

Trends in Abundance

While data on abundance across the range are not available, some reductions in abundance are likely to have occurred due to development, road mortality, and fragmentation. This species may be particularly prone to death from automobiles given its large home range size and high level of diurnal activity (Mitrovich et al. 2009).

Nature and Degree of Threat

Masticophis fuliginosus in California are mainly threatened by habitat loss, fragmentation, and road mortality due to development, as well as the inherent demographic threats associated with a very small geographic range. Climate change may affect *M. fuliginosus* through changes in fire regime and vegetation shifts. However, both increases and decreases in fire have been predicted, and there is little consensus because of the difficulty in modeling Santa Ana weather events in southern California (Westerling et al. 2004, Westerling and Bryant 2008). How *M. fuliginosus* may respond to changes in fire regime is unknown. Climate change is predicted to decrease the availability of chaparral and shrubland by up to 44%, while grassland is predicted to increase by up to 390% in southern California (Lenihan et al. 2008, PRBO 2011). Though *M. fuliginous* also uses grassland habitat, large losses in shrub habitat may negatively affect this species. Finally, development along the border may effectively isolate the population that occurs in California, making it more susceptible to decline than it otherwise would be.

Status Determination

Masticophis fuliginosus has an extremely small range in California that occurs entirely in an area with substantial urban, military, and agricultural development. However, the extent of extirpation and population decline is poorly documented, resulting in a Priority 3 Species of Special Concern designation.

Management Recommendations

Protection of remaining habitat in San Diego County is necessary to prevent further declines or extirpations. Minimizing urban edge effects by creating habitat buffers may benefit populations, particularly those living in small habitat fragments (Mitrovich et al. 2009). Given the very small range of the species, it may be possible to reduce road mortality with wildlife tunnels and associated drift fences installed beneath high-traffic roads in key areas important for population connectivity.

Monitoring, Research, and Survey Needs

Drift fence arrays with funnel traps have been successfully used to document the presence/absence of *Masticophis fuliginosus* in California (Fisher and Case 2000), and mark recapture data to establish population sizes are essential for future management. Additional genetic data would complement existing mitochondrial DNA data and radiotelemetry research, respectively. Specifically, such data are needed to further resolve the taxonomic status of this snake and to provide information on landscape-level population structure. The efficacy of road-crossing structures should be investigated for this species.

COAST PATCH-NOSED SNAKE

Salvadora hexalepis virgultea Bogert 1935

Status Summary

Salvadora hexalepis virgultea is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 54% (46/85). During the previous evaluation, it was also designated as a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Salvadora hexalepis virgultea is a medium-sized (to 115 cm TL) snake, with an enlarged rostral scale, large eyes, and a light middorsal stripe 1.5–2 scale rows in width (Jennings and Hayes 1994a, Stebbins 2003). The dorsal stripe is yellowish to grayish and extends from the tail to the rear of the neck region (Bogert 1935, Stebbins 2003). The sides of the body are dark brown, and the head is olive or brown (Bogert 1935, Perkins 1938). Ventral coloration is generally a dull white, often with an orange wash that is more prominent toward the underside of the tail (Stebbins 2003, Lemm 2006). Usually only a single supralabial contacts the eye (Bogert 1935).

The presence of both a conspicuously enlarged rostral scale and a middorsal stripe easily differentiate this species from all other snakes within its range. Along the eastern edge

Coast Patch-Nosed Snake: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	0
iv. Endemism (10)	3
v. Ecological tolerance (10)	3
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	10
Total Score	46
Total Possible	85
Total Score/Total Possible	0.54

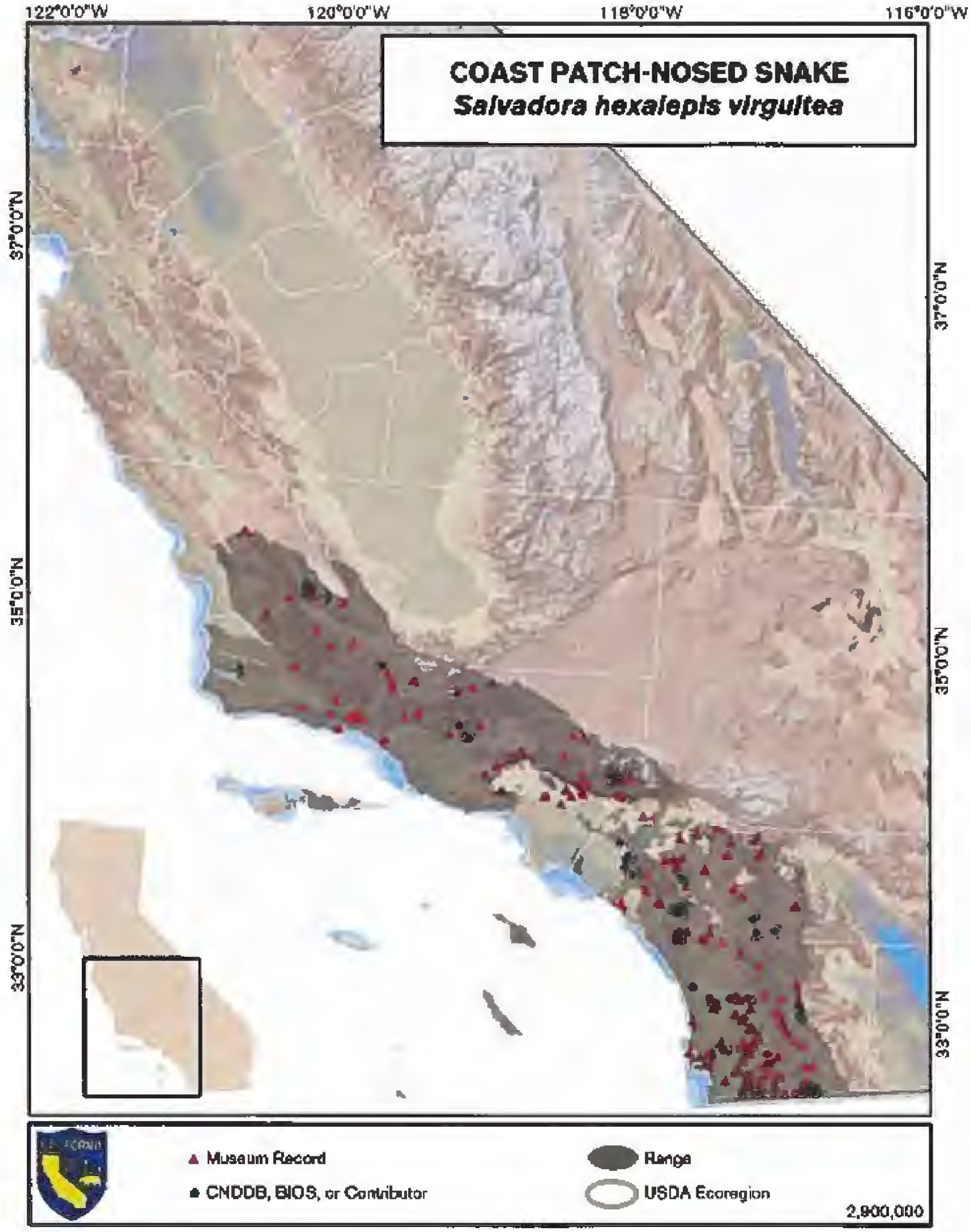


PHOTO ON PREVIOUS PAGE: Coast patch-nosed snake, San Diego County, California. Courtesy of John Andermann.

of its range, *S. h. virgultea* is parapatric with the Mojave patch-nosed snake (*S. h. mojavensis*) and the desert patch-nosed snake (*S. h. hexalepis*), and it intergrades with both (Bogert 1945). In *S. h. mojavensis*, the supralabials usually do not reach the eye, and the dark lateral coloration is sometimes less pronounced and may be somewhat discontinuous. *Salvadora hexalepis hexalepis* usually has one supralabial reaching the eye but has a wider dorsal stripe (usually three scale rows wide) than *S. h. virgultea*. The spotted leaf-nosed snake (*Phyllorhynchus decurtatus*) occurs along the eastern edge of *S. h. virgultea*'s range and also has an enlarged rostral scale. This species differs in having a light-brown spotted pattern on the dorsal surface and attaining much smaller adult sizes (to 51 cm TL) (Stebbins 2003).

Taxonomic Relationships

Salvadora hexalepis virgultea was first recognized primarily on the basis of coloration, although some scalation characters also differentiate it from other subspecies of the *S. hexalepis* complex (Bogert 1935, Bogert 1945). Genetic analysis of differentiation among the subspecies has not been undertaken and represents a clear need for future research. The placement of *Salvadora* within the colubrine phylogeny is also uncertain and requires clarification.

Life History

Salvadora hexalepis virgultea is a medium-sized, active, diurnal snake, yet remains exceedingly poorly known. Here, we use information from California populations where we can, but largely rely on information from parts of the range outside of California as well as from other species and subspecies within *Salvadora* and assume that many aspects of the life history are similar among members of the complex. This subspecies is most active in May and June, and its normal seasonal activity period appears to extend from March until October. In Ventura County, warm (and presumably active) individuals have been found under stones between January and March (S. Sweet, pers. comm.). Individuals found in December and January in gopher burrows and woodrat nests were cold and presumably dormant (S. Sweet., pers. comm.). Specimens have been documented in the literature in all months except January and February (Klauber 1939, Grismer 2002), and more recent (1995–2011) USGS pitfall trap data have documented captures in January (three records) and February (one record; C. Rochester, unpublished data). Breeding has been observed in late April, with hatchlings appearing between mid-July and October (Klauber 1931, Grismer 2002), perhaps indicating that multiple clutches may be laid in a single year. Other members of the genus are oviparous, and we expect that *S. h. virgultea* is as well, although eggs have never been described (Wright and Wright 1957).

Salvadora primarily eats lizards and probably specializes on whiptails (*Aspidoscelis* spp.) (Cunningham 1959b, Grismer 2002, Lemm 2006). The diet of *S. h. virgultea* has not been described, but other members of the complex feed nearly exclusively on members of the genus *Aspidoscelis*, with other small lizards being taken only when they are moving rapidly, as *Aspidoscelis* tends to do (Bogert 1939). Several members of the complex are also known to feed on other small lizards, mammals, and snakes (Stebbins 2003). The enlarged rostral scale may be an aid for digging and possibly a specialization for feeding on lizard eggs (Grismer 2002, Lemm 2006), although this behavior has never been described in *S. h. virgultea*. *Salvadora* species have enlarged rear teeth, suggesting that they may envenomate prey. The saliva of *Salvadora* has not been studied, though Grismer (2002) observed an *A. tigris* that died soon after being bitten by a Baja California patch-nosed snake (*S. h. klauberi*).

The movement ecology of *S. h. virgultea* is unknown. Like other members of the genus, it is a fast and active species that is probably active at higher temperatures than most other diurnal colubrids (Jacobson and Whitford 1971). The species readily climbs and has been found foraging in brush, off the ground (Grinnell and Grinnell 1907, Lemm 2006; S. Sweet, pers. comm.). The species probably ranges widely, as do other large diurnal colubrids.

Habitat Requirements

Salvadora hexalepis virgultea shows an apparent preference for brushy chaparral habitat (Bogert 1935, Grismer 2002). Klauber (1939) located 35 specimens in "Heavy brush, Chaparral" habitat, which was more than the number found in all other habitat types combined. In Baja California, they are also often seen in riparian areas in the vicinity of Tecate (Grismer 2002). In California unpublished data suggest that this species has a preference for chamise and red shank and often basks at or near the tops of bushes (S. Sweet, pers. comm.). Schoenherr (1976) noted that this taxon was widespread but uncommon on the Pacific slope of the San Gabriel Mountains and that it may prefer coastal sage scrub to chaparral, at least in this area. He recorded a sighting at 830 m elevation, the highest site for coastal sage scrub habitat in the San Gabriels. At two burn sites in southern California, brush was reduced, *Aspidoscelis* numbers increased, and *S. h. virgultea* numbers remained stable or increased (C. Rochester, pers. comm.), suggesting that prey abundance may be at least as important as vegetation per se. If the species is a wide-ranging predator, it may be susceptible to habitat fragmentation. Because a large component of its diet probably consists of *Aspidoscelis* species, *S. h. virgultea* may be susceptible to decline in areas where *Aspidoscelis* are declining. The two species found within its range, the orange-throated whiptail (*A. hyperythra*) and the coastal whiptail (*Aspidoscelis tigris stejnegeri*), are both under threat. *Aspidoscelis hyperythra* is a Watch List species, while *A. t. stejnegeri* is a Priority 3 Species of Special Concern (see accounts, this volume).

Distribution (Past and Present)

The historical distribution ranges from Ventura and Los Angeles Counties south to the United States–Mexico border and south to the vicinity of El Rosario, Baja California, Mexico. It ranges from sea level along the coast up to 2130 m (Jennings and Hayes 1994a, Lemm 2006). The eastern edge of the range extends to the vicinity of Campo, San Diego County; Banning, Riverside County; and San Bernardino, San Bernardino County (Bogert 1935).

Today, the species is declining or absent from large areas of the Los Angeles basin and along the coast to San Diego (Jennings and Hayes 1994a; R. Fisher, pers. comm.). Survey data are available from Torrey Pines State Park where this species was formerly present. It has not been re-documented there despite intensive trapping efforts (Wells 1998).

Trends in Abundance

No historical or current abundance data exist for this taxon, and anecdotal reports are rare and inconclusive. Records in southern California are infrequent (Lemm 2006), although this taxon may be more commonly encountered in riparian areas south of the United States–Mexico border (Grismer 2002). The species may have historically been rare in California. Klauber (1924) reported only two specimens despite intensive collecting efforts spanning 2 years and stated that "it seems to be uncommon." Bogert (1930) also noted that it was "uncommon in the chaparral." USGS pitfall data for San Diego, Orange, and Los Angeles Counties collected between 1995 and 2011 indicate that 2.6% of snake captures were *Salvadora hexalepis virgultea* (123 captures out of 4680 total snake captures), compared to 3.8% (61/1601) of captures for daytime road driving records for snakes recorded by Klauber (1939). These data confirm that this taxon is relatively uncommon and may suggest a moderate decline over the last 60 years.

Nature and Degree of Threat

The declines in *Salvadora hexalepis virgultea* are most likely due to the conversion of the preferred brushy habitat to other vegetation types. Development of rangeland, combined with increasingly frequent and intense wildfires, has converted large blocks of chaparral habitat to grassland (R. Fisher, pers. comm.), which appears to be unsuitable for this species (Jennings and Hayes 1994a; S. Sweet, pers. comm.). Climate change may exacerbate the intensity of wildfires in southern California (Cayan et al. 2008b), although current models range from a 29% decrease to a 28% increase in wildfires in the region (PRBO 2011). However, climate models for 2070 project an estimated 38–44% decrease in the chaparral/coastal scrub habitat preferred by this species. Ongoing urbanization in the populated areas within this taxon's range is also destroying, degrading, and fragmenting large areas of remaining habitat. *Salvadora hexalepis virgultea*'s probable preferred prey, *Aspidoscelis* lizards, are also in decline, which could cause cascading declines in snake populations. Finally, diurnally active widely foraging snakes are particularly affected by road mortality, and the volume of vehicular traffic in much of its range is large and increasing.

Status Determination

Salvadora hexalepis virgultea has a relatively small range in California and has disappeared from significant areas centered in the southern portion of its range where it was formerly known. It continues to lose habitat, which is also causing declines in a significant component of its prey-base. Habitat loss, due to direct anthropogenic changes, climate-change-driven habitat loss, and wildfire, is unlikely to stop in the near future, which we expect will cause further declines. For all of these reasons, a Priority 2 designation is justified.

Management Recommendations

The primary management goal for *Salvadore hexalepis virgultea* should be to protect large, intact patches of brushy chaparral and/or coastal sage scrub habitat that support this snake. Ideally, these patches should be those that are least likely to be directly affected by future climate change. Pending further study of the species' movement ecology, habitat protection efforts should focus on remaining large blocks of intact habitat. Habitat fragmentation from roads is a key issue in these efforts. Road overcrossings, if installed for other taxa, may function as a means to avoid habitat fragmentation. A key element of effective management is to maintain large, healthy populations of *Aspidoscelis hyperythra* and *Aspidoscelis tigris stejnegeri*, since they appear to form the primary prey base of this snake. As declines in those species are also linked to habitat loss, management efforts among these species should be coordinated.

Monitoring, Research, and Survey Needs

Much remains to be learned about the life history and ecology of *Salvadora hexalepis virgultea* in California. Surveys employing pitfall traps, snake traps, and daytime surveys should be undertaken to establish baseline abundance data in remaining populations, and to ascertain whether these or other survey methods are potentially biased for this species, as has been suggested by some biologists. Because these snakes are infrequently encountered, power analyses are particularly important to determine how sensitive the surveys would be in detecting declines.

Autecological research focusing specifically on diet requirements, habitat utilization, and their interaction are badly needed, and the lack of this basic knowledge undermines our ability to effectively manage this snake. Specifically, the extent to which this taxon specializes on *Aspidoscelis*, as opposed to utilizing alternative prey, needs to be determined. Basic data on home range size and movement patterns, and how they may vary as a function of vegetation, are entirely lacking, and some anecdotal observations suggest that they may vary across the species' range. These data are necessary to determine what habitat blocks are most important and how large they should be to effectively conserve this taxon.

Variation within the *S. hexalepis* complex has not been examined since the initial species description, and a genetic analysis has never been undertaken. A three-pronged genetic analysis is critical. First, a systematic characterization of the *Salvadora* species complex should be undertaken utilizing multiple independent nuclear markers. This will serve to clarify the reality of species and subspecies as valid evolutionary units and confirm their respective boundaries and range limits. Second, a phylogeographic study, using many independent markers, within *S. h. virgultea* is needed to identify large-scale management units within the taxon. Finally, landscape genetic studies that identify migration corridors between fragmented blocks of habitat will inform our understanding of movement ecology in this species. These would also provide an estimate of effective population sizes, augmenting ecological studies of census population sizes in protected and unprotected areas.

TWO-STRIPED GARTER SNAKE

Thamnophis hammondii (Kennicott 1860)

Status Summary

Thamnophis hammondii is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 57% (63/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Thamnophis hammondii is a medium-sized snake (102 cm TL) with keeled scales and a head slightly wider than its body (Stebbins 2003). It is called the two-striped garter snake because it lacks the longitudinal middorsal stripe that typifies many garter snakes. The middorsal stripe is either entirely absent or represented only by a nuchal spot at the base of the head (Fitch 1948, Stebbins 1985). Color is highly variable in this species, but there are two primary color morphs: striped/spotted and striped/non-spotted (Larson 1984, Stebbins, 2003). Both morphs have yellowish to gray stripes on each side with a ground color of olive, brown, or brownish gray, and both lack any red coloration dorsally or laterally. The ventral coloration is dull yellowish to orange red or salmon, with or without slight dusky markings (Stebbins 2003). The striped/spotted morph

Two-Striped Garter Snake: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	15
iii. Population concentration/migration (10)	10
iv. Endemism (10)	3
v. Ecological tolerance (10)	3
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	63
Total Possible	110
Total Score/Total Possible	0.57

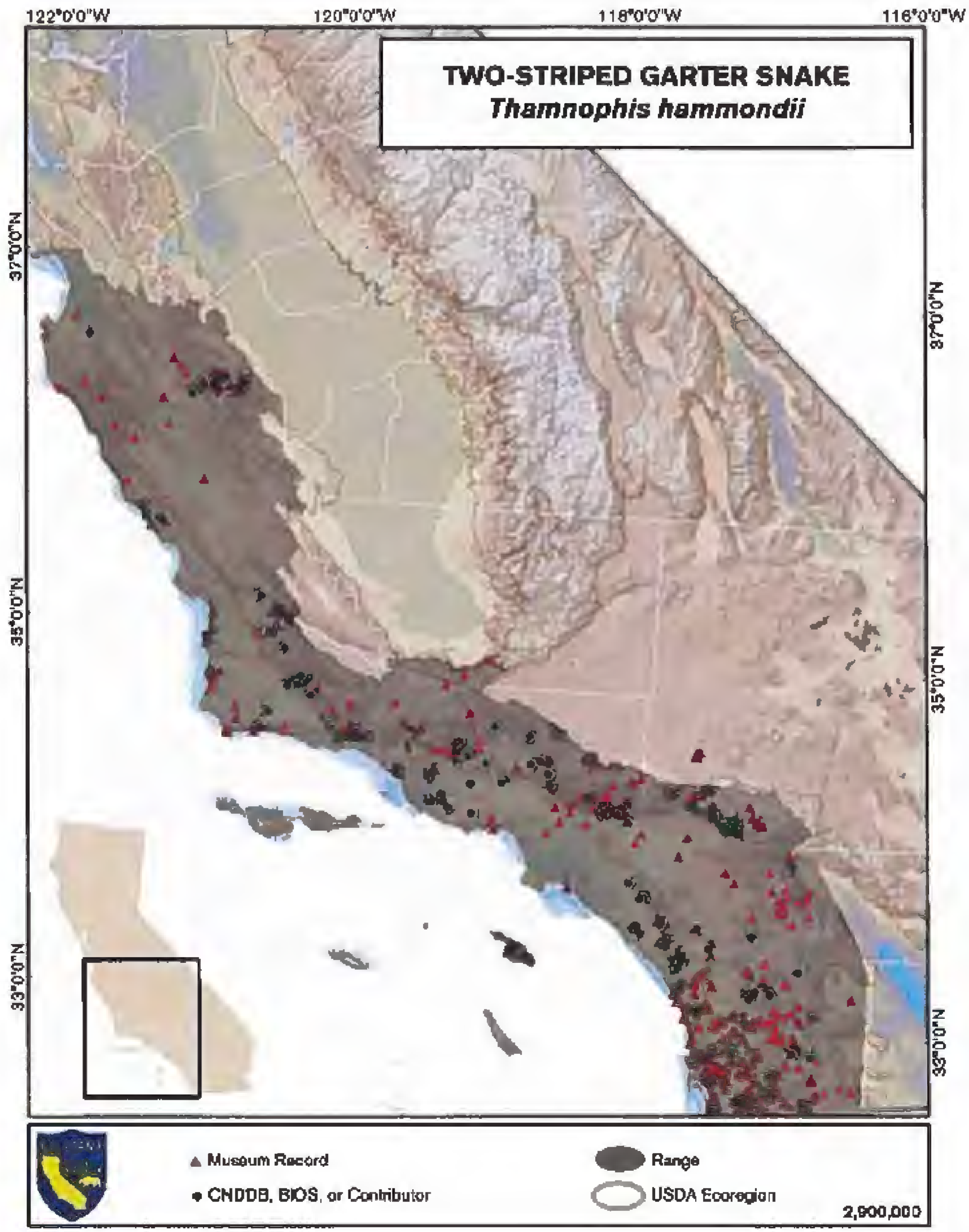


PHOTO ON PREVIOUS PAGE: Two-striped garter snake, Los Angeles County, California. Courtesy of Robert Hess.

has one or two rows of small, alternately spaced dark spots on each side of the dorsum between the lateral stripes (Fitch 1940, Fox 1951, Larson 1984, Stebbins 2003). The striped/non-spotted morph either lacks dark spots on the dorsum or only has very small ones next to the lateral stripes on the anterior part of the body (Larson 1984, Stebbins 2003). Other color variants include non-striped/spotted, with no lateral stripes and one or two rows of dark spots on each side, sometimes appearing checkered, and non-striped/non-spotted (Brown 1980, Larson 1984, Stebbins 2003). A melanistic morph, sometimes with obscure lateral stripes and/or spots, occurs along the outer coast from Oceano to San Simeon State Park in San Luis Obispo County, and can be expected from Gaviota State Beach in Santa Barbara County to Monterey Bay (Bellemin and Stewart 1977, Larson 1984, Stebbins 2003). All color morphs exhibit varying degrees of light flecking dorsally due to whitish pigment on the inter-scale skin and margins of scales (Larson 1984). Dark green and dull red color morphs occur in northeastern Ventura County (Stebbins 2003).

The lack of a vertebral stripe and absence of red coloration on the head and sides distinguishes *T. hammondii* from the co-occurring aquatic garter snake (*T. atratus*), the western terrestrial garter snake (*T. elegans*), and the common garter snake (*T. sirtalis*).

Taxonomic Relationships

Like several other garter snakes, *Thamnophis hammondii* has a complex taxonomic history. This species has at various times been considered a subspecies of the Sierra garter snake (*T. couchii*) (e.g., Cooper 1870, Rossman 1979, Lawson and Dessauer 1979), the northwestern garter snake (*T. ordinoides*) (e.g., Grinnell and Camp 1917), and the western terrestrial garter snake (*T. elegans*) (e.g., Fitch 1948). Rossman and Stewart (1987) most recently elevated *T. hammondii* to full species status. McGuire and Grismer (1993) synonymized the Baja California Sur garter snake (*T. digueti*) with *T. hammondii*.

Life History

In California, *Thamnophis hammondii* can be active for much of the year and has been found from January through November (R. Hansen and R. Tremper, unpublished data in Rossman et al. 1996). Ervin and Fisher (2001) reported *T. hammondii* foraging and basking at a site in San Diego County from early February to October. Rathbun et al. (1993) conducted surveys in San Simeon State Park and nearby Pico Creek in San Luis Obispo County and found that large snakes (>30 cm) were most often observed in the summer, peaking in May and June, while smaller animals were seen from late August through early November. Surface activity appears to be strongly affected by the availability of surface water (E. Ervin and R. Fisher, unpublished data). Southern populations receive less rainfall and experience greater variation in rainfall, likely resulting in shorter and less predictable activity periods than northern populations.

Mating has been observed in the field in late March (Cunningham 1959b), and females are known to store sperm (Fox 1956, Stewart 1972). Like all members of the genus, *T. hammondii* is live-bearing, with litters produced from July to late October (Ernst and Ernst 2003). Hansen and Tremper (unpublished data in Rossman et al. 1996) documented an average of 15.6 offspring from 7 litters (range 3–36). Young were born in late July and August and were 20.3–21.7 cm TL (R. Hansen and R. Tremper, unpublished data in Rossman et al. 1996). Cunningham (1959b) found a 46.1 cm SVL female that contained 6 embryos. Another 64.9 cm SVL female contained 19 eggs (Cunningham 1959b). Males mature at 37.3 cm SVL and females at 38.8 cm (Wright and Wright 1957).

Cunningham (1966) reported a mean body temperature of 14 field-active individuals of 22.6°C (range 18.6–31.8°C). Five of these animals were swimming in water between 14°C and 27°C (Cunningham 1966). Inactive snakes found under cover objects had body temperatures ranging from 7.2°C to 23.6°C (Cunningham 1966).

One radiotelemetry study has collected data on the movement ecology of this species at San Simeon State Park (Rathbun et al. 1993). Activity ranges of radio-tracked snakes were greater and more distant from water in the winter than in the summer. Average summer activity ranges for seven adult females were 1498.9 ± 1847.6 m² (mean ± s.d.), although the duration of the study was short (range 4–29 days), and activity may have increased with more time. Average winter activity ranges for two females and one male were 3395.7 ± 4803.5 m² (mean ± s.d.), with animals tracked for 29–57 days. Average daily distance to water was 7.2 m in summer, compared to 98.8 m in winter (Rathbun et al. 1993).

Thamnophis hammondii is a generalized predator on a variety of prey including fish, fish eggs, frogs, salamanders, leeches, and earthworms (Van Denburgh 1897, Klauber 1931, Fitch 1940, Fitch 1941, Cunningham 1959b, Bell and Haglund 1978, Rathbun et al. 1993, Rodríguez-Robles and Galina-Tessaro 2006). This species will eat introduced prey, such as sunfish, African clawed frogs, and bullfrogs (Ervin and Fisher 2001, Mullin et al. 2004, Ervin and Fisher 2007).

Habitat Requirements

Thamnophis hammondii is among the most aquatic of the garter snakes and is often found in or near permanent and intermittent freshwater streams, creeks, and pools (Grinnell and Grinnell 1907, Fitch 1940; R. Hansen and R. Tremper, unpublished data in Rossman et al. 1996). Associated vegetation types include willow, oak woodlands, cedar, coastal sage scrub, sparse pine, scrub oak, and chaparral (R. Hansen and R. Tremper, unpublished data, in Rossman et al. 1996, Ernst and Ernst 2003). *Thamnophis hammondii* will also use artificial aquatic habitats such as cattle ponds (Jennings and Hayes 1994a, Ervin and Fisher 2001, Ervin and Fisher 2007).

Surveys in San Simeon State Park in San Luis Obispo County from July to December 1992 resulted in 45 snake sightings: 33.3% on land, 53.3% on banks, and 6.7% in the water (Rathbun et al. 1993). Almost all of the sightings (44/45) were in or near pooled water sources. Sixty percent of snakes were sighted in low vegetation (e.g., herbs and grasses), 28.9% in tall vegetation (e.g., cattails), 11.1% in open areas with no vegetation, and zero in wooded areas (e.g., willow; Rathbun et al. 1993).

Habitat and movement ecology may vary seasonally, although this requires further study. *Thamnophis hammondii* have been observed to concentrate their habitat use in vernal pools in the spring and in remnant pools formed from ephemeral creeks in the summer (R. Fisher, pers. comm.). Nine radio-tracked snakes in San Simeon State Park used streamside habitats more in the summer, while chaparral and grassland upland sites were used for overwintering (Rathbun et al. 1993). Ninety-five percent of diurnal locations of radio-tracked animals were on land, usually underground. Animals were underground, presumably in rodent burrows, in 87.9% of locations on land (Rathbun et al. 1993). Two of the animals had home ranges that overlapped Highway 1 (a major highway with heavy traffic), suggesting that potential road mortality may be a management issue. Although it is generally considered to be a very aquatic snake, these observations suggest that terrestrial upland habitats and rodent burrows can be important habitat components for *T. hammondii*.

Distribution (Past and Present)

Thamnophis hammondii occurs in California from Salinas, Monterey County, south along the coast into Baja California, Mexico, occurring in the South Coast, Peninsular, and Transverse ranges (Boundy 1990, Ely 1992, McGuire and Grismer 1993). Isolated populations also occur in Baja California Sur and on Santa Catalina Island (Brown 1980, Stebbins 2003). While *T. hammondii* occurs mostly west of the deserts in California, there are populations in some perennial desert slope streams in San Bernardino, Riverside, and San Diego Counties (Perkins 1938, Fitch 1940, Boundy 1990). The elevational range is from sea level to 2450 m (Atsatt 1913). Jennings and Hayes (1994a) estimated that *T. hammondii* has been extirpated from ~40% of its historic range in California during the second half of the twentieth century. This snake may be patchily distributed even when abundant suitable habitat is available. For example, snakes were readily observed at San Simeon Creek, San Luis Obispo County, in 1992, but similar habitat about 5 km away in Pico Creek had very few snakes, even though the latter experiences less human disturbance (Rathbun et al. 1993).

Trends in Abundance

Declines in abundance appear to be less severe in the southern compared to the northern part of the range, but few quantitative data are available to support this interpretation (Jennings and Hayes 1994a). Variation in abundance over time at a particular site may be partially explained by reduction in surface activity during drought periods and not necessarily reflect mortality and declines (E. Ervin and R. Fisher, unpublished data). *Thamnophis hammondii* were rare in Carmel River fish traps in 2003–2005 (S. Barry, unpublished data) and were never encountered in extensive fieldwork in and near the Hastings Reservation in the upper Carmel Valley from 1992 to 1998 (B. Shaffer, unpublished data). Jennings and Hayes (1994a) noted that *T. hammondii* was common only in San Diego County. However, other populations in the south may be robust, such as along the Santa Clara River in Los Angeles County, along Sespe Creek in Ventura County, and in the Angeles, Los Padres, and San Bernardino national forests (S. Barry, pers. comm.). The Santa Catalina Island population was reported as small (~30 individuals) and isolated in the 1970s (Brown 1980) and is suspected to have declined since (Jennings and Hayes 1994a).

Nature and Degree of Threat

Declines in the south are thought to be due to urbanization, reservoir construction, and flood control (Jennings and Hayes 1994a). Further north, declines are suspected to have been caused by a combination of factors including habitat modification by livestock, predation by introduced vertebrates, loss of native prey, and drought (Jennings and Hayes 1994a). However, negative interactions with nonnative species have not been well documented, and in some cases *T. hammondii* may benefit from availability of introduced prey. Reliance on aquatic habitat and prey may contribute to drought sensitivity in this species (Jennings and Hayes 1994a; R. Fisher, pers. comm.).

Under climate change, mean annual temperatures are projected to increase throughout the range of *T. hammondii*, with warmer winters and summers and earlier spring warming expected (reviewed in PRBO 2011). There is less certainty about future precipitation patterns, with estimates ranging from little change to roughly 30% decreases in rainfall (Snyder and Sloan 2005, PRBO 2011). Snowpack reductions of up to 90% are predicted in the southern part of the range (Snyder et al. 2004). Warmer and potentially drier conditions may affect availability of intermittent and ephemeral water bodies and therefore limit activity. In the more northern part of the range, the probability of large (>200 ha) fires is expected to increase (Westerling and Bryant 2008) and the area burned is expected to increase by up to 50% (Lenihan et al. 2008). Both increases and decreases in fire probability and extent have been predicted for southern California under different climate change scenarios. There is little consensus on future fire dynamics in this part of the range because of the difficulty in modeling Santa Ana weather events (Westerling et al. 2004, Westerling and Bryant 2008). How *T. hammondii* may respond to fire needs to be studied. Fire may have direct mortality effects and may alter aquatic and terrestrial habitat quality. Predicted vegetation shifts due to climate change include decreases in chaparral, shrubland, and woodland, and increases in grassland area (Lenihan et al. 2008, PRBO 2011). The impact of these shifts on *T. hammondii* populations will likely be negative.

Status Determination

Thamnophis hammondii has undergone declines and extirpations and occurs in an area of high human population density and development, resulting in a Priority 2 Species of Special Concern status.

Management Recommendations

Given this species' association with aquatic habitat and apparent willingness to use artificial habitats, restoration of aquatic habitat and supplementation with artificial wetlands should be explored as a management option in extirpated sites. Eradication efforts aimed at nonnative aquatic species should consider the potential effect on *T. hammondii* populations, particularly if native prey is not abundant (Mullin et al. 2004). Rathbun et al. (1993) documented the use of upland terrestrial habitat by *T. hammondii*, and the potential importance of rodent burrows for overwintering. In order to maintain access to these habitats, they suggested protecting terrestrial habitats within 500 m of aquatic habitats, although additional study across habitat types is needed.

Monitoring, Research, and Survey Needs

Surveys to determine the abundance and distribution of remaining populations are needed and they should be conducted by individuals that are well trained to distinguish among *Thamnophis* species. Additional data on movement ecology and habitat requirements are also necessary to facilitate the design of protected areas around known aquatic habitats and to inform possible restoration efforts. The degree to which *T. hammondii* is dependent upon introduced prey should be assessed, and the quality of those introduced prey compared to native prey should be evaluated. It may be necessary to manage for both *T. hammondii* and native prey populations simultaneously for effective recovery. Finally, landscape genetic data on the degree of differentiation at the regional and watershed levels would be valuable both for the identification of management units and for possible repatriation efforts.

COMMON GARTER SNAKE, SOUTHERN POPULATIONS

Thamnophis sirtalis (Linnaeus 1758)

Status Summary

Thamnophis sirtalis is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 72% (72/100). During the previous evaluation, garter snakes in this part of the range were also considered Species of Special Concern (Jennings and Hayes 1994a).

Identification

Southern coastal populations of *Thamnophis sirtalis* have not been formally described as a distinct taxon, so we limit our description here to *T. sirtalis* in general. *Thamnophis sirtalis* is a medium-sized species, with a head slightly wider than the neck and keeled dorsal scales (Stebbins 2003). *Thamnophis sirtalis* can reach up to 128 cm TL in California, with adult males from coastal California 46.3 cm SVL on average and females 58.0 cm SVL on average (J. Boundy, unpublished data). Color pattern varies widely in this species, but garter snakes typically have a dark dorsal background color with lighter dorsal and lateral stripes which can be faint or absent. California *T. sirtalis* tend to have red or orange coloration on the head and/or sides (Stebbins 2003). *Thamnophis sirtalis* in the southern part of its California range

Common Garter Snake, Southern Populations: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	25
iii. Population concentration/ migration (10)	Data deficient
iv. Endemism (10)	10
v. Ecological tolerance (10)	7
vi. Population trend (25)	10
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	7
Total Score	72
Total Possible	100
Total Score/Total Possible	0.72

PHOTO ON PREVIOUS PAGE: Common garter snake, Orange County, California. Courtesy of Jeff Lemm.

potentially co-occurs with the coast garter snake (*Thamnophis elegans terrestris*) and the two-striped garter snake (*T. hammondii*) (Stebbins 2003).

Taxonomic Relationships

Some sources consider coastal garter snake populations from southern California to be California red-sided garter snakes (*Thamnophis sirtalis infernalis*) (e.g., Barry and Jennings 1998, Stebbins 2003) and others refer to them as red-spotted garter snakes (*T. s. concinnus*) (e.g., Boundy and Rossman 1995, Janzen et al. 2002). Jennings and Hayes (1994a) based their evaluation of the putative taxon, the South Coast garter snake, on personal communications with J. Boundy and S. Sweet. Morphological and genetic studies that will help to clarify the status of this taxon are still pending. One study is a comparison of color and morphological variation among *T. sirtalis* from the south coast, central coast, and central valley of California (E. Ervin, pers. comm.; C. Mahrdt, pers. comm.). A phylogeographic study by Janzen et al. (2002) looked at *T. sirtalis* populations along the west coast of the United States but excluded populations from southern California. Another study sequencing two populations of southern *T. sirtalis* is underway for comparison with Janzen et al.'s (2002) study (R. Fisher, pers. comm.).

Life History

While the species *Thamnophis sirtalis* is one of the most well-studied North American snakes (Rossman et al. 1996), very little is known about populations from southern coastal California. Given that *T. sirtalis* is extremely widespread, occurring throughout much of Canada and in all but one state in the continental United States, life history variation among populations is pronounced (reviewed in Rossman et al. 1996, Ernst and Ernst 2003). We therefore limit our discussion to very general *T. sirtalis* biology and documented information from California where possible.

Like all members of the genus *Thamnophis*, young are live-born from midsummer to early fall. Cunningham (1959b) reported that an 89 cm SVL female *T. sirtalis* from Tapia Park, Los Angeles County, gave birth in late August to 20 young (18 live) that were about 25 cm in TL. Another 59 cm SVL female from the same area was carrying 12 embryos (Cunningham 1959b). A single female from farther north in San Benito County gave birth to six young, also in late August (Banta and Morafka 1968). Elsewhere, average litter sizes range from 7.6 in British Columbia (Gregory and Larsen 1993) to 32.5 in Maryland (McCauley 1945). Neonates range in size from 15 cm SVL (Manitoba; Gregory 1977, Gregory and Larsen 1993, Larsen et al. 1993) to 20 cm SVL (Lassen County, California; Jayne and Bennett 1990). Females mature at SVL of 43–57 cm, and males mature at 36–38 cm SVL, although this trait is highly variable across populations (Rossman et al. 1996). Time to maturity can take up to 4 years in some populations (e.g., Lassen County; Jayne and Bennett 1990). The proportion of females that are reproductively active in a given year ranges from 29% to 88% across populations (summarized in Rossman et al. 1996), suggesting that not every female breeds every year.

Thamnophis sirtalis can be active year-round in some southern localities (e.g., the Florida Everglades; Dalrymple et al. 1991). Rüthling (1915) anecdotally reported that *T. sirtalis* was rarely encountered around Los Angeles in the winter. Hansen and Tremper (unpublished data in Rossman et al. 1996) note that lowland California *T. sirtalis* are active from February to October, but there is a post-August drop in activity associated with a seasonal reduction in aquatic habitat. Most activity is diurnal, although crepuscular and nocturnal activity has been observed when anurans, a primary prey item, are breeding (Ernst and Ernst 2003). Nocturnal activity has also been observed in lowland California, with *T. sirtalis* active at night during warm rains (R. Hansen and R. Tremper, unpublished data in Rossman et al. 1996).

Thamnophis sirtalis are generalized predators (reviewed in Rossman et al. 1996).

However, diet data are not available from the southern range of *T. sirtalis* in California. Data from northern populations show that anurans are a large part of the diet. Anurans were the most common prey observed eaten by California *T. sirtalis*, comprising 58% of prey items (n = 48 snakes, localities include Siskiyou and Humboldt Counties; Fitch 1941). Also consumed were earthworms (24% of prey items), and rarer prey (5% or less of prey items) such as fish, leeches, and slugs (Fitch 1941). In northern California at Eagle Lake (Lassen County, 1555 m), regurgitation of 36 adults revealed that 33% of individuals contained anurans (mostly western toads, *Bufo boreas*), and 90% of prey items were anurans (Kephart and Arnold 1982). Fish (6% of animals, 2% of prey items) and leeches (11% of animals and 8% of prey items) were taken less frequently (Kephart and Arnold 1982). In the northern Sierra Nevada near Truckee, Nevada County, anurans comprised 56% of prey volume (mostly Pacific tree frogs, *Pseudacris regilla*), while 33% of prey volume was fish. Rarer prey items (5% or less of total prey volume) included mice and leeches (n = 88 snakes; White and Kolb 1974). Juvenile *Thamnophis sirtalis* in California have also been observed to consume newly metamorphosed newts (*Taricha torosa*) (S. Barry, unpublished data).

Habitat Requirements

Thamnophis sirtalis in southern California is thought to be restricted to marsh and upland habitats near permanent water and riparian vegetation (Grinnell and Grinnell 1907, Fitch 1941, Von Bloeker 1942; S. Sweet, pers. comm., in Jennings and Hayes 1994a). Data are scarce, but habitat preferences may be quite narrow. Some observational data suggest that this taxon may avoid restored marshlands, although the reasons for this are not clear (R. Fisher, pers. comm.).

Distribution (Past and Present)

Thamnophis sirtalis was historically known from scattered localities along the southern coastal plain from the Santa Clara River Valley in Ventura County to around San Pasqual in San Diego County (Klauber 1929, Jennings and Hayes 1994a; S. Sweet, pers. comm. in Jennings and Hayes 1994a; E. Ervin and C. Mahrdt, unpublished data). The historical elevation range is thought to be from near sea level at Ballona Creek and Playa del Ray Marsh in Los Angeles County to ~832 m at Lake Henshaw in San Diego County (Von Bloeker 1942; R. Fisher, pers. comm. in Jennings and Hayes 1994a). Jennings and Hayes (1994a) estimated that 75% (18/24) of historic localities no longer supported populations due to anthropogenic and natural habitat loss (e.g., urbanization, flooding). Of the six extant localities identified by Jennings and Hayes (1994a), it is now suspected that populations remain in only three localities, with possible extirpations including Camp Pendleton and San Luis Rey (R. Fisher, pers. comm.).

Trends in Abundance

Historical accounts suggest that *Thamnophis sirtalis* was once quite common (Grinnell and Grinnell 1907, Bogert 1930, Von Bloeker 1942). Current populations are thought to be abundant at Lake Henshaw in San Diego County, rare along the Santa Clara River, and virtually extirpated elsewhere (S. Barry, pers. comm., R. Fisher, pers. comm.).

Nature and Degree of Threat

Extirpations and population declines in this taxon have been attributed to habitat loss and fragmentation due to urbanization, agriculture, and flood control projects, as well as natural events such as floods and droughts (De Lisle et al. 1986, Jennings and Hayes 1994a). At remaining sites, urbanization in Riverside County continues to impact the Santa Margarita River wetlands at Camp Pendleton, and increased dam height in the Prado Basin may have a negative flooding impact (R. Fisher, pers. comm.). Introduced aquatic predators and water snakes (genus *Nerodia*) may also negatively impact *Thamnophis sirtalis* (Jennings and Hayes 1994a; R. Fisher, pers. comm.).

Under climate change, mean annual temperatures are projected to increase throughout the southern California range of *T. sirtalis*, with warmer winters and summers and earlier spring warming expected (reviewed in PRBO 2011). There is less certainty about future precipitation patterns, with estimates ranging from little change to roughly 30% decreases in rainfall (Snyder and Sloan 2005, PRBO 2011). Snowpack reductions of up to 90% are predicted in southern California (Snyder et al. 2004). Warmer and potentially drier conditions may affect availability of intermittent and ephemeral water bodies and therefore limit activity. Increases and decreases in fire probability and extent have been predicted for southern California. There is little consensus on future fire dynamics because of the difficulty in modeling Santa Ana weather events (Westerling et al. 2004, Westerling and Bryant 2008). How *T. sirtalis* responds to fire is unknown. Fire may have direct mortality effects, and may alter aquatic and terrestrial habitat quality. Predicted vegetation shifts due to climate change include decreases in chaparral and shrubland and increases in grassland area (Lenihan et al. 2008, PRBO 2011). The potential impact of such vegetation shifts on *T. sirtalis* populations is unknown.

Status Determination

Thamnophis sirtalis in southern California has a very small range in a heavily human-impacted part of the state. In addition, these populations have been extirpated from most of their historical range, which justifies a Priority 1 Species of Special Concern designation.

Management Recommendations

Given the paucity of ecological information on southern populations, it is difficult to make management recommendations beyond the protection of existing habitat at this time. Future management strategies may include removal of invasive animals and plants, restoration of flow regimes, and repatriation of extirpated sites. The research needs outlined below will help to inform the eventual development of a management strategy for this taxon.

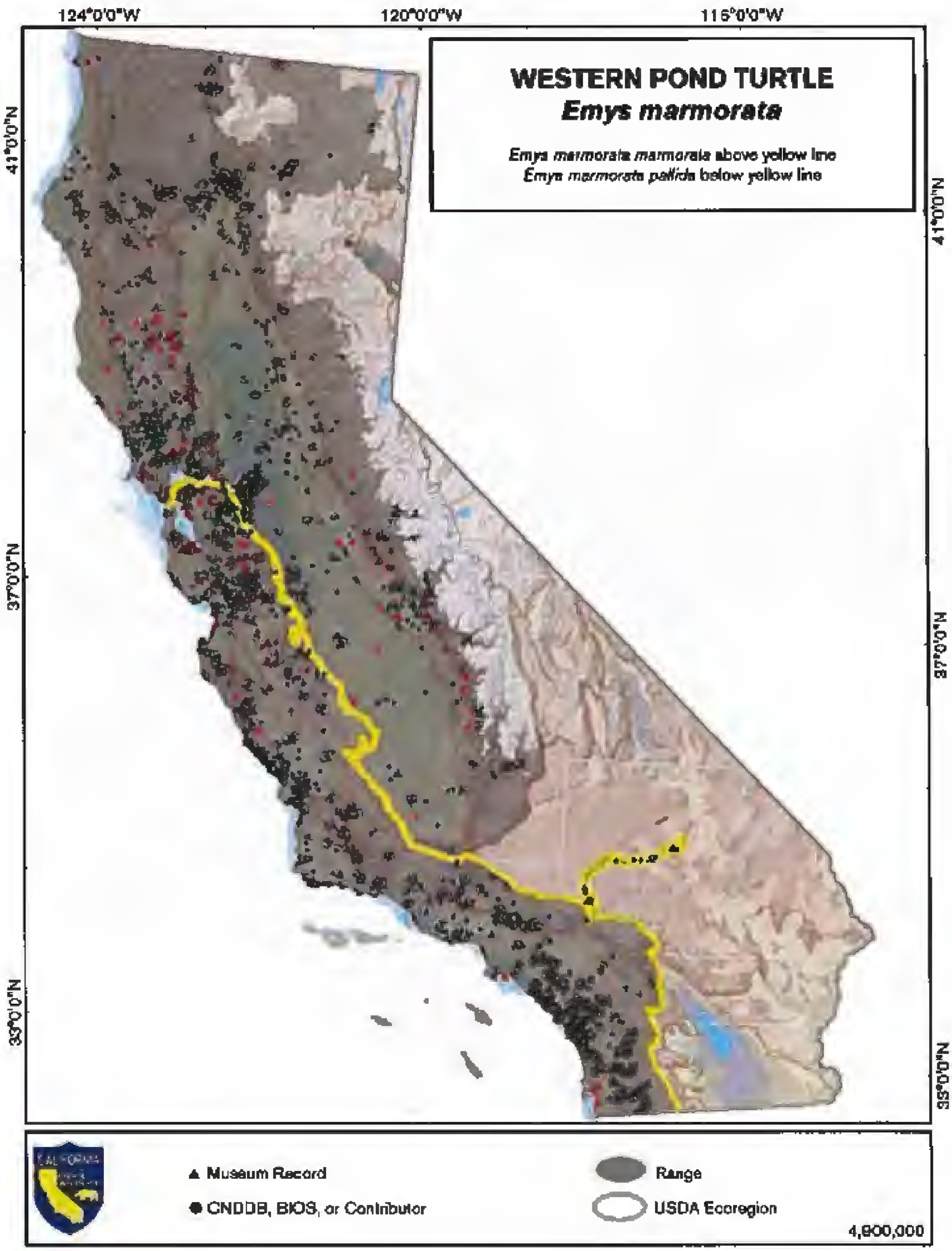
Monitoring, Research, and Survey Needs

Almost no ecological or life history information is available for this taxon, and this data gap needs to be addressed at the few remaining sites in southern California where *Thamnophis sirtalis* persists. Monitoring to determine population abundance and to verify extirpation is needed across sites. As remaining habitat is identified and extant populations are found and stabilized, human-mediated repatriation, perhaps in combination with captive breeding, may be the most effective strategy to repopulate extirpated sites. Studies on movement and dispersal are needed to determine connectivity among remaining populations, and genetic studies on both the differentiation of this taxon from other *T. sirtalis* populations and the level of among-population variability are needed. Finally, the importance and impacts of nonnative species as predators and prey should be investigated further.

WESTERN POND TURTLE

Emys [=Actinemys] marmorata Baird and Girard 1852

PHOTOS: (top) Western pond turtle, Solano County, California. Courtesy of Adam Clause. (bottom) Western pond turtle, Santa Barbara County, California. Courtesy of Robert Hansen.


124°0'0"W
120°0'0"W
116°0'0"W
41°0'0"N
37°0'0"N
33°0'0"N
WESTERN POND TURTLE
Emys marmorata
Emys marmorata marmorata above yellow line
Emys marmorata pallida below yellow line
Museum Record
CNDDB, BIOS, or Contributor
Range
USDA Ecoregion
4,900,000

Status Summary

Emys marmorata is a Priority 1 Species of Special Concern in the southern part of the range (roughly corresponding to the range of the southwestern pond turtle, *E. m. pallida*) and a Priority 3 Species of Special Concern elsewhere (roughly corresponding to the range of the northwestern pond turtle, *E. m. marmorata*; see below for additional detail). These two populations received a Total Score/Total Possible of 81% (89/110) and 65% (71/110), respectively. During the previous evaluation, both populations were considered Species of Special Concern, also with different overall levels of threat (Jennings and Hayes 1994a).

Identification

Emys marmorata is a small to medium-sized (generally 17–18 cm, rarely to 24 cm, straight carapace length) brown, tan, or olive turtle (Stebbins 2003). The carapace is low, keelless, and often marked with a pattern of dark lines and/or dots, sometimes forming a pattern that radiates from the centers of each scute. The posterior edge of the carapace forms a smooth, non-serrated rim. In some individuals, the carapace has no patterning. The plastron is lighter tan or beige, hingeless, and often marked with dark blotches (Stebbins 2003). The shell shape varies among habitat types, with turtles from foothill streams being flatter and narrower than individuals occurring at lower elevations in canals and sloughs (Lubcke and Wilson 2007).

This species is unlikely to be confused with other turtles within its range with the possible exception of melanistic individuals of the nonnative red-eared slider (*Trachemys scripta elegans*). This latter species has a much shorter tail, attains larger overall body sizes, and has a serrated rim around the posterior edge of the carapace. Most individuals of this species also have prominent yellow stripes on the neck and shell and a broad red stripe over the temporal region of the head, although older individuals often develop an overall dark melanistic coloration.

Taxonomic Relationships

Emys marmorata is a member of the family Emydidae, which encompasses the majority of North American turtle species. The relationships within this group have undergone extensive revision in recent years, leading to many taxonomic changes and some instability. Formerly, this species was included in the genus

Northern Western Pond Turtle: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	15
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	7
v. Ecological tolerance (10)	0
vi. Population trend (25)	20
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	71
Total Possible	110
Total Score/Total Possible	0.65

Southern Western Pond Turtle: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	5
ii. Distribution trend (25)	25
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	7
v. Ecological tolerance (10)	0
vi. Population trend (25)	25
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	89
Total Possible	110
Total Score/Total Possible	0.81

Clemmys along with the bog turtle (now *Glyptemys muhlenbergii*), the wood turtle (now *G. insculpta*), and the spotted turtle (now *C. guttata*). Recent molecular analyses have suggested a close relationship between *E. marmorata*, Blanding's turtle (*Emys* [=*Emydoidea*] *blandingii*), and the European pond turtles (*E. orbicularis* and *E. trinacris*) (Bickham et al. 1996, Burke et al. 1996, Feldman and Parham 2002, Spinks and Shaffer 2009, Spinks et al. 2009). This species is now generally placed in either the monotypic genus *Actinemys* (Holman and Fritz 2001) or the genus *Emys* (the arrangement that we follow here).

Intraspecific variation within *E. marmorata* is also undergoing intensive study. Two subspecies have traditionally been recognized, *E. m. marmorata* (Baird and Girard 1852) and *E. m. pallida* (Seeliger 1945). These subspecies were initially distinguished by the presence or absence of inguinal scutes in the shell and coloration of the throat and neck. Subsequent studies also detected substantial morphological variation present across the range (Holland 1992a). Genetic analyses of intraspecific variation suggest that substantial variation is present, which is generally, but not precisely, concordant with the traditionally defined subspecies (Spinks and Shaffer 2005, Spinks et al. 2010). Spinks et al. (2014) analyzed a large panel of SNPs and concluded that *E. m. sensu lato* should be divided into two species. Because this arrangement is very recent, here we follow the earlier arrangement (of a single species) but consider threats separately for southern and northern populations as was done by Jennings and Hayes (1994a).

Life History

Emys marmorata is a highly aquatic species and basks frequently. In the northern part of the range (particularly at higher elevations), this species enters a period of dormancy throughout much of the winter. It is one of relatively few emydid turtles that regularly overwinter on land (Ultsch 2006), perhaps as a mechanism to avoid mortality from increased winter water flows in the Mediterranean climate. Where it overwinters terrestrially, the species uses a variety of habitat types but chooses sites above the normal high water mark and burrows into loose soils and leaf litter (Reese 1996). In aquatic habitats that experience little change in water level (lakes, ponds, and reservoirs), pond turtles are known to overwinter in the water and will choose undercut banks, bottom mud, "snags" of downed wood, or rocks (Nussbaum et al. 1983, Ernst and Lovich 2009). Movement to overwintering sites occurs at the end of summer, most often in September, although the timing varies with the particular habitat and area (Reese 1996, Reese and Welsh 1997). In warmer areas, particularly in the southern part of the range, this species may remain active year-round.

Western pond turtles are known to mate throughout the spring, summer, and fall. Nesting usually occurs in the spring or early summer, although double clutching has been reported from several parts of the range (Goodman 1997, Germano and Bury 2001, Germano and Rathbun 2008, Scott et al. 2008). Females usually select nest sites within 100 m of a water body, although nests as far away as 500 m have occasionally been reported (Storer 1930, Holland 1994, Reese 1996, Holte 1998, Lovich and Meyer 2002). Clutch sizes vary from 1 to 13 eggs and vary depending on local conditions (Holland 1994, Lovich and Meyer 2002, Germano and Rathbun 2008). The eggs hatch in the fall and, at least in the northern part of the range, hatchlings often remain in the nest through the first winter, emerging the following spring (Holland 1994).

The diet is generalized and consists of a variety of small aquatic invertebrates (including insects, crustaceans, and mollusks) and a wide variety of algae and other plant material (Bury 1986). Carrion and small vertebrates are also occasionally consumed (Bury 1986). Growth rates vary widely depending on local conditions but appear to be highest in hatchlings and then gradually slow in adults. Reproductive maturity is widely variable and appears to be linked to size. Females generally mature at slightly over 13 cm SCL as young as 4–5 years of age, while males mature at about 12.5 cm SCL at 6–8 years of age (Holland 1994, Reese 1996, Germano and Bury 2001, Germano and Rathbun 2008, Germano and Bury 2009; T. Engstrom, pers. comm.), although maturation can happen more quickly depending on local conditions in some areas (e.g., Germano 2010).

Habitat Requirements

Emys marmorata is generalized in its habitat requirements, occurring in a broad range of aquatic water bodies including flowing rivers and streams, permanent lakes, ponds, reservoirs, settling ponds, marshes, and other wetlands. This species will also temporarily use semipermanent or ephemeral water bodies, including stock ponds, vernal pools, aud seasonal wetlands (Stebbins 2003, Bury and Germano 2008). This species will also at least occasionally enter sea water (Stebbins 1954, Holland 1989). Pond turtles require upland habitat that is suitable for nesting and overwintering use. Localized soil conditions, as well as the frequency and degree of disturbance in the upland habitat, probably limit their distribution. Soils need to be loose enough to allow nest excavation, while disturbance needs to be infrequent enough or of sufficiently low intensity that nests are not disturbed (Ernst and Lovich 2009).

This species is most frequently found in quiet reaches that experience little human impact and have abundant basking substrate in the form of downed wood and large rocks (Bury and Germano 2008, Thomson et al. 2010). The species can persist, at least over moderate periods of time, in highly modified habitats with high human traffic and/or little basking substrate (Spinks et al. 2003, Germano 2010).

Distribution (Past and Present)

Emys marmorata ranges widely along the Pacific coast from western Washington to the northern part of the Baja California Peninsula in Mexico. Within California, the species ranges from the Pacific coast inland to the Sierra Nevada foothills up to elevations of 2048 m (Ernst and Lovich 2009). Further south, it ranges from the coast inland to the peninsular ranges. Scattered populations exist in the Mojave River (e.g., Victorville, Camp Cady, and Afton Canyon, San Bernardino County, California) and in some Great Basin drainages including the Susan River (Lassen County, California), and the Truckee and Carson Rivers (Nevada, possibly extending into Nevada County, California, although this has not been documented) (Holland 1992b, Lovich and Meyer 2002). Additional scattered populations are known from the Klamath Basin (R. Bury, pers. comm.). Some or all of these populations could represent introductions. One hundred and eighty individuals of this species were introduced in the state of Nevada in 1887, and these may be the source of the population in the Truckee and/or Carson Rivers (Cary 1889).

Within *E. marmorata*, the southern subspecies (*E. m. pallida*) extends from the southern range edge in Baja California, Mexico, northward in the Coast Range to San Francisco Bay, while the northern subspecies (*E. m. marmorata*) extends from San Francisco Bay north through the Sacramento Valley and Coast Range to the northern range limit in Washington. A large intergrade zone between the two subspecies has been hypothesized to exist in the San Joaquin Valley (Seeliger 1945), although recent work has shown that this area is genetically a member of the northern subspecies (Spinks et al. 2014). The populations that we recognize correspond to these subspecies distributions.

In the north, large and relatively intact populations still exist through large areas of the Coast Range and Sierra foothills, although agriculture and habitat modification have destroyed large areas of riparian and wetland habitat in the Sacramento Valley that almost certainly supported large populations of this species in the past. Scattered populations remain throughout the Sacramento Valley, but the extensive marsh habitat that dominated much of the valley floor has been largely drained and converted to agriculture. Kelly et al. (2005) estimated that the extent of wetland habitat in the Central Valley has declined by ~80% since the 1860s when large-scale land conversion began, and this undoubtedly eliminated many *E. marmorata* populations. Holland (1992b) argued that the San Joaquin River drainage formerly represented the stronghold of this species, supporting vast numbers of individuals, and that the species has been lost from >99% of its range in the region. Overall, the number of viable populations in this area has clearly decreased, but some do remain (Holland 1992a, Jennings and Hayes 1994a, Germano 2010, Bury et al. 2012).

In the south, extensive urbanization and land conversion have caused precipitous population declines. A large fraction of remaining habitat in southern California exists as patches surrounded by large tracts of unsuitable habitat that have little suitable upland nesting habitat. Dispersal corridors between adjacent habitats have also been mostly severed by intervening urban development and heavily used roadways, resulting in heavy mortality on females searching out nest sites (R. Fisher, pers. comm.).

Trends in Abundance

Emys marmorata was formerly abundant throughout much of California. Bogert (1930) reported that *E. marmorata* was "common in larger streams along the coast and in many of the marshes adjacent to the coast," and many of these habitats still support relatively large populations (Jennings and Hayes 1994a, Germano and Rathbun 2008, Thomson et al. 2010). Elsewhere declines have occurred, particularly in southern California. Van Denburgh (1922) reported that the species was "abundant on the west fork of the San Gabriel River," but recent reports suggest that the species has declined precipitously in this area and in the Los Angeles Basin in general (Brattstrom 1988, Jennings and Hayes 1994a). Large, relatively intact populations remain through much of the northern Coast Ranges, although areas in the Central Valley and southern California that still support the species have severely declined (Holland 1992b, Jennings and Hayes 1994a). Populations that remain in the Central Valley are undoubtedly smaller and more fragmented than they once were due to the large-scale land conversion that occurred in this area beginning in the 1860s. Further, *E. marmorata* were harvested commercially for many years, selling for 3–6 dollars per dozen in San Francisco markets during the 1920s and 1930s (Pope 1939, Nussbaum et al. 1983). The overall extent of declines in abundance caused by market collection is poorly understood. However, localized declines due to market collection were noted as early as 1879 in Sacramento (Lockington 1879), and the species' life history would make it particularly susceptible to declines from intense adult mortality.

Some published and ongoing surveys suggest that population sizes are stable in several remaining populations in the southern part of the range. In particular, southern populations near Gorman, Fresno, and along the central coast of California appear to be stable in abundance with a population structure that indicates continued breeding (Germano 2010; D. Germano, pers. comm.). Unpublished field data also indicate that the species persists in some numbers throughout Merced (particularly east of Gustine) and Fresno Counties, as well as some areas of Kern County (S. Barry, pers. comm.). At least in some areas, ongoing declines in abundance may have slowed or stopped. If additional data corroborate these observations, a decrease in the population trend scores may be warranted during the next Species of Special Concern evaluation.

Nature and Degree of Threat

The largest threats currently facing *Emys marmorata* are land use changes and fragmentation of existing habitat, as well as possible impacts via competition and predation by introduced species.

Throughout the range of *E. marmorata*, extensive wetland habitats that once supported large numbers of this species have declined in extent and quality. Ongoing land use conversion to agriculture as well as urban development have degraded and fragmented habitat throughout virtually all of this taxon's range. These effects are most pronounced in southern California, where relatively few viable populations of this species now remain. Even in northern California, land use changes are having impacts. Reese and Welsh (1998) documented changes in the age structure of *E. marmorata* populations as a result of damming in the Trinity River drainage, suggesting negative impacts on juvenile turtles and therefore recruitment in populations affected by dams.

The impact of introduced species is largely unknown but could potentially be detrimental in several ways. The red-eared slider is widely established throughout the range of *E. marmorata* and may serve as a disease vector and competitor (Bury 2008a). The spiny softshell turtle (*Apalone spinifera*) is a more recent introduction to the Central Valley of California and is now breeding in at least one site in the Sacramento Valley (L. Patterson, pers. comm.). If this species becomes invasive on a larger scale, it is also likely to compete with and possibly prey on small *E. marmorata*. In Southern California, the range of these two species appears not to overlap, suggesting that softshells may have strong impacts on pond turtles (R. Fisher, pers. comm.). Additional introduced species that may affect *E. marmorata* are bullfrogs, crayfish, and introduced centrarchids. In the Salinas River, *E. marmorata* declined following the invasion of bullfrogs in the 1970s (B. Hubbs, pers. comm.). The strength and mechanism (predation or competition) of their impact is not currently clear, and further studies are needed. Ravens, crows, raccoons, and opossums are all known predators of *E. marmorata* adults and nests. The population sizes of these human commensal species have increased through time and may also be having impacts on *E. marmorata* populations via increased predation pressure. A very important source of this decline may operate through nest predation that leads to reduced or failed recruitment year to year (S. Sweet, pers. comm.).

The impacts of climate change on *E. marmorata* are still poorly understood but are likely to be significant. Climate simulation models project strong changes to river hydrology in California. In particular, decreasing snowpacks and a shift to earlier and stronger river flows (and increased frequency and strength of scouring floods) are likely to negatively affect habitat and could cause local extirpations (Cayan et al. 2008b). Because the habitat is now fragmented, recolonization of these areas following localized extirpations is unlikely, particularly in southern California where the habitat is the most fragmented. Importantly, the genetic data indicate that most of the genetic diversity within this species resides in southern California. Because of this, declines in this area could result in the extirpation of much of the genetic diversity that is currently present (Spinks et al. 2010, Spinks et al. 2014; R. Fisher, unpublished data).

Status Determination

Priority 1 Species of Special Concern status is justified for *Emys marmorata* in the southern portion of the range because these populations are experiencing ongoing and strong declines in distribution and abundance (although, as noted above, some evidence indicates these declines may be slowing in some areas). Further, this area contains most of the genetic diversity that has been identified within this taxon, so entire genetic lineages are at risk. In the north, populations are experiencing declines, although to date they are less severe than in the southern portion of the range. Many of the remaining populations in the north occur in habitats that are unlikely to experience land use changes on a scale that will threaten long-term survival, so we consider this segment of the range a Priority 3 Species of Special Concern.

Management Recommendations

Our recommendations follow those of Bury et al. (2012). We outline these recommendations below and refer readers to that document for additional discussion. Protecting habitat from further degradation and fragmentation is the highest priority for this species. Following this, habitat restoration, particularly that which increases connectivity between currently isolated habitats and increases the extent of setback or buffer habitat around wetlands that is suitable for nesting, is an important management priority. Efforts to reduce or control the impact of predators (especially on nests) are also an important way to maintain current populations and increase recruitment of juveniles. Formal headstarting programs may be a useful tool for repopulating areas where local extirpations have occurred but only as a last resort and if the habitat can be restored to an extent that a population can survive with little intervention. One encouraging observation is that *Emys marmorata* can live in close proximity to human disturbance, provided that they have adequate suitable basking and nesting sites.

Monitoring, Research, and Survey Needs

Further research on the impact of invasive species is needed. In particular, the impact of red-eared sliders, bullfrogs, and centrarchids needs to be further characterized, to understand both to what extent these species can coexist and the effects these species have on the native populations. Both nest and hatchling habitat requirements are relatively poorly characterized, and need to be clarified if the species is to persist and thrive in human-modified habitats. The effectiveness of headstarting efforts needs to be evaluated in various habitats and predation situations. Because a large amount of life history variation is present in this taxon (particularly relating to time to maturity, body size, and clutch size; e.g., Germano 2010), researchers and managers should be cautious when applying life history data collected in one population to a different population, particularly those occurring at widely different elevations, water temperatures, or habitat types.

SONORA MUD TURTLE

Kinosternon sonoriense Le Conte 1854

Status Summary

Kinosternon sonoriense is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 66% (56/85). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a). It has not been recorded from its historic range along the California–Arizona border since 1962.

Identification

Kinosternon sonoriense is a small (maximum size ~17 cm SCL) black or brown turtle, with prominent barbels on the chin and neck and usually with yellow or cream mottling on the sides of the head that form broken stripes (Ernst and Lovich 2009). The plastron is lighter than the carapace, usually pale brown to yellow, with dark pigmentation along the scute seams and well-developed anterior and posterior hinges (Ernst and Lovich 2009). The overall carapace shape is oval and moderately domed. This species is unlikely to be confused with other native California turtles, since it is the only California native that possesses barbels and the only native aquatic turtle that occurs within its range. However, many kinosternid turtle species are

Sonora Mud Turtle: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	20
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	0
v. Ecological tolerance (10)	3
vi. Population trend (25)	Data deficient
vii. Vulnerability to climate change (10)	3
viii. Projected impacts (10)	10
Total Score	56
Total Possible	85
Total Score/Total Possible	0.66

PHOTO ON PREVIOUS PAGE: Sonora mud turtle, Santa Cruz County, Arizona. Courtesy of Jeff Lemm.

difficult to distinguish, and some of these have been sporadically introduced around the state. The most common introduced kinosternid is likely the common musk (or stinkpot) turtle (*Sternotherus odoratus*). This species has two broken light stripes on each side of the head and has only a single, anterior hinge on the plastron. Other species in the genus *Kinosternon* have also been introduced (*K. flavescens* in particular; S. Sweet, pers. comm.) but are not common and will often require expert identification (Spinks et al. 2003, Spinks et al., pers. obs.). See Stebbins (2003) for additional details.

Taxonomic Relationships

Two subspecies have been described, one of which historically occurred in California. The Sonora mud turtle (*Kinosternon sonoriense sonoriense*) includes California as well as the majority of the species' range in the southwestern United States and northern Mexico. The Sonoyta mud turtle (*K. s. longifemorales*) is restricted to the Rio Sonoyta drainage in Mexico and southern Arizona (Iverson 1976). Intraspecific, including subspecific, variation has not yet been investigated genetically.

Life History

The life history of this species has not been studied in California. Life history studies in Arizona and New Mexico suggest that there is some interpopulation variation in basic life history parameters of this species. We base our life history description on work conducted primarily in Arizona and New Mexico but recognize that these data should be regarded as tentative for California populations.

Kinosternon sonoriense is active throughout the year as long as water is present, though in warmer months it may become active primarily at night (Hulse 1974, Hulse 1982). Hibernation is known to occur in high-elevation populations in New Mexico (Degenhardt et al. 1996), although it is unlikely that this occurs in California populations, which were exclusively low elevation. *Kinosternon sonoriense* aestivates terrestrially in response to seasonal drying in several populations (Ligon and Stone 2003, Hall and Steidl 2007, Hensley et al. 2010) but elsewhere may be more closely tied to permanent water (Ligon and Peterson 2002). In Arizona, females come into reproductive condition after a minimum of 5 years or with a carapace length between 115 and 125 mm, after which they produce one to four clutches per year although this varies depending on location (Van Loben Sels et al. 1997, Ernst and Lovich 2009, Lovich et al. 2012). Females become gravid between April and September, although most frequently in June and July (Lovich et al. 2012). The developing embryos apparently require a period of cooling before development restarts in the spring (Hulse 1982, Ewert 1991, Ernst and Lovich 2009). In Arizona, hatching may be associated with the summer monsoon in late summer (van Loben Sels et al. 1997).

Kinosternon sonoriense can attain high local population densities. One population in Hidalgo County, New Mexico, contained 212 turtles (Stone 2001). Another population in Yavapai County, Arizona, reached 750 individuals/ha of aquatic habitat (Hulse 1982). Individuals are known to undertake long terrestrial movements (>1 km) when water becomes limiting (Stone 2001, Hall and Steidl 2007), and Stone (2001) found that 26% of recaptured individuals had moved overland between aquatic capture sites. In the Santa Catalina Mountains (Pima County, Arizona), where the aquatic habitat consists of small and discrete pools, the presence of two or more adult turtles of the same sex within single pools was rare, suggesting that the species may be territorial where resources are limiting (Hall and Steidl 2007).

Kinosternon sonoriense is primarily carnivorous, feeding on a variety of invertebrates. It is known to shift to omnivory in suboptimal habitat (Hulse 1974) and to feed on or scavenge small vertebrates (Stone et al. 2005, Lovich et al. 2010).

Habitat Requirements

Habitat requirements for *Kinosternon sonoriense* in California are unknown but are likely tied to the presence of a reliable water source and a suitable prey base. Elsewhere in its range, it inhabits a wide variety of both permanent and temporary aquatic habitats including streams, creeks, stock ponds, and natural ponds (van Loben Sels et al. 1997, Ernst and Lovich 2009, Stanila 2009, Hensley et al. 2010, Stone et al. 2011). In California, it was known to enter artificial water bodies, although the long-term suitability of this habitat is unknown. Optimal habitat appears to be slow-moving, permanent water with a high density of aquatic invertebrates and a muddy bottom (Jennings and Hayes 1994a).

Distribution (Past and Present)

Historically, this species occurred in California along the Lower Colorado River drainage (Van Denburgh and Slevin 1913, Grinnell and Camp 1917, Dill 1944). La Rivers (1942) reported the northernmost record for the species in the Colorado River drainage from Clark County, Nevada. Cooper (1870) mentioned a specimen from an unspecified locality in the Colorado River Valley, collected while he was stationed at Fort Mohave, Arizona. Several more individuals were collected from the vicinity of Yuma, Arizona, and Palo Verde, California, in the early 1900s (Van Denburgh and Slevin 1913, Van Denburgh 1922). A 1942 record (SDNHM 17897) extended the western range in California to within ~20 km of Calexico, suggesting that this taxon was present in ditches and canals in the Imperial Valley for at least some period of time. Klauber (1934) indicates that it was not "yet" present in the Imperial Valley, though by 1942 it clearly was. The overall extent and timing of its expansion into the Imperial Valley is essentially unknown. In the Lower Colorado River Valley, the species was present at least until 1941 near Bard, Imperial County (SDNHM 33866).

The last published record of *Kinosternon sonoriense* in the Lower Colorado River drainage occurred on the Arizona side of the river ~1.6 km southwest of Laguna Dam on 31 March 1962 (Funk 1974, Lovich and Beaman 2008). Turtle trapping surveys were conducted in April of 1991 throughout much of the historic California range and failed to detect the species (King and Robbins 1991). The presence of "small black turtles along the Coachella Canal" was rumored in the 1990s, but these reports were never verified and could have been misidentified *Trachemys scripta* or *Apalone spinifera* (J. Lovich, pers. comm.).

Outside of California, *K. sonoriense* ranges through much of southern Arizona, into the southwestern corner of New Mexico and south into northern Sonora and Chihuahua, Mexico, from sea level to 2040 m (Stebbins 2003, Lovich and Beaman 2008, Ernst and Lovich 2009).

Trends in Abundance

There is no information concerning historical abundance of this species in California. Only five reliable localities have been recorded in California, and historical accounts from the early twentieth century contain few data on abundance. Van Denburgh and Slevin (1913) reported that "six or eight" specimens were collected near Yuma before 1906, and stated that "whether it ascends the Colorado River above the Gila is not known." Van Denburgh (1922) stated that the species occurred in the Lower Colorado River drainage but was aware of records only near Yuma and at Palo Verde in Imperial County. The Clark County Nevada record had not yet been reported at this time (La Rivers 1942). Dill (1944) mentioned only that this taxon occasionally stole bait from fishermen (presumably implying that it was fairly well known to fishermen). The paucity of records from California suggests that populations here may not have occurred in the high densities documented elsewhere, although this species is difficult to detect without specific trapping efforts, and it is not clear that these efforts were ever made while the species was known to be present. Thus, the historical data on abundance are inconclusive. *Kinosternon sonoriense* has not been collected in or near California in nearly 50 years, despite extensive surveys (King and Robbins 1991). It is clear that declines, and possibly extirpation, have occurred during the last century.

Nature and Degree of Threat

The causes of decline of *Kinosternon sonoriense* in California are poorly understood, but may be associated with habitat modification and water diversion along the Colorado River and the Imperial Valley (Ohmart et al. 1988). Increased use of pesticides may have modified the available prey base, forcing the species to shift to a suboptimal herbivorous diet, which has been suggested as a factor in other *K. sonoriense* declines (King et al. 1996). The impact of introduced exotic crayfish, bullfrogs, warm water fishes, and softshelled turtles, all of which were well established around the time of *K. sonoriense* declines (Dill 1944, Lovich and Beaman 2008), is unknown, but they could plausibly have had a negative impact on *K. sonoriense*. At one site in Arizona, reduced *K. sonoriense* densities appear to be associated with the presence of introduced crayfish (Lazaroff et al. 2006).

Between 1941 and 1943, the Imperial Irrigation District burned and sprayed oil on 13,000 km of ditches and canals in the Imperial Valley in an effort to control the damage being done by spreading muskrat populations (Twining and Hensley 1943). These efforts certainly destroyed a great deal of aquatic habitat in the region, and the effect of the oil residues may have also had strong impacts on *K. sonoriense* and other taxa that disappeared from this area during the same time period (e.g., *Rana yavapaiensis*, *Bufo alvarius*).

Status Determination

A Priority 1 Species of Special Concern designation is justified by the complete absence of records for this species since the 1960s. This is the primary cause for concern. Little understanding of *Kinosternon sonoriense*'s habitat requirements or factors leading to decline in California currently exists. However, given the survey efforts that have been conducted to date, we assume that any remaining California populations are small, fragmentary, and vulnerable to extirpation. The species may also be vulnerable to increasing temperatures and changing hydrology due to climate change.

Management Recommendations

If future surveys detect any remaining populations, initial management efforts should focus on protecting those populations while research is performed that focuses on expanding suitable habitat and rebuilding local populations. If initial estimates of population structure indicate that reproduction and/or recruitment is not occurring, a headstarting program could be effective as a stopgap measure to prevent local extirpation. Many aquatic turtles have very different habitat requirements for hatchlings and adults, and ecological studies of both age classes will almost certainly be necessary to ensure the survival of remnant native populations.

Monitoring, Research, and Survey Needs

Although surveys have been performed for *Kinosternon sonoriense* in California, these efforts are not yet comprehensive. As this species is generally easily captured using submersible turtle traps, more complete survey efforts will help to clarify the species' status in California. Areas that have not yet been systematically surveyed include the backwaters of the Colorado River below Needles and along Lake Havasu (R. Fisher, pers. comm.); Haughtelin, Ferguson, Taylor, Draper, and Walker Lakes (King and Robbins 1991); the Coachella Canal; and any riparian habitat remaining in the area of Laguna Dam, as well as at Topock Marsh in the Havasu National Wildlife Refuge. Because the Lower Colorado River segment of the species' range spanned both California and Arizona, additional surveys should be coordinated with wildlife managers in Arizona to search potential habitat on the eastern side of the Colorado River.

If surveys do detect any individuals, managers should immediately initiate a monitoring program to determine the size and stability of the population, as well as an ecological study of population structure and life history. This will almost certainly involve individually marking turtles with shell notches and/or PIT tags and performing mark–recapture surveys to estimate population size and individual growth rate. In particular, whether, and how much, reproduction is taking place in existing populations will be critical to determine. Juvenile turtles rarely enter submersible traps; thus, alternative methods should be employed to search for them (such as seining or snorkeling). Female turtles should also be checked for eggs using either palpation or radiographs, preferably with portable field-capable digital X-ray units.

Genetic samples from the Lower Colorado River do not exist and should be collected, should remaining populations be found. These samples will be valuable to researchers working on *Kinosternon* phylogenetics and phylogeography and will also be critical in assessing the existing diversity within remaining populations and the divergence between these and more abundant populations to the east in Arizona.

Finally, researchers should attempt to characterize differences between habitat that supports this species and nearby habitats that do not. The causes of decline are still poorly understood, so management efforts that focus on rebuilding populations must be informed with strong data on the impact of introduced predators, pesticide, and herbicide drift, introduced aquatic plants, and habitat modification on *K. sonoriense* population persistence.

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APPENDIX 1

List of Native Amphibian and Reptile Taxa Occurring in California

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
		Anura					
Ascaphidae							
Ascaphus truei	Coastal tailed frog	X		SSC	LC		
Bufonidae[6]							
Bufo alvarius	Sonoran Desert toad	X		SSC	LC		
Bufo boreas boreas	Western toad				NT		
Bufo boreas halophilus	California western toad				NT		
Bufo californicus	Arroyo toad	X	E	SSC	E		
Bufo canorus	Yosemite toad	X	T	SSC	E	S	
Bufo cognatus	Great Plains toad				LC		
Bufo exsul	Black toad	X		T, FP	V	S	S
Bufo punctatus	Red-spotted toad				LC		
Bufo woodhousii	Woodhouse's toad				LC		
Hylidae							
Pseudacris cadaverina	California treefrog				LC		
Pseudacris regilla[7]	Pacific treefrog				LC		
Ranidae							
Rana aurora	Northern red-legged frog	X		SSC	LC	S	
Rana boylii	Foothill yellow-legged frog	X		SSC	NT	S	S

(continued)

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
		Anura					
Rana cascadae	Cascades frog	X		SSC	NT	S	
Rana draytonii	California red-legged frog	X	T	SSC	V		
Rana muscosa	Southern Mountain yellow legged frog	X	E	E	E	S	
Rana pipiens[8,9]	Northern leopard frog	X		SSC	LC		
Rana pretiosa[10]	Oregon spotted frog	X	T	SSC	V		S
Rana sierrae	Sierra Nevada yellow legged frog	X	E	E	E	S	
Rana yavapaiensis	Lowland leopard frog	X		SSC	LC		S
Scaphiopodidae							
Scaphiopus couchii	Couch's spadefoot	X		SSC	LC		S
Spea hammondii	Western spadefoot	X		SSC	NT		S
Spea intermontana	Great basin spadefoot				LC		
		Caudata					
Ambystomatidae							
Ambystoma californiense	California tiger salamander	X	T	T	V		
Ambystoma californiense "Santa Barbara"	Santa Barbara tiger salamander	X	E	T	V		
Ambystoma californiense "Sonoma"	Sonoma tiger salamander	X	E	T	V		
Ambystoma gracile	Northwestern salamander				LC		
Ambystoma macrodactylum croceum	Santa Cruz long-toed salamander	X	E	E, FP	LC		
Ambystoma macrodactylum sigillatum	Southern long-toed salamander			SSC	LC		
Dicamptodontidae							
Dicamptodon ensatus	California giant salamander			SSC	NT		
Dicamptodon tenebrosus	Pacific giant salamander				LC		

Plethodontidae							
Aneides ferreus	Clouded salamander				NT		
Aneides flavipunctatus	Black salamander				NT		
Aneides flavipunctatus niger	Santa Cruz black salamander			SSC	NT		
Aneides flavipunctatus "shasta"[11]	Shasta black salamander				NT		
Aneides lugubris	Arboreal salamander				LC		
Aneides vagrans	Wandering salamander				NT		
Batrachoseps altasierrae	Greenhorn Mountains slender salamander						
Batrachoseps attenuatus	California slender salamander				LC		
Batrachoseps bramei	Fairview slender salamander					S	
Batrachoseps campi	Inyo Mountains salamander	X		SSC	E	S	S
Batrachoseps diabolicus	Hell Hollow slender salamander	X			DD		
Batrachoseps gabrieli	San Gabriel Mountains slender salamander	X			DD	S	
Batrachoseps gavilanensis	Gabilan Mountains slender salamander				LC		
Batrachoseps gregarius	Gregarius slender salamander	X			LC		
Batrachoseps incognitus	San Simeon slender salamander	X			DD	S	
Batrachoseps kawia	Sequoia slender salamander	X			DD		
Batrachoseps luciae	Santa Lucia Mountains slender salamander	X			LC		
Batrachoseps major aridus	Desert slender salamander	X	E	E	LC		
Batrachoseps major major	Garden slender salamander				LC		
Batrachoseps minor	Lesser slender salamander	X		SSC	DD	S	
Batrachoseps nigriventris	Black-bellied slender salamander				LC		

(continued)

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
		Caudata					
Batrachoseps pacificus	Channel Islands slender salamander	X			LC		
Batrachoseps regius	Kings River slender salamander	X			V	S	
Batrachoseps relictus	Relictual slender salamander	X		SSC	DD	S	
Batrachoseps robustus	Kern Plateau salamander	X			NT		
Batrachoseps simatus	Kern Canyon slender salamander	X		T	V	S	
Batrachoseps stebbinsi	Tehachapi slender salamander	X		T	V		S
Ensatina eschscholtzii croceater	Yellow blotched ensatina	X			LC	S	S
Ensatina eschscholtzii eschscholtzii	Monterey ensatina				LC		
Ensatina eschscholtzii klauberi	Large-blotched ensatina	X			LC	S	
Ensatina eschscholtzii oregonensis	Oregon ensatina				LC		
Ensatina eschscholtzii picta	Painted ensatina				LC		
Ensatina eschscholtzii platensis	Sierra Nevada ensatina				LC		
Ensatina eschscholtzii xanthoptica	Yellow-eyed ensatina				LC		
Hydromantes brunus	Limestone salamander	X		T, FP	V	S	S
Hydromantes platycephalus[12]	Mount Lyell salamander	X			LC		
Hydromantes shastae	Shasta salamander	X		T	V	S	S
Plethodon asupak	Scott River salamander	X		T	V		
Plethodon dunni	Dunn's salamander				LC		
Plethodon elongatus	Del Norte salamander	X			NT		
Plethodon stormi	Siskiyou Mountains salamander	X		T	E	S	
Rhyacotritonidae							
Rhyacotriton variegatus	Southern torrent salamander	X		SSC	LC	S	

Salamandridae							
Taricha granulosa	Rough-skinned newt				LC		
Taricha rivularis	Red bellied newt			SSC	LC		
Taricha sierrae	Sierra newt				LC		
Taricha torosa	Coast Range newt	X		SSC[1]	LC		
	Squamata—Lizards						
Anguidae							
Elgaria coerulea coerulea	San Francisco alligator lizard				LC		
Elgaria coerulea palmeri	Sierra Nevada alligator lizard				LC		
Elgaria coerulea principis	Northwestern alligator lizard				LC		
Elgaria coerulea shastensis	Shasta alligator lizard				LC		
Elgaria multicarinata multicarinata	California alligator lizard				LC		
Elgaria multicarinata scincicauda	Oregon alligator lizard				LC		
Elgaria multicarinata webbii	San Diego alligator lizard				LC		
Elgaria panamintina	Panamint alligator lizard	X		SSC	V	S	S
Anniellidae							
Anniella pulchra pulchra[16]	Silvery legless lizard	X		SSC	LC	S	
Anniella pulchra nigra	Black legless lizard	X		SSC	LC	S	
Crotophytidae							
Crotaphytus bicinctores	Great Basin collared lizard				LC		
Crotaphytus vestigium	Baja California collared lizard				LC		
Gambelia copeii	Cope's leopard lizard			SSC	LC		
Gambelia sila	Blunt-nosed leopard lizard	X	E	E, FP	E		
Gambelia wislizenii	Long-nosed leopard lizard				LC		

(continued)

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
		Squamata—Lizards					
Gekkonidae							
Coleonyx switaki	Barefoot gecko	X		T	LC		S
Coleonyx variegatus abbotti	San Diego banded gecko	X		SSC	LC		
Coleonyx variegatus variegatus	Desert banded gecko				LC		
Phyllodactylus nocticolus	Peninsular leaf-toed gecko				LC		
Helodermatidae							
Heloderma suspectum cinctum	Banded Gila monster	X		SSC	NT		S
Iguanidae							
Dipsosaurus dorsalis	Desert iguana				LC		
Sauromalus ater	Common chuckwalla				LC		
Phrynosomatidae							
Callisaurus draconoides	Zebra-tailed lizard				LC		
Petrosaurus mearnsi	Banded rock lizard				LC		
Phrynosoma blainvillii[15]	Coast horned lizard	X		SSC	LC	S	S
Phrynosoma douglasii	Pigmy short-horned lizard				LC		
Phrynosoma mcallii	Flat-tailed horned lizard	X		SSC	NT	S	S
Phrynosoma platyrhinos calidiarum	Southern desert horned lizard				LC		
Phrynosoma platyrhinos platyrhinos	Northern desert horned lizard				LC		
Sceloporus graciosus gracilis	Western sagebrush lizard				LC		
Sceloporus graciosus graciosus	Northern sagebrush lizard	X			LC		S
Sceloporus graciosus vandenburgianus	Southern sagebrush lizard				LC		

Sceloporus magister uniformis[16]	Yellow-backed desert spiny lizard				LC		
Sceloporus magister transversus	Barred desert spiny lizard				LC		
Sceloporus occidentalis becki	Island fence lizard				LC		
Sceloporus occidentalis biseriatus	San Joaquin fence lizard				LC		
Sceloporus occidentalis bocourtii	Coast Range fence lizard				LC		
Sceloporus occidentalis longipes	Great Basin fence lizard				LC		
Sceloporus occidentalis occidentalis	Northwestern fence lizard				LC		
Sceloporus occidentalis taylori	Sierra fence lizard				LC		
Sceloporus orcutti	Granite spiny lizard				LC		
Uma inornata	Coachella Valley fringe-toed lizard	X	T	E	E		
Uma notata	Colorado Desert fringe-toed lizard	X		SSC	NT		S
Uma scoparia	Mojave fringe-toed lizard	X		SSC	LC		S
Urosaurus graciosus	Long-tailed brush lizard				LC		
Urosaurus nigricaudus	Baja California brush lizard				LC		
Urosaurus ornatus	Ornate tree lizard				LC		
Uta stansburiana elegans	Western common side-blotched lizard				LC		
Uta stansburiana nevadensis	Nevada common side-blotched lizard				LC		
Uta stansburiana stansburiana	Northern common side-blotched lizard				LC		
Scincidae							
Plestiodon gilberti	Gilbert's skink				LC		
Plestiodon skiltonianus skiltonianus	Western skink				LC		
Plestiodon skiltonianus interparietalis	Coronado skink	X			LC		S
Teiidae							
Aspidoscelis hyperythra	Orange-throated whiptail	X			LC	S	

(continued)

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
		Squamata—Lizards					
Aspidoscelis tigris munda	California whiptail				LC		
Aspidoscelis tigris stejnegeri	Coastal whiptail	X		SSC	LC		
Aspidoscelis tigris tigris	Great Basin whiptail				LC		
Xantusiidae							
Xantusia gracilis	Sandstone night lizard	X		SSC	V		
Xantusia henshawi	Henshaw's night lizard				LC		
Xantusia riversiana	Island night lizard	X			LC		
Xantusia vigilis sierrae	Sierra night lizard	X		SSC	LC		
Xantusia vigilis vigilis	Desert night lizard				LC		
Xantusia wigginsi	Baja California night lizard				LC		
Xantusia sp. "Yucca Valley"	Yucca Valley night lizard				LC		
Xantusia sp. "San Jacinto"	San Jacinto night lizard				LC		
		Squamata—Snakes					
Boidae							
Charina bottae bottae	Rubber boa				LC		
Charina bottae umbratica	Southern rubber boa	X		T	LC	S	
Lichanura orcutti[13]	California rosy boa	X			LC	S	
Colubridae							
Arizona elegans candida	Mojave glossy snake				LC		
Arizona elegans eburnata	Desert glossy snake				LC		
Arizona elegans occidentalis	California glossy snake			SSC	LC		
Bogertophis rosaliae	Baja California rat snake	X			LC		

Chionactis occipitalis annulata	Colorado shovel-nosed snake				LC		
Chionactis occipitalis occipitalis	Mojave shovel-nosed snake				LC		
Chionactis occipitalis talpina	Nevada shovel-nosed snake				LC		
Coluber constrictor mormon	Western yellow-bellied racer				LC		
Contia longicauda	Forest sharp-tailed snake				LC		
Contia tenuis	Common sharp-tailed snake				LC		
Diadophis punctatus "Coastal CA"[19]	Ring-necked snake				LC		
Diadophis punctatus "Eastern CA"	Ring-necked snake				LC		
Diadophis punctatus "Southern CA"	Ring-necked snake	X			LC	S	
Diadophis punctatus "Great Basin"[20]	Ring-necked snake			SSC	LC		
Hypsiglena chlorophaea	Northern desert night snake				LC		
Hypsiglena ochrorhyncha klauberi	San Diego night snake				LC		
Hypsiglena ochrorhyncha nuchalata	California night snake				LC		
Lampropeltis californiae	Common kingsnake				LC		
Lampropeltis multifasciata[21]	California mountain kingsnake	X			LC	S	S
Lampropeltis zonata	California mountain kingsnake				LC		S
Masticophis flagellum piceus[22]	Red coachwhip				LC		
Masticophis flagellum ruddocki	San Joaquin coachwhip	X		SSC	LC		
Masticophis fuliginosus	Baja California coachwhip			SSC	LC		
Masticophis lateralis euryxanthus	Alameda striped racer	X	T	T	LC		
Masticophis lateralis lateralis	California striped racer				LC		
Masticophis taeniatus	Striped whipsnake				LC		
Phyllorhynchus decurtatus	Spotted leaf-nosed snake				LC		
Pituophis catenifer affinis	Sonoran gopher snake				LC		
Pituophis catenifer annectens	San Diego gopher snake				LC		

(continued)

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
	Squamata–Snakes						
Pituophis catenifer catenifer	Pacific gopher snake				LC		
Pituophis catenifer deserticola	Great Basin gopher snake				LC		
Pituophis catenifer pumilis	Santa Cruz Island gopher snake	X			LC		
Rhinocheilus lecontei	Long-nosed snake				LC		
Salvadora hexalepis hexalepis	Desert patch-nosed snake				LC		
Salvadora hexalepis mojavensis	Mojave patch-nosed snake				LC		
Salvadora hexalepis virgultea	Coast patch-nosed snake	X		SSC	LC		
Sonora semiannulata	Western ground snake				LC		
Tantilla hobartsmithi	Southwestern black-headed snake				LC		
Tantilla planiceps	California black-headed snake				LC		
Thamnophis atratus atratus	Santa Cruz aquatic garter snake				LC		
Thamnophis atratus hydrophilus	Oregon aquatic garter snake				LC		
Thamnophis atratus zaxanthus	Diablo Range aquatic garter snake				LC		
Thamnophis couchii	Sierra (western aquatic) garter snake				LC		
Thamnophis elegans elegans	Mountain terrestrial garter snake				LC		
Thamnophis elegans terrestris	Coast terrestrial garter snake				LC		
Thamnophis elegans vagrans	Wandering terrestrial garter snake				LC		
Thamnophis gigas	Giant garter snake	X	T	T	V		
Thamnophis hammondii	Two-striped garter snake	X		SSC	LC	S	S
Thamnophis marcianus	Checkered garter snake						
Thamnophis ordinoides	Northwestern garter snake				LC		
Thamnophis sirtalis fitchi	Valley garter snake				LC		

Thamnophis sirtalis infernalis[23]	California red-sided garter snake	X		SSC[24]	LC		
Thamnophis sirtalis tetrataenia	San Francisco garter snake	X	E	E, FP	LC		
Trimorphodon lambda	Sonoran lyre snake						
Trimorphodon lyrophanes	Peninsular lyre snake						
Leptotyphlopidae							
Rena humilis humilis[25]	Southwestern blind snake				LC		
Rena humilis cahuilae	Desert blind snake				LC		
Viperidae							
Crotalus atrox	Western diamond backed rattlesnake				LC		
Crotalus cerastes cerastes	Mojave Desert sidewinder				LC		
Crotalus cerastes laterorepens	Colorado Desert sidewinder				LC		
Crotalus mitchellii	Speckled rattlesnake				LC		
Crotalus oreganus helleri[26]	Southern Pacific rattlesnake				LC		
Crotalus oreganus lutosus	Great Basin rattlesnake				LC		
Crotalus oreganus oreganus	Northern Pacific rattlesnake				LC		
Crotalus ruber	Red diamond rattlesnake	X		SSC	LC	S	
Crotalus scutulatus	Northern Mojave rattlesnake				LC		
Crotalus stephensi	Panamint rattlesnake				LC		
	Testudines						
Emydidae							
Emys marmorata marmorata	Northern western pond turtle	X		SSC	V	S	
Emys marmorata pallida	Southern western pond turtle	X		SSC	V	S	S

(continued)

Taxon[1]	Common name	CDFG special animal	USFWS[2]	CDFW[3]	IUCN[4]	USFS[5]	BLM
		Testudines					
Kinosternidae							
Kinosternon sonoriense	Sonora mud turtle	X		SSC	V		
Testudinidae							
Gopherus agassizii	Mohave Desert tortoise	X	T	T	V		

1. Species, subspecies, or Distinct Population Segment (DPS)

2. E: Endangered, T: Threatened

3. E. Endangered; T: Threatened, FP: Fully Protect, SSC Species of Special Concern

4. E. Endangered; V: Vulnerable; NT: Near Threatened; LC Least Concern, DD: Data Deficient.

5. S: Sensitive

6. Frost et al. (2006a) recommend placing all California bufonids except *Bufo alvarius* in the genus *Anaxyrus*. Frost et al (2009b) recommend that *B. alvarius* be placed in the genus *Incilius*.

7. Recuero et al. (2006a, 2006b) propose breaking *Pseudacris regilla* (sensu lato) into three distinct species. This proposal has not been widely accepted because the range boundaries of the three taxa are poorly characterized and significant haplotype sharing exists across these putative lineages that has not been studied.

8. This frog was widely introduced in California at one point, though presumed native populations were also present. The taxon may now be extirpated.

9. Frost et al. (2006a) recommend placing *Rana pipiens* and *R. muscosa* in the genus *Lithobates*

10. It is likely that any populations on the eastern side of the Warner Mountains are actually *Rana luteiventris*. However no specimens or data exist to clarify this issue. Until new data become available, *R. luteiventris* cannot be definitively included as a member of the Californian herpetofauna.

11. Following Kissler and Apodaca (2007)

12. An Owens Valley population was formerly presumed to be an undescribed taxon and has become widely recognized in the conservation community. Rovito (2010) refutes its status as a distinct lineage and we include the Owens Valley populations with *Hydromantes platycephalus*

13. Status applies only to Monterey County, CA, and south

14. Papenfuss and Parham (2013) proposed splitting *Anniella pulchra* in California into five species.

15. Leaché et al. (2009) revised the *Phrynosoma coronatum* complex, placing California populations of *P. coronatum* into *P. blainvillii*

16. Schulte et al. (2006) propose that the *Sceloporus magister* subspecies be elevated to full species. This was refuted by Leaché and Mulcahy (2007).

17. Leavitt et al. (2007) find a significant genetic structure within the *Xantusia vigilis* complex. Taxonomic revisions may occur in the near future within this clade

18. Wood et al. (2008) divided the rosy boas into two species, *Lichanura orcutti* and *L. trivirgata*. Their mitochondrial data indicate that *L. trivirgata* is present in extreme southern California, though newer unpublished nuclear data suggest that the species break actually occurs farther south, in Baja California, Mexico (D. Wood, pers. comm.)

19. Feldman and Spicer (2006) and Fontanella et al. (2008) find evidence for lineages that are not concordant with previously described subspecies boundaries. We follow the lineage designations from the latter study

20. The Great Basin clade includes animals formerly assigned to *Diadophis punctatus regalis*. The SSC status refers only to populations occurring at isolated desert springs in Southern California.

21. Mountain kingsnake taxonomy is in flux. Rodríguez-Robles et al. (1999b) refute the formerly recognized subspecies and find evidence for four distinct lineages. Myers et al. (2013) find evidence for two species (the arrangement that we follow here). *Lampropeltis multifasciata* contains the former southern subspecies *Lampropeltis zonata parvirubra* and *L. z. pulchra*. The conservation status applies to these two subspecies

22. Nagy et al. (2004) propose combining *Masticophis* into the genus *Coluber*.

23. Southern populations of this subspecies may represent a distinct taxon and are currently under study (C. Mahrdt, pers. comm., E. Ervin, pers. comm.).

24. SSC status applies to only the southern portion of the range.

25. Adalsteinsson et al. (2009) propose placing California *Leptotyphlops* in the genus *Rena*.

26. Some authors treat the subspecies of *Crotalus oreganus* as distinct species.

27. Some authors place the western pond turtles in the monotypic genus *Actinemys*. Spinks et al. (2014) recommend elevating both pond turtle subspecies to species status.

APPENDIX 2

Public Comment Announcement

We solicited public comment on this project by posting the announcement on the right on the websites of the following organizations: California Department of Fish and Wildlife, Center for North American Herpetology, Ecological Society of America (ECOLOG-L), Partners in Amphibian and Reptile Conservation, and The Wildlife Society. In addition, we circulated the announcement widely to colleagues via email. Following the public comment period, we also contacted experts on each taxon under consideration to request advice, data, and reviews of early drafts of this document.

California's list of Amphibian and Reptile Species of Special Concern (ARSSC) is a critical component of the management and protection of amphibians and reptiles in the state. The current California ARSSC list is undergoing a complete revision to better reflect those taxa that require some measure of conservation to stabilize populations and avoid future listing under the California Endangered Species Act. To date, the ARSSC revision team has developed a set of risk metrics, compiled a list of nominee taxa, and completed a preliminary risk assessment for each nominee based on literature reviews and locality information. Now, we need your help to make sure that we have the most accurate and complete list possible of SSC for potential inclusion in the final list. The best list will require input from as many knowledgeable biologists as possible. If you have data, well-documented field experience, or unpublished observations that are relevant to California's amphibian and reptile fauna, we invite you to share them with us.

Further details, risk assessments, and instructions for submitting feedback are available at http.//arssc.ucdavis.edu. The public comment period closes August 31st, 2009.

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APPENDIX 3

Watch List

The watch list comprises taxa that were previously, but are no longer, considered Species of Special Concern. Here we include an explanation for each taxon's change in status and discuss future conservation concerns regarding Watch List taxa.

California tiger salamander

(*Ambystoma californiense*)

Jennings and Hayes (1994a) identified this species as the highest-concern vernal pool-breeding amphibian in the state. In keeping with this assessment and recent research documenting its decline range-wide, *A. californiense* was listed under the California Endangered Species Act as a Threatened species in 2010, superseding Species of Special Concern status. See Bolster (2010) for the CDFW's recent status review. The species was also listed under the federal Endangered Species Act in 2000 (Santa Barbara; Endangered), 2003 (Sonoma; Endangered), and 2004 (Central; Threatened), as three separate Distinct Population Segments. Recent multi-locus phylogeographic work indicates that the Central Distinct Population Segment is composed of two separate lineages from the Inner Coast Range and Central Valley and that these may be best considered as separate units with different management needs (J. Johnson and B. Shaffer, unpublished data).

Orange-throated whiptail

(*Aspidoscelis hyperythra*)

This taxon was included by Jennings and Hayes (1994a) primarily because of habitat loss within its relatively narrow range. We place it on the Watch List because, thus far, it appears to tolerate habitat fragmentation better than many similarly distributed taxa, including the red diamond rattlesnake (*Crotalus ruber*), coast patch-nosed snake (*Salvadora hexalepis virgultea*), and California glossy snake (*Arizona elegans occidentalis*), all of which have experienced more severe declines; and it remains relatively common in many areas throughout its range. It is possible that further development and habitat fragmentation could cause more severe declines, so this taxon should be periodically reevaluated.

Baja California rat snake

(*Bogertophis rosaliae*)

Jennings and Hayes (1994a) included the *B. rosaliae* primarily as a precaution. Virtually nothing was known about the species in California except that, if it ever naturally occurred in the state, it was probably rare and restricted in distribution (only a single specimen has ever been recorded). In the intervening time, no additional specimens have been reported, and no new information has become available for this species. If this species is found to be a native component of the California fauna, the conservation status should be reevaluated when more is known about the populations and habitat of the snake in California.

Yellow-blotched ensatina

(*Ensatina eschscholtzii croceater*)

Jennings and Hayes (1994a) included this taxon primarily over concerns about land use changes within its small range. We shared several of these concerns, although the severity of these threats appears to have decreased since 1994. As long as the planned preservation areas at Tejon Ranch remain in effect, a large amount of *E. e. croceater* habitat will remain protected, so designation as a Species of Special Concern may not be necessary. We include *E. e. croceater* on the Watch List to encourage reevaluation of habitat availability for this taxon in the future.

Large-blotched ensatina

(*Ensatina eschscholtzii klauberi*)

Jennings and Hayes (1994a) included this taxon primarily over concerns about ongoing development within its range. We agree that development has had, and is continuing to have, an impact on this species, although the severity of these impacts appears to be significantly less than those being experienced by other taxa with similar ranges. Further, the large-blotched Ensatina appears to be commonly found with stable populations throughout significant areas of its range, including protected parklands. If the extent of development increases within this salamander's range, it may become necessary to reconsider special concern status and more active management.

Mount Lyell web-toed salamander

(*Hydromantes platycephalus*)

This taxon was included by Jennings and Hayes (1994a) as a precaution, based on its patchy distribution and suspected susceptibility to local extirpations. We do not include *H. platycephalus* at this time because, although it is patchily distributed, the species appears to be stable throughout most of its range and is not experiencing appreciable risk from habitat disturbance (Wake and Papenfuss 2005). Additional populations have been found since the early 1990s, and the species appears to be relatively common at many sites. Although it is a California endemic, has a moderately small range, and is a narrow ecological specialist, this species does not appear to be currently at risk of immediate decline (Wake and Papenfuss 2005).

Owens Valley web-toed salamander

(*Hydromantes platycephalus*)

The Owens' Valley populations of *H. platycephalus* were included by Jennings and Hayes (1994a) as a precaution, both because little was known about the population biology of this elusive salamander and because it was strongly suspected that it was a distinct taxon. Research completed since 1994 suggests that these populations do not form a distinct lineage but instead are part of the more broadly distributed *H. platycephalus* lineage (Rovito 2010). As with *H. platycephalus*, additional localities have been found and populations appear to be stable, leading us to conclude that Species of Special Concern designation is not required at the present time (Wake and Papenfuss 2005).

Southern California mountain kingsnakes

(*Lampropeltis zonata parvirubra* and *L. z. pulchra*)

The two southern California subspecies *L. z. parvirubra* and *L. z. pulchra* were considered Species of Special Concern by Jennings and Hayes (1994a) on the basis of suspected declines due to illegal collecting and habitat destruction from some collectors. We agree that this has occurred, although the current scale of exploitation does not appear to threaten this species' long-term survival. We placed the species on the Watch List in recognition that collection pressure and/or habitat destruction could cause the need to provide additional protections in the future.

Santa Cruz Island gopher snake

(*Pituophis catenifer pumilis*)

Jennings and Hayes (1994a) included this taxon primarily because of its small range (it is restricted to Santa Cruz and Santa Rosa islands) and threats from feral ungulates and pigs. We removed this species from special concern status because the invasive mammals causing the primary threats have been removed from the largest part of the range, Santa Cruz Island (USNPS 2010). This island is also well protected from future development because it is a national park.

Coronado skink

(*Plestiodon skiltonianus interparietalis*)

Jennings and Hayes (1994a) included *P. s. interparietalis* primarily because it has a relatively restricted range and has disappeared from some areas. As with *Aspidoscelis hyperythra*, we agree that some declines have occurred, although their severity appears to be modest. If these declines continue, further protections may be warranted in the future.

Del Norte salamander

(*Plethodon elongatus*)

Jennings and Hayes (1994a) included the Del Norte salamander because of concerns regarding habitat specialization by inland populations and the potential for timber harvest to destroy these habitats. Although these are valid concerns, as well as for two close relatives of *P. elongatus*, the Scott Bar salamander (*Plethodon asupak*) and Siskiyou Mountains salamander (*P. stormi*), population status across most of the range of this taxon appears to be stable. Inland populations are patchy and likely more vulnerable to habitat degradation, which is why we place this taxon on our Watch List (H. Welsh, pers. comm.).

Mountain yellow-legged frogs

(*Rana muscosa* and *R. sierrae*)

Mountain yellow-legged frogs were designated as Species of Special Concern by Jennings and Hayes (1994a) under the name *R. muscosa*. Vredenburg et al. (2007) divided *R. muscosa* (sensu lato) into two species on the basis of morphometric measurements, differences in advertisement call, and mitochondrial DNA: the Sierra Madre yellow-legged frog (*R. muscosa*) in the south and the Sierra Nevada yellow-legged frog (*R. sierrae*) in the north. Both species were state listed in 2013, superseding Species of Special Concern status. See Bonham and Lockhart (2011) for the CDFW's recent status review of these taxa.

APPENDIX 4

Additional Taxa in Need of Research and Monitoring

We identified the following taxa that did not qualify for Species of Special Concern status but nonetheless would benefit from some level of additional research and/or monitoring. We provide a brief description of our concerns for each of these taxa below.

Orange-throated whiptail

(*Aspidoscelis hyperythra*)

Aspidoscelis hyperythra occurs in California in a relatively narrow region of southern California. Much of its available habitat has been destroyed or is threatened by ongoing urbanization and development. Further, many of the areas where habitat persists have become fragmented by development in intervening areas. The taxon remains locally common in several areas, although this should be reevaluated periodically. Further habitat modification could lead to more declines that warrant additional protections. Additional threats may arise from increasing intensity and/or frequency of wildfire in the region.

San Gabriel Mountains slender salamander

(*Batrachoseps gabrieli*)

Batrachoseps gabrieli occurs in a small area in Los Angeles and San Bernardino Counties (Stebbins 2003). Very few localities are known for this taxon, and its range is probably not fully characterized (Goodman et al. 1998, Hansen et al. 2005d). The salamander appears to be limited to talus slopes in the vicinity of oak, big cone spruce, and pine (Wake 1996, Goodman et al. 1998). It exhibits limited surface activity and appears to specialize on an environment that is unlikely to be developed. This species' known range lies within the boundaries of the Angeles and San Bernardino National Forests and appears to be well protected at the present time. However, other narrowly distributed species of *Batrachoseps* have undergone large and unexplained declines, and it is possible that similar declines could occur for this species (Jennings and Hayes 1994a). For this reason, periodic monitoring and reevaluation of status of *B. gabrieli* is warranted.

Baja California rat snake

(*Bogertophis rosaliae*)

Bogertophis rosaliae is known only from a single road-killed specimen in California along Interstate 8 (specimen SDNHM 64416). It is unclear if this represents an escaped or discarded pet, a rare migrant from the known range farther south in Baja California, Mexico, or a regular, infrequently encountered component of the California reptile fauna. If a population does exist in California, ongoing development along the border in both the United States and Mexico is likely to isolate these populations from the main part of the range, which occurs farther south. If so, the California populations could be susceptible to stochastic effects associated with small populations, as well as habitat loss from development. In some areas this species appears to be associated with palm oases, which are uncommon habitat patches, so any degradation of this habitat may have severe impacts on the taxon.

If this species is native to California, it appears to be encountered exceedingly rarely and is never reported. Given this complete uncertainty concerning its status and validity as a native element of the California fauna, we place this taxon on the Watch List, primarily to highlight research needs. Surveys for this taxon should be encouraged, although in the absence of additional data, specimen collection should be strictly limited to only what is needed to learn more about its natural history and status within the state. However, we emphasize that tissue samples might help determine if any California specimens are native or introduced.

Yellow-blotched ensatina

(*Ensatina eschscholtzii croceater*)

Ensatina eschscholtzii croceater occurs in a relatively small area of Kern and Ventura Counties in southern California. Some localized populations may have undergone declines or extirpations due to development, although data on this are scarce. Workers have expressed concerns about land use practices and development in the Tehachapi Mountains, Bear Valley, Cummings Valley, and Tejon Ranch, particularly in areas of oak woodlands (pers. comm. in Jennings and Hayes 1994a). One of the main concerns for this taxon was that a large fraction of its range occurs on property owned by the Tejon Ranch Company, the largest contiguous private landholding in California, and that this land would be developed in a way that was incompatible with the salamander's survival. Since the previous evaluation, a large fraction of Tejon Ranch has been set aside for preservation areas in which grazing, but not development, may continue (Tejon Ranch Conservancy 2008). In addition, many populations occur on National Forest and other public lands that are unlikely to experience intense habitat modification. The availability of suitable habitat should be monitored periodically, and habitat modification within its very restricted range should be avoided.

Southern California mountain kingsnake

(*Lampropeltis zonata* "Southern Clade" or *L. multifasciata*)

The southern clade of *L. zonata* includes the formerly recognized subspecies *L. z. pulchra* and *L. z. parvirubra* (Rodríguez-Robles et al. 1999b), and has more recently been recognized at the species level as *L. multifasciata* (Myers et al. 2013). This snake specializes on rocky outcrop habitats occurring primarily in a variety of woodland and chaparral habitats from sea level to nearly 3000 m (Stebbins 2003). It is a popular species among herpetoculturists and collectors, and some have voiced concerns that habitat destruction has caused localized declines. Overzealous collection of this snake does tend to destroy the microhabitats within rocks, which can degrade the quality of sites for a long period of time, although the species exhibits a relatively narrow window of surface activity, and much of its habitat may be relatively inaccessible to collectors. Staub and Mulks (2009) surveyed the Mount Laguna region, San Diego County, from 2006 to 2008 and found that 75% of all rock piles surveyed had some degree of damage. They concluded that collecting is ongoing and is not restricted to the vicinity of roads, supporting the concerns that the intensity of ongoing collecting could harm this species. Managers should be wary of signs of habitat destruction, stemming from either collectors or other sources, particularly in areas that experience heavy human traffic such as Mount Laguna. If surveys demonstrate that these collecting activities are depleting populations, further management and enforcement of existing collecting prohibitions may be needed.

Del Norte salamander

(*Plethodon elongatus*)

Plethodon elongatus occurs from the California–Oregon border south into Humboldt and Trinity Counties. Optimal habitat for this taxon appears to be late successional and mature forests, which may be increasingly impacted by timber harvest in the coming years (Welsh and Lind 1995; H. Welsh, pers. comm.). Prior to 2002, this species was managed under the Northwest Forest Plan (Welsh and Bury 2005, Survey and manage program 2010). These protections have now been removed, although much of the habitat that supports this taxon remains protected under the Plan (Northwest Forest Plan 1994). Monitoring efforts should focus on the impact of timber harvest on this species' ability to persist, particularly at inland sites.

Western black-headed snake

(*Tantilla planiceps*)

The natural history of *T. planiceps* is poorly understood in California. We have almost no information concerning this species' natural history, habitat requirements, or population densities. The snake seems to be patchily distributed and rarely seen, making the detection of population declines or extirpations difficult. In addition, much of its range occurs in areas that have experienced heavy development and habitat modification. Some workers have suggested that changing wildfire regimes in southern California could be having a negative impact on this species; however, relevant data are very sparse. An important priority for this taxon is an increased research effort focused on distribution and habitat surveys so that its ecological requirements and population dynamics can be better characterized. As populations are discovered, tissue samples should be collected for molecular analyses of the degree of isolation and differentiation of these apparently disjunct populations.

Baja California night lizard

(*Xantusia wigginsi*)

Xantusia wigginsi was not known to be a part of the California lizard fauna until recent genetic studies established its presence in extreme southern California (Leavitt et. al. 2007). Virtually nothing is known about this taxon's range, life history, habitat requirements, or conservation status within California. Further research on this species is needed before assessments of its conservation status and management needs can be made.

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GLOSSARY

ADPRESSED LIMBS Position of the limbs such that the forelimbs are pressed backwards against the trunk of the animal, and the hind limbs are pressed forward against the trunk. The distance between adpressed limbs, a character which measures the relative limb length with respect to the trunk length, is usually best measured in preserved specimens, since the limbs may be damaged in living animals.

ALLOPATRIC Occurring in separate areas; refers to species ranges that do not overlap

ALLOZYME Alleles of an enzyme that vary in their speed of migration through an electrophoretic gel. A common way to quantify genetic variation before DNA sequencing became routine.

AMPLEXUS Mating behavior in many aquatic anurans and some salamanders in which the male grasps the female with the front legs.

BD *Batrachochytrium dendrobatidis.* A pathogenic fungus that causes the disease chytridiomycosis in many amphibians.

CARAPACE The dorsal half of a turtle shell.

COSTAL GROOVES Lateral indentations along the trunk of many salamanders.

CRITICAL THERMAL MAXIMUM The temperature above which a given species ceases to be able to maintain normal body function. Extended temperatures above this point generally lead to death.

CRITICAL THERMAL MINIMUM The temperature below which a given species ceases to be able to maintain normal body function. Extended temperatures below this point generally lead to death.

CRYPTIC TAXA Evolutionarily distinct lineages that are morphologically conserved and are difficult to distinguish from one another on the basis of morphology alone.

DIAPAUSE A delay in the life cycle of an organism, often occurring in response to adverse environmental conditions.

DORSOLATERAL FOLDS Ridges of the skin that run along either side of the back in many frogs.

EXTANT A taxon that is still in existence, opposite of extinct.

HIBERNACULUM A place used by one or more individuals to hibernate or undergo a period of dormancy. Frequently used to refer to areas that house many hibernating individuals of the same species, especially sites that are used repeatedly over many years. The plural is hibernacula.

INTROGRESSION Transfer of genetic molecules from one species to another. In our usage, this most commonly refers to the transfer of the mitochondrial genome among species due to hybridization.

ISOLATION BY DISTANCE The genetic signature that tends to arise from the tendency of individuals within a population to mate with nearby individuals, eventually leading to the gradual accumulation of genetic differentiation across the landscape.

KEELED A spine or ridge structure that runs along the central axis of a scale or scute.

LATE-SERAL Used to describe forests that are in a later stage of succession. Typified by the presence of large, old (>100 years) trees in the overstory.

MICROSATELLITE Short repetitive regions in the DNA that often exhibit a large amount of variation due to the very high rate of mutation in these regions of the genome. Frequently employed to measure population genetic variation within species, because their high mutation rate allows them to track changes in gene flow and population size quickly.

MTDNA An abbreviation for mitochondrial DNA, the separate chromosome found in the mitochondria of all plants and animals. Until recently, it has been the standard molecule of choice for most systematic, population genetic, and phylogeographic research.

NASOLABIAL GROOVES Characteristic grooves that run from each naris (external nostril) down to the upper lip in plethodontid salamanders.

NUCHAL Relating to or lying in the region of the nape.

OCELLUS An eye-like spot.

OVIPAROUS A mode of reproduction in which embryos develop inside of eggs.

OVOVIVIPAROUS A mode of reproduction in which embryos develop inside of eggs which are retained in the mother's body until hatching.

PAEDOMORPHOSIS The retention of larval traits into adulthood. In ambystomatid and dicamptodontid salamanders, it is also used to refer to reproduction in the larval condition.

PARAPHYLETIC A group of taxa, all descending from of a common ancestor, that does not contain all descendants of that ancestor. For examples, "reptiles" as traditionally defined are paraphyletic because they do not contain birds as a contained taxon.

PARATOID GLANDS External skin glands that lie along the back of the head or neck region and are prominent in most toads and several species of salamander.

PCA Principle component analysis. A multivariate ordination approach that reduced the variability among large sets of measured variables down to a (usually) smaller number of independent (orthogonal) variables.

PIT TAG Passive integrated transponder tag. A small injectable tag that emits a unique electronic signal that can be read using specialized instruments. A frequently used method for uniquely labeling individual organisms in a population.

PLASTRON The ventral part of a turtle shell.

POLYTYPIC Having several morphological forms. These may or may not correspond to evolutionary lineages.

POND TYPE LARVAE Salamander larvae that develop in ponds are characterized by having relatively large long fins associated with a relatively strong swimming ability.

SCUTE An enlarged scale, such as those on a turtle shell.

SCL Straight carapace length. The distance from the anterior to the posterior end of the carapace taken along the midline and measured as a straight distance (i.e., not measuring along the curvature of the shell). A standard way of measuring body length in turtles.

SNP Single nucleotide polymorphism. A homologous nucleotide position in a DNA sequence that is variable among conspecific individuals. SNPs are increasingly used instead of allozymes, microsatellites, and mtDNA for population genetic and species delimitation studies.

STREAM TYPE LARVAE Salamander larvae that develop in streams are typically smaller than pond type larvae and have smaller tail fins. Behaviorally, they tend not to swim in the open water and instead remain near the substrate.

SVL Snout to vent length. The distance from the tip of the snout to the anterior edge of the cloaca. A standard way of measuring length in many amphibians and reptiles.

TL Total length. The distance from the tip of the snout to the end of the tail.

VIVIPAROUS A mode of reproduction in which females give birth to live young that are not retained in shelled eggs (compare with ovoviviparous).

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Note: Page number followed by (f), (m), and (t) indicates figure, map, and table respectively.

U.S. Fish and Wildlife Service
California - Great Basin Region
Migratory Birds Program

Recommended Buffer Zones for Ground-based Human Activities around Nesting Sites of Golden Eagles in California and Nevada

May 2021

For most ground-based human activities, we recommend a one-mile no-disturbance buffer surrounding golden eagle nesting sites in California and Nevada; see table below for specifics on activity and buffer recommendations. Recommended buffers may increase or decrease, depending on specific site or activity circumstances. Buffers may be reduced in consultation with the U.S Fish and Wildlife Service (Service) when the nest is not in use or activities are not in line-of-sight of the nest[a]. In parts of California, eagles maintain year-round territories that may require additional protection. We recommend consultation with the Service for determining buffer zones for high intensity or long duration activities, unique circumstances, activities not listed in the table below, or when historic levels of human activity are a consideration.

Activity	Recommended No-Disturbance Buffer
Use of Motorized Vehicles off-road and on water: Including, but not limited to, passenger vehicles, all-terrain vehicles, dirt bikes, and snowmobiles. Any passenger vehicle driving on dirt or gravel roads that are not part of a routinely used transportation corridor. Also includes motorized boating activities.	1 mile
Pedestrian and Non-Motorized Activity[b]: Including, but not limited to, walking, running, hiking, biking, camping, rock climbing, bird watching, fishing, hunting, horseback riding, canoeing, kayaking, and biological surveys[c].	1 mile
Developed Sites: Including, but not limited to, facilities, developed campground sites, and designated snowmobile and off-road vehicle courses.	1 mile
Industrial, Municipal, and Construction Activity: Including, but not limited to, urbanization; mining; oil and gas development; solar development; logging; power line construction; road construction & maintenance; facilities construction; and agricultural operations.	1 mile
Blasting and other loud non-regular noise: Including, but not limited to, detonation devices, fireworks classified by the Federal Department of Transportation as Class B explosives, recreational shooting, and outdoor concerts.	2 miles

[a] An *in-use nest* is defined as a "golden eagle nest characterized by the presence of one of more eggs, dependent young, or adult eagles on the nest in the past 10 days during the breeding season" (50 CFR 22.3) and "(b)reeding begins… with the start of courtship…" (*Programmatic Environmental Impact Statement for the Eagle Rule Revision*, United States Department of the Interior, Fish and Wildlife Service, December 2016).

[b] Many existing nest sites experience some level of intermittent and on-going low levels of disturbance from these types of human activities, and the resident pair of eagles may have acclimated to these existing levels of disturbance. However, increases in human activity may not be tolerated by nesting eagles.

[c] Qualified biologists conducting ground-based eagle monitoring may follow distance recommendations in Pagel et al (2010).

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Joshua tree (*Yucca brevifolia*) seeds are dispersed by seed-caching rodents[1]

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Abstract: Joshua tree (*Yucca brevifolia*) is a distinctive and charismatic plant of the Mojave Desert. Although floral biology and seed production of Joshua tree and other yuccas are well understood, the fate of Joshua tree seeds has never been studied. We tested the hypothesis that Joshua tree seeds are dispersed by seed-caching rodents. We radioactively labelled Joshua tree seeds and followed their fates at five source plants in Potosi Wash, Clark County, Nevada, USA. Rodents made a mean of 30.6 caches, usually within 30 m of the base of source plants. Caches contained a mean of 5.2 seeds buried 3–30 mm deep. A variety of rodent species appears to have prepared the caches. Three of the 836 Joshua tree seeds (0.4%) cached germinated the following spring. Seed germination using rodent exclosures was nearly 15%. More than 82% of seeds in open plots were removed by granivores, and neither microsite nor supplemental water significantly affected germination. Joshua tree produces seeds in indehiscent pods or capsules, which rodents dismantle to harvest seeds. Because there is no other known means of seed dispersal, it is possible that the Joshua tree–rodent seed dispersal interaction is an obligate mutualism for the plant.
Keywords: Joshua tree, scatter hoarding, seed caching, seed dispersal, *Yucca brevifolia*.

Résumé : L'arbre de Josué (*Yucca brevifolia*) est une plante distinctive et charismatique du désert de Mojave. Quoique la biologie de la floraison et la production de graines de cet arbre et des autres yuccas sont bien comprises, le sort des graines de l'arbre de Josué n'a jamais été étudié. Nous avons testé l'hypothèse que les graines de cet arbre sont dispersées par des rongeurs cacheurs de graines. Nous avons marqué radioactivement des graines d'arbres de Josué et suivi leur sort à partir de cinq plants sources à Potosi Wash, Clark County, Nevada, É.-U. Les rongeurs ont fait en moyenne 30.6 caches habituellement à 30 m de la base des plants sources. Les caches contenaient en moyenne 5.2 graines enterrées à des profondeurs de 3 à 30 mm. Une variété d'espèces de rongeurs semblaient avoir fait ces caches. Trois des 836 graines d'arbres de Josué cachées (0,4%) ont germé le printemps suivant. La germination de graines dans des exclos à rongeurs atteignait presque 15%. Plus de 82% des graines dans les sites ouverts ont été enlevées par des granivores et ni le microsite ni l'ajout d'eau n'ont affecté significativement la germination. L'arbre de Josué produit des graines dans des capsules indéhiscentes que les rongeurs brisent pour récolter les graines. Dans le cas de l'arbre de Josué, puisque aucun autre moyen de dispersion des graines n'est connu, il est possible que la dispersion par les rongeurs soit une interaction mutualiste obligée pour cette plante.
Mots-clés : arbre de Josué, cache de graines, dispersion de graines, éparpiller en divers caches, *Yucca brevifolia*.

Nomenclature: Hickman, 1993; Pellmyr, 2003.

Introduction

Yuccas (*Yucca* and *Hesperoyucca*) are prominent and charismatic plants of warm temperate and arid regions of the southern United States and Mexico. Yuccas have received a great deal of scientific attention, especially in the area of floral biology. The yucca moth *Yucca* flower pollination interaction, often cited in textbooks and popular works, is one of the most convincing examples of a coevolved, obligate mutualism (Pellmyr *et al.*, 1996). The interaction, which may have arisen more than 40 million years ago (Pellmyr & Leebens-Mack, 1999), was first described by George Engelmann and Charles Riley in the 1870s and is still an object of intense scientific investigation (Pellmyr, 2003). Although much is known about their genesis, virtually nothing is known about the fate of yucca seeds (*e.g.*, seed dispersal, post-dispersal seed predation, seedling establishment).

Some yuccas (*e.g.*, Joshua tree, *Yucca brevifolia*) produce relatively large seeds. These large seeds are likely to be attractive food items for granivorous rodents, birds, and ants. Further, a number of seed-caching rodents live in the

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same habitat as these yuccas, and it is possible that a fraction of the yucca seeds are scatter hoarded by rodents and that rodents might serve as dispersal agents for these yuccas. Some alternative means of seed dispersal have been suggested. In 1893, William Trelease wrote of Joshua tree fruits that "their rounded form and very light specific gravity render them well developed 'tumble fruits' and point to their dissemination across the desert by the aid of strong winds that prevail there." A number of authors state that yuccas (*i.e.*, *Y. elata*, *Y. filamentosa*, *Y. whipplei*, and *Y. glauca*) are wind/gravity dispersed (Powell & Mackie, 1966; Wallen & Ludwig, 1978; Aker, 1982; Dodd & Linhart, 1994; Massey & Hamrick, 1998; Huth & Pellmyr, 2000), but the effectiveness of this mechanism has never been assessed. Wallen and Ludwig (1978) suggested that *Y. baccata* is dispersed by herbivores, but did not present conclusive evidence. Yeaton *et al.* (1985) also noted that *Y. schidigera*, in addition to *Y. baccata*, is dispersed by vertebrate means, but neither produced evidence or specified which vertebrates. Janzen (1986) suggested that the seeds of *Y. decipiens* in the southern Chihuahuan Desert, Mexico, might have been dispersed by extinct megafauna.

A novel and important reproductive feature of some *Yucca* species is the production of indehiscent seed pods. This is a derived condition, nested within the Agavaceae, while all other members have dry, dehiscent capsules. There are two types of indehiscence: one found in the fleshy-fruited yuccas of the section Sarcocarpa and the other in the spongy-fruited *Yucca brevifolia*, the sole member of the section Clistocarpa. Developmentally, they differ in that the mesocarp layer thickens in sarcocarps, whereas the exocarp layer thickens in *Y. brevifolia*, creating a less dense fruit in the latter. Many yucca species produce dehiscent seed pods, and these may well be dispersed by wind or some other abiotic vector, but it is unclear how seeds from the species with indehiscent pods are dispersed. The yuccas with indehiscent pods produce relatively large seeds that would appear to be sought after by granivores, however. The conjunction of these two traits, indehiscent pods and large seeds, led us to hypothesize that these yuccas, which include Joshua tree, are dispersed by seed-caching rodents that remove the seeds directly from pods. The goal of this study was to assess whether Joshua tree, one of the most familiar and important yuccas in the southwestern United States, is dispersed by seed-caching rodents.

Methods

We studied the dispersal of Joshua tree seeds at Potosi Wash on the southwest slope of the Spring Mountains, about 50 km southwest of Las Vegas, Clark County, Nevada (35° 52' N, 115° 33' W, elev. 1660–1700 m). The site is Mojave Desert scrub dominated by blackbrush (*Coleogyne ramosissima*), Nevada ephedra (*Ephedra nevadensis*), desert almond (*Prunus fasciculata*), and creosote bush (*Larrea tridentata*), with moderately dense populations of Joshua tree and banana yucca (*Y. baccata*) just below pinon–juniper woodland. The site is federal land administered by the US Department of Agriculture, Forest Service and US Department of the Interior, Bureau of Land Management. We studied seedling recruitment of Joshua tree along Walking Box Ranch Road off Nevada route 164 about 12 km west of Searchlight, Nevada.

On 15 August 2002, we placed 200 Joshua tree seeds around each of two source plants (≈20 seeds at each of 10 scrapes in the soil within 1 m of the tree trunk). On 26 September 2002, we placed three more batches of 200 seeds under three other source plants in a similar fashion. Seeds were numbered individually and labelled with radioactive scandium, a gamma-emitting radionuclide with a half-life of 83.8 d. Scandium is biologically inert (does not pass through the food chain), decays to non-toxic, non-radioactive titanium, and appears harmless to rodents and seeds. The way we presented seeds was unnatural, but it is not feasible to present radioactively labelled seeds within pods and, in this study, we were interested in the dispersal process, not how seeds might be harvested from pods. We returned the day after deploying seed to check seed fates. We collected unharvested seeds and then searched for missing seeds using Geiger counters, following the procedure of Vander Wall (2002). We excavated caches to determine number of seeds and burial depth. We recorded the numbers on the seeds and returned them to the cache sites. We recorded caching microhabitat (under shrub [> 10 cm from canopy edge], edge of shrub [< 10 cm of canopy edge], or space between shrubs (> 10 cm from canopy edge]) and substrate (mineral soil or plant litter) and measured the coordinates of cache sites using the source plant as the origin of a graph space and cardinal directions as axes. We returned to source plants later in the fall of 2002 and spring of 2003 and resurveyed the area to determine which caches were present and to search for new (secondary) caches. After the last survey, all remaining labelled seeds were gathered and disposed of.

During fall 2003, after production of relatively large seed crops, we monitored fruits of Joshua tree, banana yucca, and Mojave yucca at our Potosi Wash study site and at Wallace Canyon in the Spring Mountains of southern Nevada to determine how seeds might be removed. Like Joshua tree, banana and Mojave yuccas produce indehiscent seed pods with relatively large seeds.

To determine rodent species composition and relative abundances at the Potosi Wash site, we trapped rodents on the evenings of 30 April and 1 May 2003. We used 98 Sherman live traps in a 7 × 7 grid with 12 m spacing and 2 traps at each point. Two trapping grids were laid out, one located in burned shrublands and the other in unburned vegetation. Bait consisted of mixed bird seed. We baited and set traps just before dusk, and we checked and closed traps at dawn. Trapping sessions lasted two nights.

We conducted seed germination trials using Joshua tree seeds gathered from Carpenter Canyon (Spring Mountains, southern Nevada) in October 2003. We removed seeds from pods and placed them in water; seeds that floated were considered inviable and were discarded. We also excluded any seeds with visible damage from moth larvae. Viable seeds were air dried and stored in paper bags.

To prevent pilferage by rodents, we conducted seed germination experiments within protective exclosures constructed with 6 mm hardware cloth 50 × 50 × 20 cm high, buried 10 cm deep in the soil. In each cage, we planted seeds ≈3 cm deep in a 5 × 10 grid. Wire mesh lids were placed over the cages. We repeated this procedure in 24 exclosures (1200 seeds), spacing them ≈30 m apart along a transect. We placed half (12) of the exclosures under shrubs and the other half in the open, alternating site conditions along the transect. Six exclosures in each microsite treatment were watered each week for five weeks with one litre of water, again alternating along the transect. The other six received no supplemental water. We monitored exclosures weekly for seedling emergence. We initiated this study in March of 2004 and gathered the final data on 29 January 2005.

Nearby, we established 12 unfenced plots to measure the rate of rodent seed pilferage. Fifty Joshua tree seeds were buried ≤3 cm deep in a 5 × 10 pattern within a 50- × 50-cm plot. As with the exclosures, half of these plots were watered each week. We checked plots weekly for seed removal and seedling emergence; however, the number of seeds removed was difficult to determine without disturbing the plots. At the end of two months, the top 5 cm of soil in the plots was removed and sifted for remaining seeds.

We analyzed germination and seed removal data using a two-factor ANOVA for microsite and supplemental watering. Data were square root transformed to meet assumptions of equal variance where necessary (Box & Cox, 1964). Data are reported as mean ± 1 SD unless otherwise noted.

Results

Rodents removed labelled seeds from four of the source plants within 1 d, and seeds at the fifth source plant were removed during the second day. Only 5 of 1000 seeds (0.5%) were not harvested by rodents. None of the seeds had been eaten under the source plants. When we searched for seeds with Geiger counters, we found 84.1 ± 13.1% of removed seeds (range 67.7 to 97.5%) in caches. We found a total of 153 primary caches (30.6 ± 7.4 caches·plant^{-1}) located 0.5–56.6 m away from the base of source plants (Table I). Primary caches contained a mean of 5.2 ± 3.9 seeds (range = 1–16 seeds) and were widely scattered around source plants (Figure 1). Mean maximum dispersal distance was 30.0 ± 16.8 m (range = 12.5–56.6 m). Five banana yucca seeds were found among three of the caches, indicating that rodents were foraging for and caching indigenous yucca seeds. We also found one larder containing 44 Joshua tree seeds in a dead, hollow, banana yucca stem.

We resurveyed the areas around source plants later in the fall and found that 70 caches were still intact (45.8%), four caches had most seeds removed but one or more seeds still present (2.6%), 79 caches were missing (51.6%), and 50 new caches had been made (Table I). Rodents made these new (secondary) caches from seeds they had taken from the original or primary caches. Secondary caches were smaller, containing an average of 3.3 ± 2.5 seeds (1–14 seeds; Table I), and usually were widely scattered (Figure 1). The transport distance between primary and secondary caches averaged 6–13 m (full range = 0.2–32.2 m) (Table II).

Combining both primary and secondary caches, rodents placed nearly equal numbers of caches under shrubs (32.0%), near the edge of shrubs (36.5%), and in the open (31.5%). Caches were usually close to or under blackbrush, ephedra, lycium (*Lycium andersonii*), or creosote bush, which are the most abundant shrubs at the study site. Rodents made caches in sandy or gravelly soil (68.2%) or under thin deposits of plant litter (31.8%).

During the course of the fall and winter, rodents ate many of the seeds in primary and secondary caches. On 23 March 2003, we found only 3 seedlings emerging from caches (the total number was probably greater because most seed had been removed, and rodents may have cached many seeds elsewhere).

From 392 trap-nights, we caught only 21 rodents (13 individuals captured). Trapping yielded two longtail pocket mice (*Chaetodipus formosus*), one canyon mouse (*Peromyscus crinitus*), and 10 Merriam's kangaroo rats (*Dipodomys merriami*). All but one of the Merriam's kangaroo rats were found on the burned site, and all other rodents were caught on the unburned site. The diurnal whitetail antelope ground squirrels (*Ammospermophilus leucurus*) were seen on the site but not captured.

The fruits of Joshua tree, banana yucca, and Mojave yucca that we monitored in fall 2003 had been gnawed on by rodents, both still attached to the plants and on the ground nearby. Rodents had dragged some pods away from productive plants, opened the pods, and appeared to have removed seeds. Based on teeth marks on the pods and foot prints in the soil, this activity appeared to be caused by antelope ground squirrels and perhaps woodrats (*Neotoma* sp.), but other species of rodents (*e.g.*, mice or kangaroo rats) may have removed seeds from the partially opened pods. Over a period of 2–3 months, rodents dismantled most of the pods, and most of the seeds in these pods disappeared. We did not observe any other means of seed removal.

A total of 178 Joshua tree seedlings (14.8%) emerged in the germination exclosure study. More seedlings emerged under shrubs (22.8 ± 19.4%) than in the open (8.8 ± 8.6%), but the difference is only marginally significant ($F_{1,20}$ = 3.658, P < 0.070). More seedlings emerged in

TABLE I. Results of radioactive seed studies around five source plants. Data are means ± 1 SD. Primary caches are from seeds taken from beneath source plants; secondary caches are from seeds taken from primary caches. Distance is the shortest distance between source plant and cache. Cache depth was measured to the top of seeds. Between 67.7 and 97.5% of the 200 seeds placed at each source plant were found

	Primary caches				Secondary caches			
Tree	n	Seeds·cache^{-1}	Distance (m)	Depth (mm)	n	Seeds·cache^{-1}	Distance (m)	Depth (mm)
1	26	6.4 ± 3.7	12.7 ± 7.1	17.0 ± 31.5	5	2.6 ± 1.5	15.9 ± 2.5	5.0 ± 0.7
2	38	3.6 ± 3.3	18.1 ± 13.1	19.5 ± 21.6	6	2.2 ± 0.8	11.1 ± 8.0	3.7 ± 1.6
3	27	7.2 ± 4.4	10.6 ± 6.6	11.8 ± 9.3	16	2.9 ± 1.7	9.0 ± 4.4	7.1 ± 7.9
4	39	2.8 ± 0.9	15.3 ± 9.5	3.1 ± 5.0	0	-	-	-
5	23	8.3 ± 3.8	8.6 ± 2.8	7.0 ± 7.5	23	3.9 ± 3.3	13.7 ± 7.5	8.5 ± 13.9

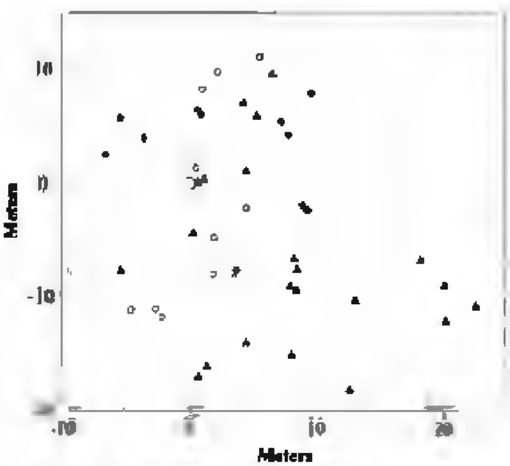

FIGURE 1. The dispersion of caches around Joshua tree 5 (large circle). Closed circles represent primary caches, open circles represent empty primary caches; triangles represent secondary caches made from the emptied primary caches. Units are in metres.

TABLE II. The distance that rodents moved seeds between primary and secondary caches at four source plants. There are more 1°-to-2° cache distances than secondary caches (Table I) because rodents often buried seeds from two primary caches in the same secondary cache.

Source plant	n	Mean ± SD	Range
1	7	13.2 ± 8.4	0.9–23.0
2	7	11.0 ± 12.1	0.2–32.2
3	18	6.0 ± 2.8	1.6–12.3
4	0		
5	27	12.0 ± 6.9	3.6–24.9

unwatered exclosures (16.6 ± 18.2%) than in watered exclosures (13.2 ± 13.8%), but the difference was not significant ($F_{1,20} = 0.303$, $P = 0.585$). The shrub–water interaction also was not significant ($F_{1,20} = 0.017$, $P = 0.897$). Over a period of two months, 82% of the 600 seeds in unfenced plots were removed by animals. There was no significant effect of watering ($F_{1,8} = 0.17$, $P = 0.6888$) on the number of seeds removed, but animals took significantly more seeds under shrub canopies (16.2 ± 15.9) than in the open (1.5 ± 1.8) ($F_{1,8} = 5.49$, $P = 0.0472$).

Discussion

Our source plant studies demonstrated that rodents will gather and cache Joshua tree seeds. Since we saw essentially the same pattern at all five source plants, scatter hoarding of these yucca seeds appears to be a consistent behavioural response. However, we are not certain which rodent species were responsible for the caches. At source plant 2, several primary caches were pilfered during the day while we were excavating and mapping primary caches, suggesting that white-tailed antelope ground squirrels, the only diurnal rodent at the site, were probably involved. At source plant 4, caches were uniformly small (1–4 seeds) and shallow (≈3 mm deep), suggesting that a small rodent such as a *Peromyscus* might have cached the seeds. Deer mice and pinyon mice make caches similar to the caches at source plant 4 (Vander Wall *et al.*, 2001; Hollander & Vander Wall, 2004). Caches made at source plants 1, 3, and 5 were larger and deeper and might have been made by a larger rodent such as *Dipodomys merriami*.

Only three of the Joshua tree seeds remained in caches until they germinated in the spring and established seedlings. This is only a tiny percentage (0.36%) of the 836 seeds found in primary caches and 0.66% of the 456 seeds still in place at our last autumn survey (mid October to mid November). However, this study took place in the middle of an extensive drought period in which conditions for germination were far from optimal. Labeled seeds were removed after the last cache survey, and thus it was not possible to quantify germination in subsequent years, which may have been closer to optimal conditions.

The germination study inside exclosures, which prevented pilferage by rodents, indicates that seeds in shallow surface caches similar to those made by rodents can germinate and establish seedlings. In the laboratory, germination rates of moistened Joshua tree seeds are typically 90–95% over ≈5 d (Garnett & Esque, unpubl. data). The much lower percentage in the field (14.8%) indicates that many of the viable seeds did not germinate successfully during trials. More seeds successfully germinated and produced seedlings under shrubs, but the variation was so great that this trend was not statistically significant. Because we controlled for the loss of seeds by rodents, we assume that abiotic factors, soil invertebrates, or microbes reduced germination.

The pilfering of seeds from caches and the moving of pilfered seeds to secondary caches suggest that the dynamics of scatter-hoarded seeds is similar to that observed with some pine seeds (Vander Wall & Joyner, 1998; Vander Wall, 2002). Most yucca seeds reside at several different sites before finally being consumed or germinating, and removal of seeds from cache sites is not an accurate measure of seed predation. The recaching of seeds has important consequences for seed dispersal; as seeds are dispersed from original caches they may be moved further from the original source plant, split into additional caches, and moved to a greater variety of habitats (Hoshizaki, Suzuki & Sasaki, 1997; Bohning-Gaese, Gaese & Rabemanantsoa, 1999; Roth & Vander Wall, 2005). Because many seeds were moved from their original cache site and probably recached, our estimate of establishment from natural caches is likely to be an underestimate of true seedling establishment. We were unable to conduct extensive, repeated surveys of the areas around source plants, so many secondary (and tertiary) caches were probably missed, and the fate of these seeds is unknown. Also, some of the recached seeds were probably carried farther from the source plant, suggesting that maximum seed dispersal distances were probably greater than the mean maximum distance of 30 m recorded for primary caches.

Rodents dismantled mature pods of Joshua tree and other yucca species and removed seeds. Most of the seeds appeared to have been carried away intact, because there was little evidence that rodents were eating seeds near the

CALIFORNIA
Water Boards
STATE WATER RESOURCES CONTROL BOARD
REGIONAL WATER QUALITY CONTROL BOARDS

State Policy for Water Quality Control: State Wetland Definition and Procedures for Discharges of Dredged or Fill Material to Waters of the State

[Also for Inclusion in the Water Quality Control Plan for Ocean Waters of California, and the Water Quality Control Plan for Inland Surface Waters and Enclosed Bays and Estuaries for Waters of the United States]

STATE WATER RESOURCES CONTROL BOARD

Adopted April 2, 2019
and Revised April 6, 2021

Procedures for Discharges of Dredged or Fill Material to Waters of the State

I. Introduction[1]

The mission of the State Water Resources Control Board and the Regional Water Quality Control Boards (Water Boards) includes the preservation, enhancement, and restoration of the quality of California's water resources for the protection of the environment and all beneficial uses for the benefit of present and future generations. In accordance with the Porter-Cologne Water Quality Control Act (Water Code, § 13000 et seq.), the Water Boards are authorized to regulate discharges of waste, which includes discharges of dredged or fill material, that may affect the quality of waters of the state. As described below, waters of the state include some, but not all, features that are defined as wetlands, as well as other features, including the ocean, lakes, and rivers. These wetlands provide environmental and economic benefits to the people of this state, including flood and stormwater control, surface and ground water supply, fish and wildlife habitat, erosion control, pollution treatment, nutrient cycling, and public enjoyment. Wetlands ameliorate the effects of global climate change by providing floodwater storage, sequestering carbon, and maintaining vulnerable plant and animal communities. Many of these invaluable areas statewide have been lost to fill and development. Presently, wetlands are threatened by impacts from increasing population growth, land development, sea level rise, and climate change. These Procedures for the Discharges of Dredged or Fill Material to Waters of the State (Procedures) conform to Executive Order W-59-93, commonly referred to as California's "no net loss" policy for wetlands. In accordance with Executive Order W-59-93, the Procedures ensure that the Water Boards' regulation of dredge or fill activities will be conducted in a manner "to ensure no overall net loss and long-term net gain in the quantity, quality, and permanence of wetlands acreage and values..." The Water Boards are committed to increasing the quantity, quality, and diversity of wetlands that qualify as waters of the state.

These Procedures contain a wetland definition in section II and wetland delineation procedures in section III, both of which apply to all Water Board programs. The wetland definition encompasses the full range of wetland types commonly recognized in California, including some features not protected under federal law, and reflects current scientific understanding of the formation and functioning of wetlands. These Procedures also include procedures for the submission, review and approval of applications for activities that could result in the discharge of dredged or fill material to any waters of the state in section IV. The Procedures include elements of the Clean Water Act Section 404(b)(1) Guidelines, thereby bringing uniformity to Water Boards' regulation of discharges of dredged or fill material to all waters of the state. The effective date of these Procedures shall be May 28, 2020.

II. Wetland Definition

The Water Boards define an area as wetland as follows:

An area is wetland if, under normal circumstances, (1) the area has continuous or recurrent saturation of the upper substrate caused by groundwater, or shallow surface water, or both; (2) the duration of such saturation is sufficient to cause anaerobic conditions in the upper substrate; and (3) the area's vegetation is dominated by hydrophytes or the area lacks vegetation.

[1] [NOTE. These Procedures will be incorporated into the Water Quality Control Plans for (1) Inland Surface Waters Enclosed Bays and Estuaries for waters for which water quality standards are required by the Federal Water Pollution Control Act and acts amendatory thereof or supplementary thereto and (2) Ocean Waters of California. Future incorporation of the Procedures, as adopted, into the water quality control plans will be considered non-substantive amendments. At that time, formatting and other organizational edits necessary for incorporation into the water quality control plans will be addressed.]

The Water Code defines "waters of the state" broadly to include "any surface water or groundwater, including saline waters, within the boundaries of the state." "Waters of the state" includes all "waters of the U.S."[2] The following wetlands are waters of the state:

1. Natural wetlands,
2. Wetlands created by modification of a surface water of the state,[3] and
3. Artificial wetlands[4] that meet any of the following criteria:
   a. Approved by an agency as compensatory mitigation for impacts to other waters of the state, except where the approving agency explicitly identifies the mitigation as being of limited duration;
   b. Specifically identified in a water quality control plan as a wetland or other water of the state;
   c. Resulted from historic human activity, is not subject to ongoing operation and maintenance, and has become a relatively parmanent part of the natural landscape; or
   d. Greater than or equal to one acre in size, unless the artificial wetland was constructed, and is currently used and maintained, primarily for one or more of the following purposes (i.e., the following artificial wetlands are not waters of the state unless they also satisfy the criteria set forth in 2, 3a, or 3b):
      i. Industrial or municipal wastewater treatment or disposal,
      ii. Settling of sediment,
      iii. Detention, retention, infiltration, or treatment of stormwater runoff and other pollutants or runoff subject to regulation under a municipal, construction, or industrial stormwater permitting program,
      iv. Treatment of surface waters,
      v. Agricultural crop irrigation or stock watering,
      vi. Fire suppression,
      vii. Industrial processing or cooling,
      viii. Active surface mining – even if the site is managed for interim wetlands functions and values,

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[2] Therefore, wetlands that meet the current definition, or any historic definition, of waters of the U.S. are waters of the state. In 2000 the State Water Resources Control Board determined that all waters of the U.S. are also waters of the state by regulation, prior to any regulatory or judicial limitations on the federal definition of waters of the U.S. (California Code or Regulations title 23, section 3831(w).) This regulation has remained in effect despite subsequent changes to the federal definition. Therefore, waters of the state includes features that have been determined by the U.S. Environmental Protection Agency (U.S. EPA) or the U.S. Army Corps of Engineers (Corps) to be "waters of the U.S." in an approved jurisdictional determination; "waters of the U.S." identified in an aquatic resource report verified by the Corps upon which a permitting decision was based; and features that are consistent with any current or historic final judicial interpretation of "waters of the U.S." or any current or historic federal regulation defining "waters of the U.S." under the federal Clean Water Act.

[3] "Created by modification of a surface water of the state" means that the wetland that is being evaluated was created by modifying an area that was a surface water of the state at the time of such modification. It does not include a wetland that is created in a location where a water of the state had existed historically, but had already been completely eliminated at some time prior to the creation of the wetland. The wetland being evaluated does not become a water of the state due solely to a diversion of water from a different water of the state.

[4] Artificial wetlands are wetlands that result from human activity.

ix. Log storage,

x. Treatment, storage, or distribution of recycled water, or

xi. Maximizing groundwater recharge (this does not include wetlands that have incidental groundwater recharge benefits); or

xii. Fields flooded for rice growing. [5]

All artificial wetlands that are less than an acre in size and do not satisfy the criteria set forth in 2, 3.a, 3.b, or 3.c are not waters of the state. If an aquatic feature meets the wetland definition, the burden is on the applicant to demonstrate that the wetland is not a water of the state.

III. Wetland Delineation

The permitting authority shall rely on any wetland area delineation from a final aquatic resource report verified by the U.S. Army Corps of Engineers (Corps) for the purposes of determining the extent of wetland waters of the U.S. A delineation of any wetland areas potentially impacted by the project that are not delineated in a final aquatic resource report verified by the Corps shall be performed using the methods described in the three federal documents listed below (collectively referred to as "1987 Manual and Supplements") to determine whether the area meets the state definition of a wetland as defined above. As described in the 1987 Manual and Supplements, an area "lacks vegetation" if it has less than 5 percent areal coverage of plants at the peak of the growing season. The methods shall be modified only to allow for the fact that the lack of vegetation does not preclude the determination of such an area that meets the definition of wetland. Terms as defined in these Procedures shall be used if there is conflict with terms in the 1987 Manual and Supplements.

- Environmental Laboratory. 1987. U.S. Army Corps of Engineers Wetlands Delineation Manual. Technical Report Y-87-1. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.
- U.S. Army Corps of Engineers. 2008. Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Arid West Region (Version 2.0). ed. J. S. Wakeley, R. W. Lichvar, and C. V. Noble. ERDC/EL TR-08-28. Vicksburg, MS: U.S. Army Engineer Research and Development Center.
- U.S. Army Corps of Engineers. 2010. Regional Supplement to the Corps of Engineers Wetland Delineation Manual: Western Mountains, Valleys, and Coast Region (Version 2.0). ed. J. S. Wakeley, R. W. Lichvar, and C. V. Noble. ERDC/EL TR-10-3. Vicksburg, MS: U.S. Army Engineer Research and Development Center.

[5] Fields used for the cultivation of rice (including wild rice) that have not been abandoned due to five consecutive years of non-use for the cultivation of rice (including wild rice) that are determined to be a water of the state in accordance with these Procedures shall not have beneficial use designations applied to them through the Water Quality Control Plan for the Sacramento and San Joaquin River Basins, except as otherwise required by federal law for fields that are considered to be waters of the United States. Further, agricultural inputs legally applied to fields used for the cultivation of rice (including wild rice) shall not constitute a discharge of waste to a water of the state. Agricultural inputs that migrate to a surface water or groundwater may be considered a discharge of waste and are subject to waste discharge requirements or waivers of such requirements pursuant to the Water Board's authority to issue or waive waste discharge requirements or take other actions as applicable.

IV. Procedures for Regulation of Discharges of Dredged or Fill Material to Waters of the State

The purpose of this section is to establish application procedures for discharges of dredged or fill material to waters of the state, which includes both waters of the U.S. and non-federal waters of the state. This section supplements existing state requirements for discharges of dredged or fill material to waters of the U.S.[6] These Procedures include Appendix A, which contains relevant portions of the U.S. EPA's Section 404(b)(1) "Guidelines for Specification of Disposal Sites for Dredge or Fill Material"[7] (Guidelines), with minor modifications to make them applicable to the state dredged or fill program (hereafter State Supplemental Dredge or Fill Guidelines).[8] This section applies to all applications for discharges of dredged or fill material to waters of the state submitted after [insert date that is nine months after approval by the Office of Administrative Law].[9] The Procedures do not apply to applications that are submitted prior to [insert date that is nine months after approval by the Office of Administrative Law].

Unless excluded by section IV.D, applicants must file an application with the Water Boards for any activity that could result in the discharge of dredged or fill material to waters of the state in accordance with California Code of Regulations, title 23, section 3855.[10] This application requirement applies to new discharges, proposed material changes in the character, location, or volume of existing discharges, and upon renewal of existing Orders for existing discharges. The permitting authority may amend an existing Order solely for the purpose of extending the expiration date without requiring a new application.

The applicant may consult with the Water Boards to determine whether a project could result in a discharge of dredged or fill material to waters of the state and/or discuss submittals that would meet the application requirements listed below. Discharges of dredged or fill material or other waste material to

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[6] California Code of Regulations, title 23, sections 3830-3869 (state's Clean Water Act (CWA) section 401 (33 USC § 1341) water quality certification program)

[7] 40 C.F.R. § 230.

[8] The State Supplemental Dredge or Fill Guidelines are included as Appendix A. Because the State Supplemental Dredge or Fill Guidelines are derived directly from the U.S. EPA's 404(b)(1) Guidelines, it uses slightly different terms than terms used in sections I through V of these Procedures. The State Supplemental Dredge or Fill Guidelines will be applied in a manner consistent with sections I through V of these Procedures.

[9] In cases where the applicant is a state agency and is acting as the CEQA lead agency for one or more projects otherwise subject to this section, and that state agency is a party to an existing written agreement (e.g., memorandum of understanding) with the State Water Board that sets out alternative procedures and requirements regarding the submission, review, or approval of project applications, the permitting authority shall apply the terms and conditions of the agreement in lieu of the terms and conditions of this section. After adoption of these Procedures, the State Water Board may also enter into such written agreements after consideration at a public meeting; such an agreement may include, for example, early consultation regarding potential project applications, early identification and analysis of project alternatives and mitigation measures, and dispute resolution. Any written agreements, whether existing or entered into after the adoption of these Procedures, may be amended in writing at any time by joint agreement of the parties, and such amended agreements shall govern in lieu of the terms and conditions of this section. All other applicable laws, including requirements for public notice and comment, apply to the permitting authorities' approval of projects under such an agreement.

[10] Note that California Code of Regulations, title 23, section 3855 applies only to individual water quality certifications, but these Procedures extend the application of section 3855 to individual waste discharge requirements for discharges of dredged or fill material to waters of the state and waivers thereof.

areas that are not waters of the state, but that could affect the quality of waters of the state, may be addressed under other Water Board regulatory programs.

A. Project Application Submittal for Individual Orders

The requirements set forth in sections IV.A and IV.B apply only to individual orders. Applicants must submit the items listed in subsection 1 to the permitting authority. In addition, applicants shall consult with the permitting authority about the items listed in subsection 2. Within 30 days of receiving the items listed in subsection 1, the permitting authority may require the applicant to submit one or more of the items in subsection 2 for a complete application. Applicants are encouraged to consult with the permitting authority to determine the appropriate level of detail for the items in subsections 1 (and 2, if applicable). Within 30 days of receiving all of the required items, the permitting authority shall determine whether the application is complete and notify the applicant accordingly. If the applicant's federal license or permit application includes any of the information required in subsections 1 or 2 below, the applicant may submit the federal application materials to satisfy the corresponding state application information. If federal application materials are submitted as part of the state application, the applicant shall indicate where the corresponding state application information can be found in the federal application materials.

1. Items Required for a Complete Application

   a. All items listed in California Code of Regulations, title 23, section 3856 "Contents of a Complete Application."[11]

   b. If the Corps requires an aquatic resource delineation report, a copy of the report verified by the Corps.

   c. A delineation of any waters that are not delineated in an aquatic resource delineation report verified by the Corps. If such waters include wetlands, the wetlands must be delineated as described in section III.

   d. The dates upon which the overall project activity will begin and end, and, if known, the date(s) upon which the discharge(s) will take place.

   e. Map(s) with a scale of at least 1:24000 (1" = 2000') and of sufficient detail to accurately show (1) the boundaries of the lands owned or to be utilized by the applicant in carrying out the proposed activity, including the grading limits, proposed land uses, and the location, dimensions and type of any structures erected (if known) or to be erected and (2) all aquatic resources that may qualify as waters of the state, within the boundaries of the project, and all aquatic resources that may qualify as waters of the state outside of the boundary of the project that could be impacted by the project. A map verified by the Corps may satisfy this requirement if it includes all potential waters of the state. The permitting authority may require that the map(s) be submitted in electronic format (e.g., GIS shapefiles).

   f. A description of the waters proposed to be impacted by the dredge or fill activity. The description should include the beneficial uses as listed in the applicable water quality control plan; a description of the activity at each individual discharge or dredge location; quantity of impacts to waters proposed to receive a discharge of dredged or fill material at each location rounded to at least the nearest one-hundredth (0.01) of an acre, nearest linear foot, and quantity

[11] Note that California Code of Regulations, title 23, section 3856 applies only to individual water quality certifications, but these Procedures extend the application of section 3856 to individual waste discharge requirements for discharges of dredged or fill material to waters of the state and waivers thereof.

of impacts to waters proposed to be dredged at each dredging location to the nearest cubic yard (as applicable); assessment of potential direct and indirect impacts resulting from the discharge or dredging activity and potential mitigation measures for those potential impacts, identification of existing water quality impairment(s); the source of water quality impairment(s), if known; and the presence of rare, threatened or endangered species[12] habitat.

g. An alternatives analysis,[13] unless any of the following exemptions apply. The exemption from the alternatives analysis requirement does not preclude a permitting authority from requiring the applicant to demonstrate in its application that the project complies with section IV.B.1.a.

   i. The project includes discharges to waters of the state outside of federal jurisdiction, but the entire project would meet the terms and conditions of one or more Water Board-certified Corps' General Permits, including any Corps District's regional terms and conditions, if all discharges were to waters of the U.S. The permitting authority will verify that the entire project would meet the terms and conditions of the Corps' General Permit(s) if all discharges, including discharges to waters of the state outside of federal jurisdiction, were to waters of the U.S. based on information supplied by the applicant.

   ii. The project includes only discharges to waters of the U.S. and meets the terms and conditions for coverage under an uncertified Corps' General Permit, including any Corps District's regional terms and conditions. This exemption does not apply if the discharge of dredged or fill material will directly impact:

      a) more than two-tenths (0.2) of an acre or 300 linear feet of waters of the state;

      b) rare, threatened, or endangered species habitat in waters of the state;

      c) wetlands or eel grass beds; or

      d) Outstanding National Resource Waters or Areas of Special Biological Significance.

   iii. The project would be conducted in accordance with a watershed plan that has been approved for use by the permitting authority and analyzed in an environmental document that includes a sufficient alternatives analysis, monitoring provisions, and guidance on compensatory mitigation opportunities.

   iv. The project is an Ecological Restoration and Enhancement Project.

   v. The project has no permanent impacts to aquatic resources and no impacts to rare, threatened or endangered species habitat in waters of the state, wetlands or eel grass beds, Outstanding National Resource Waters or Areas of Special Biological Significance,

[12] "Rare, threatened, or endangered species" as used in the Procedures refers to plant and animal species listed as rare, threatened, or endangered pursuant to the California Endangered Species Act of 1984 (Fish & Game Code, § 2050 et seq.), the Native Plant Protection Act of 1977 (Fish & Game Code, § 1900 et seq.), or the Federal Endangered Species Act of 1973 (16 U.S.C. § 1531 et seq.).

[13] "Alternatives analysis" as used in these Procedures refer to the analysis required by section IV.A.1.h and is a means to comply with the State Supplemental Dredge or Fill Guidelines, section 230.10(a). An alternatives analysis also may be required in order to comply with other statutory or regulatory requirements, such as CEQA or a Regional Board water quality control plan discharge prohibition. The exemptions and the tiers set forth below do not affect any alternatives analysis conducted pursuant to another statutory or regulatory requirement. To the extent that the permitting authority is acting as the lead agency under CEQA, it may be necessary for the permitting authority to conduct further analysis to comply with CEQA.

and all implementation actions in the restoration plan can reasonably be concluded within one year.

h. If none of the above exemptions apply, the applicant must submit an alternatives analysis consistent with the requirements of section 230.10 of the State Supplemental Dredge or Fill Guidelines that allows the permitting authority to determine whether the proposed project is the Least Environmentally Damaging Practicable Alternative (LEDPA). If the applicant submitted information to the Corps to support an alternatives analysis, the applicant shall provide that information to the permitting authority. Such information may satisfy some or all of the following requirements in accordance with section IV.B.3. Alternatives analyses shall be completed in accordance with the following tiers. The level of effort required for an alternatives analysis within each of the three tiers shall be commensurate with the significance of the impacts resulting from the discharge.[14]

   i. Tier 3 projects include any discharge of dredged or fill material that directly impacts more than two-tenths (0.2) of an acre or 300 linear feet of waters of the state, rare, threatened or endangered species habitat in waters of the state, wetlands or eel grass beds, or Outstanding National Resource Waters or Areas of Special Biological Significance, and is not a project that inherently cannot be located at an alternate location. Tier 3 projects shall provide an analysis of off-site and on-site alternatives.

   ii. Tier 2 projects include any discharge of dredged or fill material that directly impacts more than one tenth (0.1) and less than or equal to two tenths (0.2) of an acre or more than 100 and less than or equal to 300 linear feet of waters of the state unless it meets the criteria for a Tier 3 project, or any project that inherently cannot be located at an alternate location (unless it meets the size requirements set forth in Tier 1). Tier 2 projects shall provide an analysis of only on-site alternatives. For routine operation and maintenance of existing facilities, analysis of on-site alternatives is limited to operation and maintenance alternatives for the facility.

   iii. Tier 1 projects include any discharge of dredged or fill material that directly impacts less than or equal to one tenth (0.1) of an acre or less than or equal to 100 linear feet of waters of the state, unless it meets the criteria for a Tier 3 project. Tier 1 projects shall provide a description of any steps that have been or will be taken to avoid and minimize loss of, or significant adverse impacts to, beneficial uses of waters of the state.

2. Additional Information Required for a Complete Application

   a. If required by the permitting authority on a case-by-case basis, supplemental field data from the wet season to substantiate dry season delineations, as is consistent with the 1987 Manual and Supplements.

   b. If compensatory mitigation is required by the permitting authority, on a case-by-case basis, a draft compensatory mitigation plan developed using a watershed approach containing the items listed below. Compensatory mitigation plans are not required for Ecological Restoration and Enhancement Projects. For permittees who intend to fulfill their compensatory mitigation

[14] As used below, "impacts" include both permanent and temporary impacts.

obligations by securing credits from approved mitigation banks or in-lieu fee programs, their mitigation plans need include only items i, ii, and iii, as described below, as well as information required in the State Supplemental Dredge or Fill Guidelines, section 230.94 (c)(5) and (c)(6), and the name of the specific mitigation bank or in-lieu fee program proposed to be used.

Draft compensatory mitigation plans shall comport with the State Supplemental Dredge or Fill Guidelines, Subpart J, and include the items listed below.

i. A watershed profile for the project evaluation area for both the proposed dredged or fill project and the proposed compensatory mitigation project.

ii. An assessment of the overall condition of aquatic resources proposed to be impacted by the project and their likely stressors, using an assessment method approved by the permitting authority.

iii. A description of how the project impacts and compensatory mitigation would not cause a net loss of the overall abundance, diversity, and condition of aquatic resources, based on the watershed profile. If the compensatory mitigation is located in the same watershed as the project, no net loss will be determined on a watershed basis. If the compensatory mitigation and project impacts are located in multiple watersheds, no net loss will be determined considering all affected watersheds collectively. The level of detail in the plan shall be sufficient to accurately evaluate whether compensatory mitigation offsets the adverse impacts attributed to a project.

iv. Preliminary information about ecological performance standards, monitoring, and long-term protection and management, as described in the State Supplemental Dredge or Fill Guidelines.

v. A timetable for implementing the compensatory mitigation plan.

vi. If the compensatory mitigation plan includes buffers, design criteria and monitoring requirements for those buffers.

vii. If the compensatory mitigation involves restoration or establishment as the form of mitigation, applicants shall notify, as applicable, state and federal land management agencies, airport land use commission, fire control districts, flood control districts, local mosquito-vector control district(s), and any other interested local entities prior to initial site selection. These entities should be notified as early as possible during the initial compensatory mitigation project design stage.

viii. If required by the permitting authority, an assessment of reasonably foreseeable impacts to the compensatory mitigation associated with climate change, and any measures to avoid or minimize those potential impacts.

c. If required by the permitting authority on a case-by-case basis, if project activities include in-water work or water diversions, a proposed water quality monitoring plan to monitor compliance with water quality objectives of the applicable water quality control plan. At a minimum, the plan should include type and frequency of sampling for each applicable parameter.

d. In all cases where temporary impacts are proposed, a draft restoration plan that outlines design, implementation, assessment, and maintenance for restoring areas of temporary impact to pre-project conditions. The design components shall include the objectives of the restoration plan; grading plan of disturbed areas to pre-project contours; a planting palette with plant species native to the area; seed collection locations; and an invasive species management plan. The implementation component shall include all proposed actions to implement the plan (e.g., re-contouring, initial planting, site stabilization, removal of temporary structures) and a schedule for completing those actions. The maintenance and assessment components shall include a description of performance standards used to evaluate attainment of objectives; the timeframe for determining attainment of performance standards; and maintenance requirements (e.g., watering, weeding, replanting and invasive species control). If temporary impacts are proposed to be restored through passive restoration, the draft restoration plan shall include an explanation of how passive restoration will restore the area to pre-project conditions, assessment components, and an estimated date for expected restoration. The level of detail in the restoration plan shall be sufficient to accurately evaluate whether the restoration addresses the adverse temporary impacts attributed to a project. The applicant shall submit a final restoration plan that describes the restoration of all temporarily disturbed areas to pre-project conditions, consistent with section IV.B.4.

For Ecological Restoration and Enhancement Projects, a restoration plan for temporary impacts provided as part of the binding stream or wetland enhancement or restoration agreement or wetland establishment agreement may satisfy this requirement.

e. For all Ecological Restoration and Enhancement Projects, a draft assessment plan including the following: project objectives; description of performance standards used to evaluate attainment of objectives; protocols for condition assessment; the timeframe and responsible party for performing condition assessment; and assessment schedule. A draft assessment plan shall provide for at least one assessment of the overall condition of aquatic resources and their likely stressors, using an appropriate assessment method approved by the permitting authority, prior to restoration and/or enhancement and two years following restoration and/or enhancement to determine success of the restoration and/or enhancement. An assessment plan approved by a federal or state resource agency, or a local agency with the primary function of managing land or water for wetland habitat purposes in accordance with a binding stream or wetland enhancement agreement, restoration agreement, or establishment agreement, will satisfy these requirements. An assessment plan approved by a non-governmental conservation organization or a state or federal agency that is statutorily tasked with natural resource management may satisfy some or all of these requirements.

B. Permitting Authority Review and Approval of Applications for Individual Orders

1. The permitting authority will evaluate the potential impacts on the aquatic environment from the proposed project and determine whether the proposed project complies with these Procedures. The permitting authority has the discretion to approve a project only if the applicant has demonstrated the following:

   a. A sequence of actions has been taken to first avoid, then to minimize, and lastly compensate for adverse impacts that cannot be practicably avoided or minimized to waters of the state;

   b. The potential impacts will not contribute to a net loss of the overall abundance, diversity, and condition of aquatic resources in a watershed (or multiple watersheds when compensatory mitigation is permitted in another watershed as set forth in section IV.B.5(d));

   c. The discharge of dredged or fill material will not violate water quality standards and will be consistent with all applicable water quality control plans and policies for water quality control; and

   d. The discharge of dredged or fill material will not cause or contribute to significant degradation of the waters of the state.

2. The permitting authority shall rely on any final aquatic resource report verified by the Corps to determine boundaries of waters of the U.S. For all other wetland area delineations, the permitting authority shall review and approve delineations that are performed using the methods described in section III.

3. Alternatives Analysis Review Requirements:

    a. The purpose of the alternatives analysis is to identify the LEDPA. The permitting authority will be responsible for determining the sufficiency of an alternatives analysis except as described in 3(b) below. In all cases, the alternatives analysis must establish that the proposed project alternative is the LEDPA in light of all potential direct, secondary (indirect), and cumulative impacts on the physical, chemical, and biological elements of the aquatic ecosystem.

    b. <u>Discharges to waters of the U.S.</u>

    In reviewing and approving the alternatives analysis for discharges of dredged or fill material that impact waters of the U.S., the permitting authority shall defer to the Corps' determinations on the adequacy of the alternatives analysis, or rely on a draft alternatives analysis if no final determination has been made, unless the Executive Officer or Executive Director determines that (1) the permitting authority was not provided an adequate opportunity to collaborate in the development of the alternatives analysis, (2) the alternatives analysis does not adequately address aquatic resource issues identified in writing by the Executive Officer or Executive Director to the Corps during the development of the alternatives analysis, or (3) the proposed project and all of the identified alternatives would not comply with water quality standards.

    If the project also includes discharges to waters of the state outside of federal jurisdiction, the permitting authority shall require the applicant to supplement the alternatives analysis to include waters of the state outside of federal jurisdiction unless the applicant has consulted with the permitting authority and the alternatives analysis addresses all issues identified by the permitting authority during the consultation process. If an alternatives analysis is not required by the Corps for discharges of dredged or fill material to waters of the U.S., the permitting authority shall require an alternatives analysis for the entire project in accordance with the State Supplemental Dredge or Fill Guidelines, unless the project is exempt under section IV.A.1(g) above.

    The permitting authority shall not apply the presumption set forth in the State Supplemental Dredge or Fill Guidelines, section 230.10(a)(3) to any non-vegetated waters of the U.S. that the Corps does not classify as a special aquatic site (as defined in subpart E of U.S. EPA's section 404(b)(1) Guidelines).

4. Prior to or concurrent with issuance of the Order, the permitting authority will approve the final restoration plan for temporary impacts. Generally, the permitting authority will approve the final restoration plan when it issues the Order. The permitting authority may approve the final restoration plan after it issues the Order. In such cases the permitting authority shall include as a condition of the Order that the applicant receive approval of the final restoration plan prior to initiating the temporary impacts and shall specify a process for approving the final restoration plan.

5. Compensatory Mitigation

    a. Compensatory mitigation, in accordance with the State Supplemental Dredge or Fill Guidelines, Subpart J, may be required to ensure that an activity complies with these Procedures. Consistent with section 230.93(a)(2) of the State Supplemental Guidelines, subject to the permitting authority's approval, compensatory mitigation may be performed using methods of restoration, enhancement, establishment, and in certain circumstances preservation. Restoration should generally be the first option considered because the likelihood of success is greater and the impacts to potentially ecologically important uplands are reduced compared to establishment, and the potential gains in terms of aquatic resource functions are greater, compared to enhancement and preservation.

b. Where feasible, the permitting authority will consult and coordinate with any other public agencies that have concurrent mitigation requirements in order to achieve multiple environmental benefits with a single mitigation project, thereby reducing the cost of compliance to the applicant.

c. Amount: The amount of compensatory mitigation will be determined on a project-by-project basis in accordance with State Supplemental Dredge or Fill Guidelines, section 230.93(f). The permitting authority may take into account recent anthropogenic degradation to the aquatic resource and the potential and existing functions and conditions of the aquatic resource. The permitting authority may reduce the amount of compensatory mitigation if buffer areas adjacent to the compensatory mitigation are also required to be maintained as part of the compensatory mitigation management plan. The amount of compensatory mitigation required by the permitting authority will vary depending on which of the following strategies the applicant uses to locate the mitigation site within a watershed.

Strategy 1: Applicant locates compensatory mitigation using a watershed approach based on a watershed profile developed from a watershed plan that: (1) has been approved for use by the permitting authority and analyzed in an environmental document, (2) includes monitoring provisions, and (3) includes guidance on compensatory mitigation opportunities.

Strategy 2: Applicant locates compensatory mitigation using a watershed approach based on a watershed profile developed for a project evaluation area, and demonstrates that the mitigation project will contribute to the sustainability of watershed functions and the overall health of the watershed area's aquatic resources.

Generally, the amount of compensatory mitigation required under Strategy 1 will be less than the amount of compensatory mitigation required under Strategy 2 since the level of certainty that a compensatory mitigation project will meet its performance standards increases if the compensatory mitigation project complies with a watershed plan as described above. Certainty increases when there is a corresponding increase in understanding of watershed conditions, which is increased when using a watershed plan as described above to determine compensatory mitigation requirements.

A minimum of one-to-one mitigation ratio,[15] measured as area or length, is required to compensate for wetland or stream losses when compensatory mitigation is required. Subject to the permitting authority's approval, the ratio may be satisfied using any of the methods identified in section IV.B.5(a). A higher overall mitigation ratio shall be used where necessary to ensure replacement of lost aquatic resource functions, as described in the State Supplemental Dredge or Fill Guidelines, section 230.93(f). Where temporary impacts will be restored to pre-project conditions, the permitting authority may require compensatory mitigation for temporal loss from the temporary impacts.

[15] For temporary impacts, the minimum one-to-one mitigation ratio for wetland or stream losses is not applicable for temporal losses for impacts that are fully restored to pre-project conditions.

d. Type and Location: The permitting authority will evaluate the applicant's proposed mitigation type and location based on the applicant's use of a watershed approach based on a watershed profile. The permitting authority will determine the appropriate type and location of compensatory mitigation based on watershed conditions, impact size, location and spacing, aquatic resource values, relevant watershed plans, and other considerations.

   In general, the required compensatory mitigation should be located within the same watershed as the impact site, but the permitting authority may approve compensatory mitigation in a different watershed. For example, if a proposed project may affect more than one watershed, then the permitting authority may determine that locating all required project mitigation in one area is ecologically preferable to requiring mitigation within each watershed.

e. Final Compensatory Mitigation Plan: The permitting authority will review and approve the final compensatory mitigation plan submitted by the applicant to ensure mitigation comports with the State Supplemental Dredge or Fill Guidelines, Water Code requirements, applicable water quality standards, and other appropriate requirements of state law. The level of detail in the final plan shall be sufficient to accurately evaluate whether compensatory mitigation offsets the adverse impacts attributed to a project considering the overall size and scope of impact. The compensatory mitigation plan shall be sufficient to provide the permitting authority with a reasonable assurance that replacement of the full range of lost aquatic resource(s) and/or functions will be provided in perpetuity.

   Generally, the permitting authority will approve the final compensatory mitigation plan when it issues the Order. Where compliant with CEQA, the permitting authority may approve the final compensatory mitigation plan after it issues the Order. In such cases the permitting authority shall include as a condition of the Order that the applicant receive approval of the final mitigation plan prior to discharging dredged or fill material to waters of the state and shall specify a process for approving the final mitigation plan.

f. Financial Security: Where deemed necessary by the permitting authority, provision of a financial security (e.g., letter of credit or performance bond) shall be a condition of the Order. In this case, the permitting authority will approve the financial security to ensure compliance with compensatory mitigation plan requirements. The financial security shall be in a form consistent with the California Constitution and state law.

g. Term of Mitigation Obligation: The permitting authority may specify in the Order the conditions that must be met in order for the permitting authority to release the permittee from the mitigation obligation, including compensatory mitigation performance standards and long-term management funding obligations.

6. The permitting authority shall provide public notice in accordance with Water Code section 13167.5 for waste discharge requirements. The permitting authority shall provide public notice of an application for water quality certification in accordance with California Code of Regulations, title 23, section 3858. If the permitting authority receives comments on the application or there is substantial public interest in the project, the permitting authority shall also provide public notice of the draft Order, or draft amendment of the Order, unless circumstances warrant otherwise.

7. The permitting authority will review and approve the final monitoring and reporting requirements for all projects. Monitoring and reporting may be required to demonstrate compliance with the terms of the Order.

C. General Orders

Discharges of dredged or fill material to waters of the state that are regulated under a general order are not subject to the requirements set forth in sections IV.A and IV.B. Applicants applying to enroll under a general order shall follow the instructions specified in the general order for obtaining coverage.

The permitting authority may issue general orders for specific classes of dredged or fill discharge activities that are similar; involve the same or similar types of discharges and possible adverse impacts requiring the same or similar conditions or limitations in order to alleviate potential adverse impacts to water quality; and are determined by the permitting authority to more appropriately be regulated under a general order rather than under an individual Order.

General orders shall be reviewed, noticed, and issued in accordance with the applicable requirements of division 7 of the Water Code and the California Code of Regulations, division 3 of title 23.

D. Activities and Areas Excluded from the Application Procedures for Regulation of Discharges of Dredged or Fill Material to Waters of the State

The application procedures specified in sections IV.A and IV.B do not apply to proposed discharges of dredged or fill material to waters of the state from the following activities[16] or to the following areas. These exclusions do not, however, affect the Water Board's authority to issue or waive waste discharge requirements (WDRs) or take other actions for the following activities or areas to the extent authorized by the Water Code.

1. Activities excluded from application procedures in sections IV.A and IV.B:

   a. Activities that are exempt under CWA section 404(f) (33 USC § 1344(f)).[17] The permitting authority shall use 33 CFR 323.4 (1986) and 40 CFR 232.3 (1988) to determine whether certain activities are exempt under CWA section 404(f). These regulations are hereby incorporated by reference and shall apply to all waters of the state. Consistent with CWA section 404(f)(2) and 40 CFR section 232.3, any discharge of dredged or fill material to a water of the state incidental to any of these activities is not exempt under CWA section 404(f) and shall be subject to the application procedures set forth in sections IV.A and IV.B, if (1) the purpose of the activity is bringing a water of the state into a use to which it was not previously subject, where the flow or circulation of water of the state may be impaired or the reach of such waters be reduced, or (2) the discharge contains any toxic pollutant listed in CWA section 307.

   b. Suction dredge mining activities for mineral recovery regulated under CWA section 402.

   c. Routine and emergency operation and maintenance activities conducted by public agencies, water utilities, or special districts that result in discharge of dredged or fill material to artificial, existing waters of the state:

      i. currently used and maintained primarily for one or more of the purposes listed in section II.3.d. (ii), (iii), (iv), (x), or (xi); or

      ii. for the purpose of preserving the line, grade, volumetric or flow capacity within the existing footprint of a flood control or stormwater conveyance facility.

   This exclusion does not relieve public agencies, water utilities or special districts of their

[16] Note that not all activities identified in this section necessarily result in discharges of dredged or fill material to waters of the state.

[17] Unless otherwise specified all federal statutes and regulations that are incorporated by reference into these Procedures are the versions of those federal statutes and regulations that are in effect as of April 2, 2019.

obligation to submit an application for a water quality certification consistent with California Code of Regulations, title 23, section 3856 or waste discharge requirements consistent with Water Code section 13260, whichever is applicable, to the permitting authority for these activities; or their responsibility to avoid and minimize adverse impacts to aquatic resources and beneficial uses from these activities. The permitting authority has full discretion to determine whether an activity described above qualifies for this exclusion based on the application submitted and other relevant information. If the permitting authority determines that an activity qualifies for this exclusion, the permitting authority retains full authority and discretion under the Porter-Cologne Water Quality Control Act to determine how to regulate the discharge of dredged or fill material. Where a permitting authority has already determined it appropriate to regulate these types of activities in specific instances, this exclusion in no way disturbs or limits the permitting authority's current regulation of these types of activities. This exclusion does not apply to the discharge of dredged or fill material to a water of the state approved by an agency as compensatory mitigation.

d. Routine operation and maintenance activities that result in discharge of dredged or fill material to artificially-created waters currently used and maintained primarily for one or more of the purposes listed in section II.3.d. (i), (ii), (iii), (vi), (vii), (x), or (xi). This exclusion does not apply to the discharge of dredged or fill material to (a) a water of the U.S., (b) a water specifically identified in a water quality control plan, (c) a water created by modification of a water of the state, or (d) a water approved by an agency as compensatory mitigation.

2. Areas excluded from application procedures in sections IV.A and IV.B:

a. Wetland areas that qualify as prior converted cropland (PCC) within the meaning of 33 CFR section 328.3(b)(2). The applicant may establish that the area is PCC by providing relevant documentary evidence that the area qualifies as PCC and has not been abandoned due to five consecutive years of non-use for agricultural purposes, or by providing a current PCC certification by the Natural Resources Conservation Service, the Corps, or the U.S. EPA to the permitting authority.

b. Wetlands that are, or have been, in rice cultivation (including wild rice) within the last five years as of April 2, 2019 and have not been abandoned due to five consecutive years of non-use in rice production.

c. The following features used for agricultural purposes:

i. Ditches with ephemeral flow that are not a relocated water of the state or excavated in a water of the state;

ii. Ditches with intermittent flow that are not a relocated water of the state or excavated in a water of the state, or that do not drain wetlands other than any wetlands described in sections (iv) or (v);

iii. Ditches that do not flow, either directly or through another water, into another water of the state;

iv. Artificially irrigated areas that would revert to dry land should application of waters to that area cease; or

v. Artificial, constructed lakes and ponds created in dry land such as farm and stock watering ponds, irrigation ponds, and settling basins.

The exclusions in section IV.D.2 do not apply to discharges of dredged or fill material that convert wetland areas to a non-agricultural use.

For requests for approvals from the Division of Water Rights for activities associated with (1) an appropriation of water subject to Part 2 (commencing with section 1200) of Division 2 of the Water Code, (2) a hydroelectric facility where the proposed activity requires a Federal Energy Regulatory Commission (FERC) license or amendment to a FERC license, or (3) any other diversion of water for beneficial use where approval by the Division of Water Rights is required, the Division of Water Rights will inform the applicant whether the application procedures in sections IV.A and IV.B will apply to the application.

V. Definitions

The following definitions apply to these Procedures, including the State Supplemental Dredge or Fill Guidelines. Unless otherwise indicated, any term that is not defined in these Procedures shall have the same meaning as defined in Water Code section 13050, and title 23, section 3831 of the California Code of Regulations.

Abundance means an estimate of the amount of aquatic resources by type in a watershed area, and what types of aquatic resources are most and least prevalent.

Active Surface Mining means operations that, in accordance with Division 2, Chapter 9 of the Surface Mining and Reclamation Act of 1975, have an approved reclamation plan, and for which reclamation has not been certified as complete by the local lead agency with the concurrence of the Department of Conservation.

Alternatives Analysis is the process of analyzing project alternatives, including the proposed project, to determine the alternative that is the least environmentally damaging practicable alternative (LEDPA).

Application means a written request, including a report of waste discharge or request for water quality certification, for authorization of any activity that may result in the discharge of dredged or fill material and is subject to these Procedures.

Discharge of Dredged or Fill Material shall have the same meanings as they are used in the federal Clean Water Act and 40 CFR section 232.2, but (1) shall include discharges to waters of the state that are not waters of the U.S. and (2) any demonstrations described in 40 CFR section 232.2(3)(i) shall be made to the permitting authority instead of the Corps or U.S. EPA. Placement of dredged or fill material in a manner that could not affect the quality of waters of the state is not considered a discharge of dredged or fill material.

Diversity means the relative proportion of aquatic resource types, classification, connectivity, and spatial distribution in a watershed area.

Ecological Restoration and Enhancement Project means the project is voluntarily undertaken for the purpose of assisting or controlling the recovery of an aquatic ecosystem that has been degraded, damaged or destroyed to restore some measure of its natural condition and to enhance the beneficial uses, including potential beneficial uses of water.

Such projects are undertaken:

1) in accordance with the terms and conditions of a binding stream or wetland enhancement or restoration agreement, or a wetland establishment agreement, between the real property interest owner or the entity conducting the habitat restoration or enhancement work and:

    a. a federal or state resource agency, including, but not limited to, the U.S. Fish and Wildlife Service, Natural Resources Conservation Service, Farm Service Agency, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, U.S. Forest Service, U.S. Bureau of Land Management, California Department of Fish and Wildlife, California Wildlife Conservation Board, California Coastal Conservancy or the Delta Conservancy;

    b. a local agency with the primary function of managing land or water for wetland habitat purposes; or

    c. a non-governmental conservation organization; or

2) by a state or federal agency that is statutorily tasked with natural resource management.

These projects do not include the conversion of a stream or natural wetland to uplands or stream channelization. It is recognized that Ecological Restoration and Enhancement Projects may require ongoing maintenance or management to maximize fish, wildlife, habitat, or other ecological benefits, or filling gullied stream channels and similar rehabilitative activities to re-establish stream and meadow hydrology. Changes in wetland plant communities that occur when wetland hydrology is more fully restored during rehabilitation activities are not considered a conversion to another aquatic habitat type. These projects also do not include actions required under a Water Board Order for mitigation, actions to service required mitigation, or actions undertaken for the primary purpose of land development.

Environmental Document means a document prepared for compliance with the California Environmental Quality Act (CEQA) or the National Environmental Policy Act (NEPA).

Hydrophyte means any macrophyte that grows in water or on a substrate that is at least periodically deficient in oxygen as a result of excessive water content; plants typically found in wet habitats.

LEDPA means the least environmentally damaging practicable alternative. The determination of practicable alternatives shall be consistent with the State Supplemental Dredge or Fill Guidelines, section 230.10(a).

Normal Circumstances is the soil and hydrologic conditions that are normally present, without regard to whether the vegetation has been removed. The determination of whether normal circumstances exist in a disturbed area involves an evaluation of the extent and relative permanence of the physical alteration of wetland hydrology and hydrophytic vegetation, and consideration of the purpose and cause of the physical alterations to hydrology and vegetation.

Order means waste discharge requirements, waivers of waste discharge requirements, or water quality certification.

Permitting Authority means the entity or person issuing the Order (i.e., the applicable Water Board, Executive Director or Executive Officer, or his or her designee).

Project means the whole of an action that includes a discharge of dredged or fill material to waters of the state.

Project Evaluation Area means an area that includes the project impact site, and/or the compensatory mitigation site, and is sufficiently large to evaluate the effects of the project and/or the compensatory mitigation on the abundance, diversity, and condition of aquatic resources in an ecologically meaningful unit of the watershed. The size and location of the ecologically meaningful unit shall be based on a reasonable rationale.

Water Boards mean any of the nine Regional Water Quality Control Boards, the State Water Resources Control Board, or all of them collectively.

Watershed means a land area that drains to a common waterway, such as a stream, lake, estuary, wetland, or ultimately the ocean.

Watershed Approach means an analytical process for evaluating the environmental effects of a proposed project and making decisions that support the sustainability or improvement of aquatic resources in a watershed. The watershed approach recognizes that the abundance, diversity, and condition of aquatic resources in a watershed support beneficial uses. Diversity of aquatic resources includes both the types of aquatic resources and the locations of those aquatic resources in a watershed. Consideration is also given to understanding historic and potential aquatic resource conditions, past and projected aquatic resource impacts in the watershed, and terrestrial connections between aquatic resources. The watershed approach can be used to evaluate avoidance and minimization of direct, secondary (indirect), and cumulative project impacts. It also can be used in determining compensatory mitigation requirements.

Watershed Plan means a document, or a set of documents, developed in consultation with relevant stakeholders, a specific goal of which is aquatic resource restoration, establishment, enhancement, and preservation within a watershed. A watershed plan addresses aquatic resource conditions in the watershed, multiple stakeholder interests, and land uses. Watershed plans should include information about implementing the watershed plan. Watershed plans may also identify priority sites for aquatic resource restoration and protection. Examples of watershed plans include special area management plans, advance identification programs, and wetland management plans. The permitting authority may approve the use of other plans, including for example, Habitat Conservation Plans (HCPs), Natural Community Conservation Plans (NCCPs), or municipal stormwater permit watershed management programs as watershed plans, if they substantially meet the stated above. Any NCCP approved by the California Department of Fish and Wildlife before December 31, 2020, and any regional HCP approved by the United States Fish and Wildlife Service before December 31, 2020, which includes biological goals for aquatic resources, shall be used by the permitting authority as a watershed plan for such aquatic resources, unless the permitting authority determines in writing that the HCP or NCCP does not substantially meet the definition of a watershed plan for such aquatic resources.

Watershed Profile means a compilation of data or information on the abundance, diversity, and condition of aquatic resources in a project evaluation area. The watershed profile shall include a map and a report characterizing the location, abundance and diversity of aquatic resources in the project evaluation area, assessing the condition of aquatic resources in the project evaluation area, and describing the environmental stress factors affecting that condition.

The watershed profile shall include information sufficient to evaluate direct, secondary (indirect), and cumulative impacts of project and factors that may favor or hinder the success of compensatory mitigation projects and help define watershed goals. It may include such things as current trends in habitat loss or conservation, cumulative impacts of past development activities, current development trends, the presence and need of sensitive species, and chronic environmental problems or site conditions such as flooding or poor water quality.

The scope and detail of the watershed profile shall be commensurate with the magnitude of impact associated with the proposed project. Information sources include online searches, maps, watershed plans, and possibly some fieldwork if necessary. In some cases, field data may need to be collected in the project evaluation area to confirm the reported condition. Some or all of the information may be obtained from a watershed plan. Watershed profiles for subsequent projects in a watershed can be used to track the cumulative effectiveness of the permitting authority's decisions.

Wetland Delineation means the application of a technical and procedural method to identify the boundary of a wetland area within a specified study site by identifying the presence or absence of wetland indicators at multiple points at the site and by establishing boundaries that group together sets of points that share the same status as wetland versus non-wetland.

Appendix A: State Supplemental Dredge or Fill Guidelines

It is the intent of the Water Boards to be consistent with the U.S. EPA's 404(b)(1) Guidelines where feasible. Due to jurisdictional and procedural differences, some modifications to the U.S. EPA's 404(b)(1) Guidelines were necessary. Generally, these changes or deletions were made to reduce redundancy (especially where sufficiently described elsewhere in these Procedures) and to account for other state requirements. Note that the numbering scheme of the U.S. EPA's 404(b)(1) Guidelines has been retained in these State Supplemental Dredge or Fill Guidelines for the benefit of practitioners who are familiar with the U.S. EPA's 404(b)(1) Guidelines. The State Supplemental Dredge or Fill Guidelines describe how the Water Boards will implement the U.S. EPA's 404(b)(1) Guidelines under these Procedures. The definitions contained herein apply to these Procedures, including the State Supplemental Dredge or Fill Guidelines.

Subpart A – General

§ 230.3 Definitions.

For purposes of these Procedures, the following terms shall have the meanings indicated:

(c) The terms aquatic environment and aquatic ecosystem mean waters of the state, including wetlands, that serve as habitat for interrelated and interacting communities and populations of plants and animals.

(h) The term discharge point means the point within the disposal site at which the dredged or fill material is released.

(i) The term disposal site means that portion of the "waters of the state" where the discharge of dredged or fill material is permitted and involves a bottom surface area and any overlying volume of water. In the case of wetlands or ephemeral streams on which surface water is not present, the disposal site consists of the wetland or ephemeral stream surface area.

(k) The term extraction site means the place from which the dredged or fill material proposed for discharge is to be removed.

(n) The term permitting authority means as defined above in the main text of these Procedures.

(q) The term practicable means available and capable of being done after taking into consideration cost, existing technology, and logistics in light of overall project purposes.

(q1) Special aquatic sites means those sites identified in subpart E. Special aquatic sites are geographic areas, large or small, possessing special ecological characteristics of productivity, habitat, wildlife protection, or other important and easily disrupted ecological values. These areas are generally recognized as significantly influencing or positively contributing to the general overall environmental health or vitality of the entire ecosystem of a region. (See § 230.10 (a)(3))

§ 230.6 Adaptability

(a) The manner in which these Guidelines are used depends on the physical, biological, and chemical nature of the proposed extraction site, the material to be discharged, and the candidate disposal site, including any other important components of the ecosystem being evaluated. Documentation to demonstrate knowledge about the extraction site, materials to be extracted, and the candidate disposal site is an essential component of guideline application. These Guidelines allow evaluation and documentation for a variety of activities, ranging from those with large, complex impacts on the aquatic environment to those for which the impact is likely to be innocuous.

It is unlikely that the Guidelines will apply in their entirety to any one activity, no matter how complex. It is anticipated that substantial numbers of applications will be for minor, routine activities that have little, if any, potential for significant degradation of the aquatic environment. It generally is not intended or expected that extensive testing, evaluation or analysis will be needed to make findings of compliance in such routine cases.

(b) The Guidelines user, including the agency or agencies responsible for implementing the Guidelines, must recognize the different levels of effort that should be associated with varying degrees of impact and require or prepare commensurate documentation. The level of documentation should reflect the significance and complexity of the discharge activity.

(c) An essential part of the evaluation process involves making determinations as to the relevance of any portion(s) of the Guidelines and conducting further evaluation only as needed. However, where portions of the Guidelines review procedure are "short form" evaluations, there still must be sufficient information (including consideration of both individual and cumulative impacts) to support the decision of whether to specify the site for disposal of dredged or fill material and to support the decision to curtail or abbreviate the evaluation process. The presumption against the discharge in § 230.10 applies to this decision-making.

Subpart B – Compliance with Guidelines

§ 230.10 Restrictions on Discharge

(a) No discharge of dredged or fill material shall be permitted if there is a practicable alternative to the proposed discharge which would have less adverse impact on the aquatic ecosystem, so long as the alternative does not have other significant adverse environmental consequences.

(1) For the purpose of this requirement, practicable alternatives include, but are not limited to:

(i) Activities which do not involve a discharge of dredged or fill material to waters of the state or ocean waters;

(ii) Discharges of dredged or fill material at other locations in waters of the state or ocean waters;

(2) An alternative is practicable if it is available and capable of being done after taking into consideration cost, existing technology, and logistics in light of overall project purposes. If it is otherwise a practicable alternative, an area not presently owned by the applicant which could reasonably be obtained, utilized, expanded or managed in order to fulfill the basic purpose of the proposed activity may be considered.

(3) Where activity associated with a discharge which is proposed for a special aquatic site (as defined in subpart E) does not require access or proximity to or siting within the special aquatic site in question to fulfill its basic purpose (i.e., is not "water dependent"), practicable alternatives that do not involve special aquatic sites are presumed to be available, unless clearly demonstrated otherwise. In addition, where a discharge is proposed for a special aquatic site, all practicable alternatives to the proposed discharge which do not involve a discharge into a special aquatic site are presumed to have less adverse impact on the aquatic ecosystem, unless clearly demonstrated otherwise.

(b) No discharge of dredged or fill material shall be permitted if it:

(1) Causes or contributes, after consideration of disposal site dilution and dispersion, to violations of any applicable State water quality standard;

(2) Violates any applicable toxic effluent standard or prohibition under section 307 of the Clean Water Act;

(c) No discharge of dredged or fill material shall be permitted which will cause or contribute to significant degradation of the waters of the state. Under these Guidelines, effects contributing to significant degradation considered individually or collectively, include:

(1) Significantly adverse effects of the discharge of pollutants on human health or welfare, including but not limited to effects on municipal water supplies, plankton, fish, shellfish, wildlife, and special aquatic sites;

(2) Significantly adverse effects of the discharge of pollutants on life stages of aquatic life and other wildlife dependent on aquatic ecosystems, including the transfer, concentration, and spread of pollutants or their byproducts outside of the disposal site through biological, physical, and chemical processes.

(3) Significantly adverse effects of the discharge of pollutants on aquatic ecosystem diversity, productivity, and stability. Such effects may include, but are not limited to, loss of fish and wildlife habitat or loss of the capacity of a wetland to assimilate nutrients, purify water, or reduce wave energy; or

(4) Significantly adverse effects of the discharge of pollutants on recreational, aesthetic, and economic values.

(d) No discharge of dredged or fill material shall be permitted unless appropriate and practicable steps have been taken which will minimize potential adverse impacts of the discharge on the aquatic ecosystem. Subpart H identifies such possible steps.

Subpart E –Special Aquatic Sites

§ 230.40 Sanctuaries and refuges

(a) Sanctuaries and refuges consist of areas designated under State and Federal laws or local ordinances to be managed principally for the preservation and use of fish and wildlife resources.

§ 230.41 Wetlands.

(a)(1) Wetlands are as defined above in the main text of these Procedures.

§ 230.42 Mud Flats.

(a) Mud flats are broad flat areas along the sea coast and in coastal rivers to the head of tidal influence and inland lakes, ponds, and riverine systems. When mud flats are inundated, wind and wave action may resuspend bottom sediments. Coastal mud flats are exposed at extremely low tides and inundated at high tides with the water table at or near the surface of the substrate. The substrate of mud flats contains organic material and particles smaller in size than sand. They are either unvegetated or vegetated only by algal mats.

§ 230.43 Vegetated shallows.

(a) Vegetated shallows are permanently inundated areas that under normal circumstances support communities of rooted aquatic vegetation, such as turtle grass and eel grass in estuarine or marine systems as well as a number of freshwater species in rivers and lakes.

§ 230.45 Riffle and Pool Complexes.

(a) Steep gradient sections of streams are sometimes characterized by riffle and pool complexes. Such stream sections are recognizable by their hydraulic characteristics. The rapid movement of water over a coarse substrate in riffles results in a rough flow, a turbulent surface, and high dissolved oxygen levels in the water. Pools are deeper areas associated with riffles. Pools are characterized by a slower stream velocity, a streaming flow, a smooth surface, and a finer substrate. Riffle and pool complexes are particularly valuable habitat for fish and wildlife.

Subpart H – Actions to Minimize Adverse Effects

Note: There are many actions which can be undertaken in response to 230.10(d) to minimize the adverse effects of discharges of dredged or fill material. Some of these, grouped by type of activity, are listed in this subpart. Additional criteria for compensation measures are provided in subpart J of these Procedures.

§ 230.70 Actions concerning the location of the discharge.

The effects of the discharge can be minimized by the choice of the disposal site. Some of the ways to accomplish this are by:

(a) Locating and confining the discharge to minimize smothering of organisms;

(b) Designing the discharge to avoid a disruption of periodic water inundation patterns;

(c) Selecting a disposal site that has been used previously for dredged material discharge;

(d) Selecting a disposal site at which the substrate is composed of material similar to that being discharged, such as discharging sand on sand or mud on mud;

(e) Selecting a disposal site, the discharge point, and the method of discharge to minimize the extent of any plume;

(f) Designing the discharge of dredged or fill material to minimize or prevent the creation of standing bodies of water in areas of normally fluctuating water levels, and minimize or prevent the drainage of areas subject to such fluctuations.

§ 230.71 Actions concerning the material to be discharged

The effects of a discharge can be minimized by treatment of, or limitations on the material itself, such as:

(a) Disposal of dredged material in such a manner that physiochemical conditions are maintained, and the potency and availability of pollutants are reduced.

(b) Limiting the solid, liquid, and gaseous components of material to be discharged at a particular site;

(c) Adding treatment substances to the discharge material;

(d) Utilizing chemical flocculants to enhance the deposition of suspended particulates in diked disposal areas.

§ 230.72 Actions controlling the material after discharge.

The effects of the dredged or fill material after discharge may be controlled by:

(a) Selecting discharge methods and disposal sites where the potential for erosion, slumping or leaching of materials into the surrounding aquatic ecosystem will be reduced. These sites or methods include, but are not limited to:

(1) Using containment levees, sediment basins, and cover crops to reduce erosions;

(2) Using lined containment areas to reduce leaching where leaching of chemical constituents from the discharged material is expected to be a problem;

(b) Capping in-place contaminated material with clean material or selectively discharging the most contaminated material first to be capped with the remaining material;

(c) Maintaining and containing discharged material properly to prevent point and nonpoint sources of pollution;

(d) Timing the discharge to minimize impact, for instance during periods of unusual high-water flows, wind, wave, and tidal actions.

§ 230.73 Actions affecting the method of dispersion.

The effects of a discharge can be minimized by the manner in which it is dispersed, such as:

(a) Where environmentally desirable, distributing the dredged material widely in a thin layer at the disposal site maintain natural substrate contours and elevation;

(b) Orienting a dredged or fill material mound to minimize undesirable obstruction to the water current or circulation pattern, and utilizing natural bottom contours to minimize the size of the mound;

(c) Using silt screens or other appropriate methods to confine suspended particulate/turbidity to a small area where settling or removal can occur;

(d) Making use of currents and circulation patterns to mix, disperse and dilute the discharge;

(e) Minimizing water column turbidity by using a submerged diffuser system. A similar effect can be accomplished by submerging pipeline discharges or otherwise releasing materials near the bottom;

(f) Selecting sites or managing discharges to confine and minimize the release of suspended particulates to give decreased turbidity levels and to maintain light penetration for organisms;

(g) Setting limitations on the amount of material to be discharged per unit of time or volume of receiving water.

§ 230.74 Actions related to technology.

Discharge technology should be adapted to the needs of each site. In determining whether the discharge operation sufficiently minimizes adverse environmental impacts, the applicant should consider:

(a) Using appropriate equipment or machinery, including protective devices, and the use of such equipment or machinery in activities related to the discharge of dredged or fill material;

(b) Employing appropriate maintenance and operation on equipment or machinery, including adequate training, staffing, and working procedures;

(c) Using machinery and techniques that are especially designed to reduce damage to wetlands. This may include machines equipped with devices that scatter rather than mound excavated materials, machines with specially designed wheels or tracks, and the use of mats under heavy machines to reduce wetland surface compaction and rutting;

(d) Designing access roads and channels spanning structures using culverts, open channels, and diversions that will pass both low and high-water flows, accommodate fluctuating water levels, and maintain circulation and faunal movement;

(e) Employing appropriate machinery and methods of transport of the material for discharge.

§ 230.75 Actions affecting plant and animal populations.

Minimization of adverse effects on populations of plant and animals can be achieved by:

(a) Avoiding changes in water current and circulation patterns which would interfere with the movement of animals;

(b) Selecting sites or managing discharges to prevent or avoid creating habitat conducive to the development of undesirable predators or species which have a competitive edge ecologically over indigenous plants or animals;

(c) Avoiding sites having unique habitat or other value, including habitat of threatened or endangered species;

(d) Using planning and construction practices to institute habitat development and restoration to produce a new or modified environmental state of higher ecological value by displacement of some or all of the existing environmental characteristics. Habitat development and restoration techniques can be used to minimize adverse impacts and to compensate for destroyed habitat. Additional criteria for compensation measures are provided in subpart J of this part. Use techniques that have been demonstrated to be effective in circumstances similar to those under consideration wherever possible. Where proposed development and restoration techniques have not yet advanced to the pilot demonstration stage, initiate their use on a small scale to allow corrective action if unanticipated adverse impacts occur;

(e) Timing discharge to avoid spawning or migration seasons and other biologically critical time periods;

(f) Avoiding the destruction of remnant natural sites within areas already affected by development.

§ 230.76 Actions affecting human use.

Minimization of adverse effects on human use potential may be achieved by:

(a) Selecting discharge sites and following discharge procedures to prevent or minimize any potential damage to the aesthetically pleasing features of the aquatic site (e.g. viewscapes), particularly with respect to water quality;

(b) Selecting disposal sites which are not valuable as natural aquatic areas;

(c) Timing the discharge to avoid the seasons or periods when human recreational activity associated with the aquatic site is most important;

(d) Following discharge procedures which avoid or minimize the disturbance of aesthetic features on an aquatic site or ecosystem;

(e) Selecting sites that will not be detrimental or increase incompatible human activity, or require the need for frequent dredge or fill maintenance activity in remote fish and wildlife areas;

(f) Locating the disposal site outside of the vicinity of a public water supply intake.

§ 230.77 Other actions.

(a) In the case of fills, controlling runoff and other discharges from activities to be conducted on the fill;

(b) In the case of dams, designing water releases to accommodate the needs of fish and wildlife;

(c) In dredging projects funded by Federal agencies other than the Corps of Engineers, maintain desired water quality of the return discharge through agreement with the Federal funding authority on scientifically defensible pollutant concentration levels in addition to any applicable water quality standards;

(d) When a significant ecological change in the aquatic environment is proposed by the discharge of dredged or fill material, the permitting authority should consider the ecosystem that will be lost as well as the environmental benefits of the new system.

Subpart J – Compensatory Mitigation for Losses of Aquatic Resources

§ 230.91 Purpose and general considerations.

(a) Purpose.

(1) The purpose of this subpart is to establish standards and criteria for the use of all types of compensatory mitigation, including on-site and off-site permittee-responsible mitigation, mitigation banks, and in-lieu fee mitigation to offset unavoidable impacts to waters of the state authorized through the issuance of Orders.

(d) Accounting for regional variations. Where appropriate, the permitting authority shall account for regional characteristics of aquatic resource types, functions and services when determining performance standards and monitoring requirements for compensatory mitigation projects.

§ 230.92 Definitions.

For the purposes of this subpart, the following terms are defined:

Adaptive management means the development of a management strategy that anticipates likely challanges associated with compensatory mitigation projects and provides for the implementation of actions to address those challenges, as well as unforeseen changes to those projects. It requires consideration of the risk, uncertainty, and dynamic nature of compensatory mitigation projects and guides modification of those projects to optimize performance. It includes the selection of appropriate measures that will ensure that the aquatic resource functions are provided and involves analysis of monitoring results to identify potential problems of a compensatory mitigation project and the identification and implementation of measures to rectify those problems.

Buffer means an upland, wetland, and/or riparian area that protects and/or enhances aquatic resource functions associated with waters of the state from disturbances associated with adjacent land uses.

Compensatory mitigation means the restoration (re-establishment or rehabilitation), establishment (creation), enhancement, and/or in certain circumstances preservation of aquatic resources for the purposes of offsetting unavoidable adverse impacts which remain after all appropriate and practicable avoidance and minimization has been achieved.

Compensatory mitigation project means compensatory mitigation implemented by the permittee as a requirement of an Order (i.e., permittee-responsible mitigation), or by a mitigation bank or an in-lieu fee program.

Condition means the relative ability of an aquatic resource to support and maintain a community of organisms having a species composition, diversity, and functional organization comparable to reference aquatic resources in the region.

Credit means a unit of measure (e.g., a functional or areal measure or other suitable metric) representing the accrual or attainment of aquatic functions at a compensatory mitigation site. The measure of aquatic functions is based on the resources restored, established, enhanced, or preserved.

Days means calendar days.

Debit means a unit of measure (e.g., a functional or areal measure or other suitable metric) representing the loss of aquatic functions at an impact or project site. The measure of aquatic functions is based on the resources impacted by the authorized activity.

Enhancement means the manipulation of the physical, chemical, or biological characteristics of an aquatic resource to heighten, intensify, or improve a specific aquatic resource function(s). Enhancement results in the gain of selected aquatic resource function(s), but may also lead to a decline in other aquatic resource function(s). Enhancement does not result in a gain in aquatic resource area.

Establishment (creation) means the manipulation of the physical, chemical, or biological characteristics present to develop an aquatic resource that did not previously exist at an upland site. Establishment results in a gain in aquatic resource area and functions.

Functional capacity means the degree to which an area of aquatic resource performs a specific function.

Functions means the physical, chemical, and biological processes that occur in ecosystems.

Impact means adverse effect.

In-kind means a resource of a similar structural and functional type to the impacted resource.

In-lieu fee program means a program involving the restoration, establishment, enhancement, and/or preservation of aquatic resources through funds paid to a governmental or non-profit natural resources management entity to satisfy compensatory mitigation requirements for Orders. Similar to a mitigation bank, an in-lieu fee program sells compensatory mitigation credits to permittees whose obligation to provide compensatory mitigation is then transferred to the in-lieu program sponsor. However, the rules governing the operation and use of in-lieu fee programs are somewhat different from the rules governing operation and use of mitigation banks. The operation and use of an in-lieu fee program are governed by an in-lieu fee program instrument.

In-lieu fee program instrument means the legal document for the establishment, operation, and use of an in-lieu fee program.

Instrument means mitigation banking instrument or in-lieu fee program instrument.

Mitigation bank means a site, or suite of sites, where resources (e.g., wetlands, streams, riparian areas) are restored, established, enhanced, and/or preserved for the purpose of providing compensatory mitigation for impacts authorized by Orders. In general, a mitigation bank sells compensatory mitigation credits to permittees whose obligation to provide compensatory mitigation is then transferred to the mitigation bank sponsor. The operation and use of a mitigation bank are governed by a mitigation banking instrument.

Mitigation banking instrument means the legal document for the establishment, operation, and use of an in-lieu fee program.

Off-site means an area that is neither located on the same parcel of land as the impact site, nor on a parcel of land contiguous to the parcel containing the impact site.

On-site means an area located on the same parcel of land as the impact site, or on a parcel of land contiguous to the impact site.

Out-of-kind means a resource of a different structural and functional type from the impacted resource.

Performance standards are observable or measurable physical (including hydrological), chemical and/or biological attributes that are used to determine if a compensatory mitigation project meets its objectives.

Permittee-responsible mitigation means an aquatic resource restoration, establishment, enhancement, and/or preservation activity undertaken by the permittee (or an authorized agent or contractor) to provide compensatory mitigation for which the permittee retains full responsibility.

Preservation means the removal of a threat to, or preventing the decline of, aquatic resources by an action in or near those aquatic resources. This term includes activities commonly associated with the protection and maintenance of aquatic resources through the implementation of appropriate legal and physical mechanisms. Preservation does not result in a gain of aquatic resource area or functions.

Re-establishment means the manipulation of the physical, chemical, or biological characteristics of a site with the goal of returning natural/historic functions to a former aquatic resource. Re-establishment results in rebuilding a former aquatic resource and results in a gain in aquatic resource area and functions.

Reference aquatic resources are a set of aquatic resources that represent the full range of variability exhibited by a regional class of aquatic resources as a result of natural processes and anthropogenic disturbances.

Rehabilitation means the manipulation of the physical, chemical, or biological characteristics of a site with the goal of repairing natural/historic functions to a degraded aquatic resource. Rehabilitation results in a gain in aquatic resource function, but does not result in a gain in aquatic resource area.

Restoration means the manipulation of the physical, chemical, or biological characteristics of a site with the goal of returning natural/historic functions to a former or degraded aquatic resource. For the purpose of tracking net gains in aquatic resource area, restoration is divided into two categories: re-establishment and rehabilitation.

Service area means the geographic area within which impacts can be mitigated at a specific mitigation bank or an in-lieu fee program, as designated in its instrument.

Services mean the benefits that human populations receive from functions that occur in ecosystems.

Sponsor means any public or private entity responsible for establishing, and in most circumstances, operating a mitigation bank or in-lieu fee program.

Temporal loss is the time lag between the loss of aquatic resource functions caused by the permitted impacts and the replacement of aquatic resource functions at the compensatory mitigation site. Higher compensation ratios may be required to compensate for temporal loss. When the compensatory mitigation project is initiated prior to, or concurrent with, the permitted impacts, the permitting authority may determine that compensation for temporal loss is not necessary, unless the resource has a long development time.

Watershed means a land area that drains to a common waterway, such as a stream, lake, estuary, wetland, or ultimately the ocean.

Watershed approach is defined above in the main text of these Procedures.

Watershed plan is defined above in the main text of these Procedures.

§ 230.93 General compensatory mitigation requirements.

(a) General Considerations.

(1) The fundamental objective of compensatory mitigation is to offset environmental losses resulting from unavoidable impacts to waters of the state authorized by Orders. The permitting authority must determine the compensatory mitigation to be required in an Order, based on what would be environmentally preferable. In making this determination, the permitting authority must assess the likelihood for ecological success and sustainability, and the location of the compensation site relative to the impact site and their significance within the watershed, and the costs of the compensatory mitigation project. In many cases, the environmentally preferable compensatory mitigation may be provided through mitigation banks or in-lieu fee programs because they usually involve consolidating compensatory mitigation projects where ecologically appropriate, consolidating resources, providing financial planning and scientific expertise (which often is not practical for permittee-responsible compensatory mitigation projects), reducing temporal losses of functions, and reducing uncertainty over project success. Compensatory mitigation requirements must be commensurate with the amount and type of impact that is associated with a particular Order. Applicants are responsible for proposing an appropriate compensatory mitigation option to offset unavoidable impacts.

(2) Compensatory mitigation may be performed using methods of restoration, enhancement, establishment, and in certain circumstances preservation. Restoration should generally be the first option considered because the likelihood of success is greater and the impacts to potentially ecologically important uplands are reduced compared to establishment, and the potential gains in terms of aquatic resource functions are greater, compared to enhancement and preservation.

(3) Compensatory mitigation projects may be sited on public or private lands. Credits for compensatory mitigation projects on public land must be based solely on aquatic resource functions provided by the compensatory mitigation project, over and above those provided by public programs already planned or in place. All compensatory mitigation projects must comply with the standards in section IV of these Procedures, if they are to be used to provide compensatory mitigation for activities authorized by Orders, regardless of whether they are sited on public or private lands and whether the sponsor is a governmental or private entity.

(b) Type and location of compensatory mitigation.

(1) In general, the required compensatory mitigation should be located within the same watershed as the impact site, and should be located where it is most likely to successfully replace lost functions and services, taking into account such watershed scale features as aquatic habitat diversity, habitat connectivity, relationships to hydrologic sources (including the availability of water rights), trends in land use, ecological benefits, and compatibility with adjacent land uses. When compensating for impacts to marine resources, the location of the compensatory mitigation site should be chosen to replace lost functions and services within the same marine ecological system (e.g., reef complex, littoral drift cell). Compensation for impacts to aquatic resources in coastal watersheds (watersheds that include a tidal water body) should also be located in a coastal watershed where practicable. Compensatory mitigation projects should not be located where they will increase risks to aviation by attracting wildlife to areas where aircraft-wildlife strikes may occur (e.g., near airports).

(2) Mitigation bank credits. When permitted impacts are located within the service area of an approved mitigation bank, and the bank has the appropriate number and resource type of

credits available, the permittee's compensatory mitigation requirements may be met by securing those credits from the sponsor. Since an approved instrument (including an approved mitigation plan and appropriate real estate and financial assurances) for a mitigation bank is required to be in place before its credits can begin to be used to compensate for authorized impacts, use of a mitigation bank can help reduce risk and uncertainty, as well as temporal loss of resource functions and services. Mitigation bank credits are not released for debiting until specific milestones associated with the mitigation bank site's protection and development are achieved, thus use of mitigation bank credits can also help reduce risk that mitigation will not be fully successful. Mitigation banks typically involve larger, more ecologically valuable parcels, and more rigorous scientific and technical analysis, planning and implementation than permittee-responsible mitigation. Also, development of a mitigation bank requires site identification in advance, project-specific planning, and significant investment of financial resources that is often not practicable for many in-lieu fee programs. For these reasons, the permitting authority should give preference to the use of mitigation bank credits when these considerations are applicable. However, these same considerations may also be used to override this preference, where appropriate, as, for example, where an in-lieu fee program has released credits available from a specific approved in-lieu fee project, or a permittee-responsible project will restore an outstanding resource based on rigorous scientific and technical analysis.

(3) In-lieu fee program credits. Where permitted impacts are located within the service area of an approved in-lieu fee program, and the sponsor has the appropriate number and resource type of credits available, the permittee's compensatory mitigation requirements may be met by securing those credits from the sponsor. Where permitted impacts are not located in the service area of an approved mitigation bank, or the approved mitigation bank does not have the appropriate number and resource type of credits available to offset those impacts, in-lieu fee mitigation, if available, is generally preferable to permittee-responsible mitigation. In-lieu fee projects typically involve larger, more ecologically valuable parcels, and more rigorous scientific and technical analysis, planning and implementation than permittee-responsible mitigation. They also devote significant resources to identifying and addressing high-priority resource needs on a watershed scale, as reflected in their compensation planning framework. For these reasons, the permitting authority should give preference to in-lieu fee program credits over permittee-responsible mitigation, where these considerations are applicable. However, as with the preference for mitigation bank credits, these same considerations may be used to override this preference where appropriate. Additionally, in cases where permittee-responsible mitigation is likely to successfully meet performance standards before advance credits secured from an in-lieu fee program are fulfilled, the permitting authority should also give consideration to this factor in deciding between in-lieu fee mitigation and permittee-responsible mitigation.

(4) Permittee-responsible mitigation under a watershed approach. Where permitted impacts are not in the service area of an approved mitigation bank or in-lieu fee program that has the appropriate number and resource type of credits available, permittee-responsible mitigation is the only option. Where practicable and likely to be successful and sustainable, the resource type and location for the required permittee-responsible compensatory mitigation should be determined using the principles of a watershed approach as outlined in paragraph (c) of this section.

(5) Permittee-responsible mitigation through on-site and in-kind mitigation. In cases where a watershed approach is not practicable, the permitting authority should consider opportunities to

offset anticipated aquatic resource impacts by requiring on-site and in-kind compensatory mitigation. The permitting authority must also consider the practicability of on-site compensatory mitigation and its compatibility with the proposed project.

(6) Permittee-responsible mitigation through off-site and/or out-of-kind mitigation. If, after considering opportunities for on-site, in-kind compensatory mitigation as provided in paragraph (b)(5) of this section, the permitting authority determines that these compensatory mitigation opportunities are not practicable, are unlikely to compensate for the permitted impacts, or will be incompatible with the proposed project, and an alternative, practicable off-site and/or out-of-kind mitigation opportunity is identified that has a greater likelihood of offsetting the permitted impacts or is environmentally preferable to on-site or in-kind mitigation, the permitting authority should require that this alternative compensatory mitigation be provided.

(c) Watershed approach to compensatory mitigation.

(1) The permitting authority must use a watershed approach to establish compensatory mitigation requirements in Orders as described in the main text of the Procedures. Where a watershed plan is available, the permitting authority will determine whether the plan meets the definition of watershed plan in the Procedures and therefore is appropriate for use in the watershed approach for compensatory mitigation. In cases where the permitting authority determines that an appropriate watershed plan is available, the watershed approach should be based on that plan. Where no such plan is available, the watershed approach should be based on information provided by the project sponsor or available from other sources. The ultimate goal of a watershed approach is to maintain and improve the abundance, diversity, and condition of aquatic resources within watersheds through strategic selection of compensatory mitigation sites.

(2) Considerations.

(i) A watershed approach to compensatory mitigation considers the importance of condition, landscape position and resource type of compensatory mitigation projects for the sustainability of aquatic resource functions within the watershed. Such an approach considers how the condition, types, and locations of compensatory mitigation projects will provide the desired aquatic resource functions, and will continue to function over time in a changing landscape. It also considers the habitat requirements of important species, habitat loss or conversion trends, sources of watershed impairment, and current development trends, as well as the requirements of other regulatory and non-regulatory programs that affect the watershed, such as storm water management or habitat conservation programs. It includes the protection and maintenance of terrestrial resources, such as non-wetland riparian areas and uplands, when those resources contribute to or improve the overall ecological functioning of aquatic resources in the watershed. Compensatory mitigation requirements determined through the watershed approach should not focus exclusively on specific functions (e.g., water quality or habitat for certain species), but should provide, where practicable, the suite of functions typically provided by the affected aquatic resource.

(ii) Locational factors (e.g., hydrology, surrounding land use) are important to the success of compensatory mitigation for impacted habitat functions and may lead to siting of such mitigation away from the project area. However, consideration should also be given to functions and services (e.g., water quality, flood control, shoreline protection) that will likely need to be addressed at or near the areas impacted by the permitted impacts.

(iii) A watershed approach may include on-site compensatory mitigation, off-site compensatory mitigation (including mitigation banks or in-lieu fee programs), or a combination of on-site and off-site compensatory mitigation.

(iv) A watershed approach to compensatory mitigation should include, to the extent practicable, inventories of historic and existing aquatic resources, including identification of degraded aquatic resources, and identification of immediate and long-term aquatic resource needs within watersheds that can be met through permittee-responsible mitigation projects, mitigation banks, or in-lieu fee programs. Planning efforts should identify and prioritize aquatic resource restoration, establishment, and enhancement activities, and preservation of existing aquatic resources that are important for maintaining or improving ecological functions of the watershed. The identification and prioritization of resource needs should be as specific as possible, to enhance the usefulness of the approach in determining compensatory mitigation requirements.

(v) A watershed approach is not appropriate in areas where watershed boundaries do not exist, such as marine areas. In such cases, an appropriate spatial scale should be used to replace lost functions and services within the same ecological system (e.g., reef complex, littoral drift cell).

(3) Information Needs.

(i) In the absence of a watershed plan determined by the permitting authority under paragraph (c)(1) of this section to be appropriate for use in the watershed approach, the permitting authority will use a watershed approach based on analysis of information regarding watershed conditions (as identified in the watershed profile) and needs, including potential sites for aquatic resource restoration activities and priorities for aquatic resource restoration and preservation. Such information includes: Current trends in habitat loss or conversion; cumulative impacts of past development activities, current development trends, the presence and needs of sensitive species; site conditions that favor or hinder the success of compensatory mitigation projects; and chronic environmental problems such as flooding or poor water quality.

(ii) This information may be available from sources such as wetland maps; soil surveys; U.S. Geological Survey topographic and hydrologic maps; aerial photographs; information on rare, endangered and threatened species and critical habitat; local ecological reports or studies; and other information sources that could be used to identify locations for suitable compensatory mitigation projects in the watershed.

(iii) The level of information and analysis needed to support a watershed approach must be commensurate with the scope and scale of the proposed impacts requiring an Order, as well as the functions lost as a result of those impacts.

(4) Watershed Scale. The size of watershed addressed using a watershed approach should not be larger than is appropriate to ensure that the aquatic resources provided through

compensation activities will effectively compensate for adverse environmental impacts resulting from activities authorized by Orders. The permitting authority should consider relevant environmental factors and appropriate locally-developed standards and criteria when determining the appropriate watershed scale in guiding compensation activities.

(d) Site selection.

(1) The compensatory mitigation project site must be ecologically suitable for providing the desired aquatic resource functions. In determining the ecological suitability of the compensatory mitigation project site, the permitting authority must consider, to the extent practicable, the following factors:

(i) Hydrological conditions, soil characteristics, and other physical and chemical characteristics;

(ii) Watershed-scale features, such as aquatic habitat diversity, habitat connectivity, and other landscape scale functions;

(iii) The size and location of the compensatory mitigation site relative to hydrologic sources (including the availability of water rights) and other ecological features;

(iv) Compatibility with adjacent land uses and watershed management plans;

(v) Reasonably foreseeable effects the compensatory mitigation project will have on ecologically important aquatic or terrestrial resources (e.g., shallow sub-tidal habitat, mature forests), cultural sites, or habitat for federally- or state-listed threatened and endangered species; and

(vi) Other relevant factors including, but not limited to, development trends, anticipated land use changes, habitat status and trends, the relative locations of the impact and mitigation sites in the stream network, local or regional goals for the restoration or protection of particular habitat types or functions (e.g., re-establishment of habitat corridors or habitat for species of concern), water quality goals, floodplain management goals, and the relative potential for chemical contamination of the aquatic resources.

(2) Permitting authorities may require on-site, off-site, or a combination of on-site and off-site compensatory mitigation to replace permitted losses of aquatic resource functions and services.

(3) Applicants should propose compensation sites adjacent to existing aquatic resources or where aquatic resources previously existed.

(e) Mitigation type.

(1) In general, in-kind mitigation is preferable to out-of-kind mitigation because it is most likely to compensate for the functions and services lost at the impact site. For example, tidal wetland compensatory mitigation projects are most likely to compensate for unavoidable impacts to tidal wetlands, while perennial stream compensatory mitigation projects are most likely to

compensate for unavoidable impacts to perennial streams. Thus, except as provided in paragraph (e)(2) of this section, the required compensatory mitigation shall be of a similar type to the affected aquatic resource.

(2) If the permitting authority determines, using the watershed approach in accordance with paragraph (c) of this section that out-of-kind compensatory mitigation will serve the aquatic resource needs of the watershed, the permitting authority may authorize the use of such out-of-kind compensatory mitigation. The basis for authorization of out-of-kind compensatory mitigation must be documented in the administrative record for the Order action.

(3) For difficult-to-replace resources (e.g., bogs, fens, springs, streams, vegetated seasonal wetlands, slope and seep wetlands, vernal pools, and wet meadows) if further avoidance and minimization is not practicable, the required compensation should be provided, if practicable, through in-kind rehabilitation, enhancement, or preservation since there is greater certainty that these methods of compensation will successfully offset permitted impacts.

(f) Amount of compensatory mitigation.

(1) If the permitting authority determines that compensatory mitigation is necessary to offset unavoidable impacts to aquatic resources, the amount of required compensatory mitigation must be, to the extent practicable, sufficient to replace lost aquatic resource functions. In cases where appropriate functional or condition assessment methods or other suitable metrics are available, these methods should be used where practicable to determine how much compensatory mitigation is required.

(2) The permitting authority must require a mitigation ratio greater than one-to-one where necessary to account for the method of compensatory mitigation (e.g., preservation), the likelihood of success, differences between the functions lost at the impact site and the functions expected to be produced by the compensatory mitigation project, temporal losses of aquatic resource functions, the difficulty of restoring or establishing the desired aquatic resource type and functions, and/or the distance between the affected aquatic resource and the compensation site. The rationale for the required replacement ratio must be documented in the administrative record for the Order action.

(3) If an in-lieu fee program will be used to provide the required compensatory mitigation, and the appropriate number and resource type of released credits are not available, the permitting authority must require sufficient compensation to account for the risk and uncertainty associated with in-lieu fee projects that have not been implemented before the permitted impacts have occurred.

(g) Use of mitigation banks and in-lieu fee programs. Mitigation banks and in-lieu fee programs may be used to compensate for impacts to aquatic resources authorized by general Orders and individual Orders in accordance with the preference hierarchy in paragraph (b) of this section. Mitigation banks and in-lieu fee programs may also be used to satisfy requirements arising out of an enforcement action, such as supplemental environmental projects.

(h) Preservation.

(1) Preservation may be used to provide compensatory mitigation for activities authorized by Orders when all the following criteria are met:

(i) The resources to be preserved provide important physical, chemical, or biological functions for the watershed;

(ii) The resources to be preserved contribute significantly to the ecological sustainability of the watershed. In determining the contribution of those resources to the ecological sustainability of the watershed, the permitting authority must use appropriate quantitative assessment tools where available;

(iii) Preservation is determined by the permitting authority to be appropriate and practicable;

(iv) The resources are under threat of destruction or adverse modifications; and

(v) The preserved site will be permanently protected through an appropriate real estate or other legal instrument (e.g., easement, title transfer to state resource agency or land trust).

(2) Where preservation is used to provide compensatory mitigation, to the extent appropriate and practicable the preservation shall be done in conjunction with aquatic resource restoration, establishment, and/or enhancement activities. This requirement may be waived by the permitting authority where preservation has been identified as a high priority using a watershed approach described in paragraph (c) of this section, but compensation ratios shall be higher.

(i) Buffers. The permitting authority may require the restoration, establishment, enhancement, and preservation, as well as the maintenance, of riparian areas and/or buffers around aquatic resources where necessary to ensure the long-term viability of those resources. Buffers may also provide habitat or corridors necessary for the ecological functioning of aquatic resources. If buffers are required by the permitting authority as part of the compensatory mitigation project, compensatory mitigation credit will be provided for those buffers, as provided in section IV B.5 (c).

(j) Relationship to other federal, tribal, state, and local programs.

(1) Compensatory mitigation projects for Orders may also be used to satisfy the environmental requirements of other programs, such as tribal, state, or local wetlands regulatory programs, other federal programs such as the Surface Mining Control and Reclamation Act, Corps civil works projects, and Department of Defense military construction projects, consistent with the terms and requirements of these programs and subject to the following considerations:

(i) The compensatory mitigation project must include appropriate compensation required by the Order for unavoidable impacts to aquatic resources authorized by that Order.

(ii) Under no circumstances may the same credits be used to provide mitigation for more than one permitted activity. However, where appropriate, compensatory mitigation projects, including mitigation banks and in-lieu fee projects, may be designed to holistically address requirements under multiple programs and authorities for the same activity.

(2) Except for projects undertaken by federal agencies, or where federal funding is specifically authorized to provide compensatory mitigation, federally-funded aquatic resource restoration or conservation projects undertaken for purposes other than compensatory mitigation, such as the Wetlands Reserve Program, Conservation Reserve Program, and Partners for Wildlife Program activities, cannot be used for the purpose of generating compensatory mitigation credits for activities authorized by Orders. However, compensatory mitigation credits may be generated by activities undertaken in conjunction with, but supplemental to, such programs in order to maximize the overall ecological benefits of the restoration or conservation project.

(3) Compensatory mitigation projects may also be used to provide compensatory mitigation under the federal and state Endangered Species Act or for Natural Community Conservation Plans and Habitat Conservation Plans, as long as they comply with the requirements of paragraph (j)(1) of this section.

(k) Order conditions.

(1) The compensatory mitigation requirements for an Order, including the amount and type of compensatory mitigation, must be clearly stated in the special conditions of the individual Order or authorization to use the general Order. The special conditions must be enforceable.

(2) For an Order that requires permittee-responsible mitigation, the special conditions must:

(i) Identify the party responsible for providing the compensatory mitigation;

(ii) Incorporate, by reference, the final or draft mitigation plan approved by the permitting authority;

(iii) State the objectives, performance standards, and monitoring required for the compensatory mitigation project, unless they are provided in the approved final mitigation plan; and

(iv) Describe any required financial assurances or long-term management provisions for the compensatory mitigation project, unless they are specified in the approved final mitigation plan.

(4) If a mitigation bank or in-lieu fee program is used to provide the required compensatory mitigation, the special conditions must indicate whether a mitigation bank or in-lieu fee program will be used, and specify the number and resource type of credits the permittee is required to secure. In the case of an individual Order, the special condition must also identify the specific mitigation bank or in-lieu fee program that will be used. For authorizations to use a general Order, the special conditions may either identify the specific mitigation bank or in-lieu fee program, or state that the specific mitigation bank or in-lieu fee program used to provide the required compensatory mitigation must be approved by the permitting authority before the credits are secured.

(l) Party responsible for compensatory mitigation.

(1) For permittee-responsible mitigation, the special conditions of the Order must clearly indicate the party or parties responsible for the implementation, performance, and long-term management of the compensatory mitigation project.

(3) If use of a mitigation bank or in-lieu fee program is approved by the permitting authority to provide part or all of the required compensatory mitigation for an Order, the permittee retains responsibility for providing the compensatory mitigation until the appropriate number and resource type of credits have been secured from a sponsor and the permitting authority has received documentation that confirms that the sponsor has accepted the responsibility for providing the required compensatory mitigation. This documentation may consist of a letter or form signed by the sponsor, with the Order number and a statement indicating the number and resource type of credits that have been secured from the sponsor. Copies of this documentation will be retained in the administrative records for both the Order and the instrument. If the sponsor fails to provide the required compensatory mitigation, the permitting authority may pursue measures against the sponsor to ensure compliance.

(m) Timing. Implementation of the compensatory mitigation project shall be, to the maximum extent practicable, in advance of or concurrent with the activity causing the authorized impacts. The permitting authority shall require, to the extent appropriate and practicable, additional compensatory mitigation to offset temporal losses of aquatic functions that will result from the permitted activity.

(n) Financial assurances.

(1) The permitting authority shall require sufficient financial assurances to ensure a high level of confidence that the compensatory mitigation project will be successfully completed, in accordance with applicable performance standards. In cases where an alternate mechanism is available to ensure a high level of confidence that the compensatory mitigation will be provided and maintained (e.g., a formal, documented commitment from a government agency or public authority) the permitting authority may determine that financial assurances are not necessary for that compensatory mitigation project.

(2) The amount of the required financial assurances must be determined by the permitting authority, in consultation with the project sponsor, and must be based on the size and complexity of the compensatory mitigation project, the degree of completion of the project at the time of project approval, the likelihood of success, the past performance of the project sponsor, and any other factors the permitting authority deems appropriate. Financial assurances may be in the form of performance bonds, escrow accounts, casualty insurance, letters of credit, legislative appropriations for government sponsored projects, or other appropriate instruments, subject to the approval of the permitting authority. The rationale for determining the amount of the required financial assurances must be documented in the administrative record for either the Order or the instrument. In determining the assurance amount, the permitting authority shall consider the cost of providing replacement mitigation, including costs for land acquisition, planning and engineering, legal fees, mobilization, construction, and monitoring.

(3) If financial assurances are required, the Order must include a special condition requiring the financial assurances to be in place prior to commencing the permitted activity.

(4) Financial assurances shall be phased out once the compensatory mitigation project has been determined by the permitting authority to be successful in accordance with its performance standards. The Order or instrument must clearly specify the conditions under which the financial assurances are to be released to the permittee, sponsor, and/or other financial assurance provider, including, as appropriate, linkage to achievement of performance standards, adaptive management, or compliance with special conditions.

(5) A financial assurance must be in a form that ensures that the permitting authority will receive notification at least 120 days in advance of any termination or revocation. For third-party assurance providers, this may take the form of a contractual requirement for the assurance provider to notify the permitting authority at least 120 days before the assurance is revoked or terminated.

(6) Financial assurances shall be payable at the direction of the permitting authority to his designee or to a standby trust agreement. When a standby trust is used (e.g., with performance bonds or letters of credit) all amounts paid by the financial assurance provider shall be deposited directly into the standby trust fund for distribution by the trustee in accordance with the permitting authority's instructions.

(o) Compliance with applicable law. The compensatory mitigation project must comply with all applicable federal, state, and local laws. The Order, mitigation banking instrument, or in-lieu fee program instrument must not require participation by the permitting authority in project management, including receipt or management of financial assurances or long-term financing mechanisms, except as determined by the permitting authority to be consistent with its statutory authority, mission, and priorities.

§ 230.94 Planning and documentation.

(a) Pre-application consultations. Potential applicants for Orders are encouraged to participate in pre-application meetings with the permitting authority and appropriate agencies to discuss potential mitigation requirements and information needs.

(c) Mitigation plan.

(1) Preparation and Approval.

(i) For individual Orders, the permittee must prepare a draft mitigation plan and submit it to the permitting authority for review prior to issuing the Order. After addressing any comments provided by the permitting authority, the permittee must prepare a final mitigation plan, which must be approved by the permitting authority prior to commencing work in waters of the state. The approved final mitigation plan must be incorporated into the individual Order either as an attachment or by reference. The final mitigation plan must include the items described in paragraphs (c)(2) through (c)(14) of this section, but the level of detail of the mitigation plan should be commensurate with the scale and scope of the impacts. As an alternative, the permitting authority may determine that it would be more appropriate to address any of the items described in paragraphs (c)(2) through (c)(14) of this section as Order conditions, instead of components of a compensatory mitigation plan. For permittees who intend to fulfill their compensatory mitigation obligations by securing credits from approved mitigation banks or in-lieu fee programs, their mitigation plans need include only the items described in paragraphs (c)(5) and (c)(6) of this section, and the name of the specific mitigation bank or in-lieu fee program to be used.

(2) Objectives. A description of the resource type(s) and amount(s) that will be provided, the method of compensation (i.e., restoration, establishment, enhancement, and/or preservation), and the manner in which the resource functions of the compensatory mitigation project will address the needs of the watershed, ecoregion, physiographic province, or other geographic area of interest.

(3) Site selection. A description of the factors considered during the site selection process. This should include consideration of watershed needs, on-site alternatives where applicable, and the practicability of accomplishing ecologically self-sustaining aquatic resource restoration, establishment, enhancement, and/or preservation at the compensatory mitigation project site. (See § 230.93(d).)

(4) Site protection instrument. A description of the legal arrangements and instrument, including site ownership, that will be used to ensure the long-term protection of the compensatory mitigation project site (see § 230.97(a)).

(5) Baseline information. A description of the ecological characteristics of the proposed compensatory mitigation project site and, in the case of an application for an Order, the impact site. This may include descriptions of historic and existing plant communities, historic and existing hydrology, soil conditions, a map showing the locations of the impact and mitigation site(s) or the geographic coordinates for those site(s), and other site characteristics appropriate to the type of resource proposed as compensation. The baseline information should also include a delineation of waters of the state on the proposed compensatory mitigation project site. A prospective permittee planning to secure credits from an approved mitigation bank or in-lieu fee program only needs to provide baseline information about the impact site, not the mitigation bank or in-lieu fee project site.

(6) Determination of credits. A description of the number of credits to be provided, including a brief explanation of the rationale for this determination. (See § 230.93(f).)

(i) For permittee-responsible mitigation, this should include an explanation of how the compensatory mitigation project will provide the required compensation for unavoidable impacts to aquatic resources resulting from the permitted activity.

(ii) For permittees intending to secure credits from an approved mitigation bank or in-lieu fee program, it should include the number and resource type of credits to be secured and how these were determined.

(7) Mitigation work plan. Detailed written specifications and work descriptions for the compensatory mitigation project, including, but not limited to, the geographic boundaries of the project; construction methods, timing, and sequence; source(s) of water, including connections to existing waters and uplands; methods for establishing the desired plant community; plans to control invasive plant species; the proposed grading plan, including elevations and slopes of the substrate; soil management; and erosion control measures. For stream compensatory mitigation projects, the mitigation work plan may also include other relevant information, such as planform geometry, channel form (e.g., typical channel cross-sections), watershed size, design discharge, and riparian area plantings.

(8) Maintenance plan. A description and schedule of maintenance requirements to ensure the continued viability of the resource once initial construction is completed.

(9) Performance standards. Ecologically-based standards that will be used to determine whether the compensatory mitigation project is achieving its objectives. (See § 230.95.)

(10) Monitoring requirements. A description of parameters to be monitored in order to determine if the compensatory mitigation project is on track to meet performance standards and if adaptive management is needed. A schedule for monitoring and reporting on monitoring results to the permitting authority must be included. (See § 230.96.)

(11) Long-term management plan. A description of how the compensatory mitigation project will be managed after performance standards have been achieved to ensure the long-term sustainability of the resource, including long-term financing mechanisms and the party responsible for long-term management. (See § 230.97(d).)

(12) Adaptive management plan. A management strategy to address unforeseen changes in site conditions or other components of the compensatory mitigation project, including the party or parties responsible for implementing adaptive management measures. The adaptive management plan will guide decisions for revising compensatory mitigation plans and implementing measures to address both foreseeable and unforeseen circumstances that adversely affect compensatory mitigation success. (See § 230.97(c).)

(13) Financial assurances. A description of financial assurances that will be provided and how they are sufficient to ensure a high level of confidence that the compensatory mitigation project will be successfully completed, in accordance with its performance standards (see § 230.93(n)).

(14) Other information. The permitting authority may require additional information as necessary to determine the appropriateness, feasibility, and practicability of the compensatory mitigation project.

§ 230.95 Ecological performance standards.

(a) The approved mitigation plan must contain performance standards that will be used to assess whether the project is achieving its objectives. Performance standards should relate to the objectives of the compensatory mitigation project, so that the project can be objectively evaluated to determine if it is developing into the desired resource type, providing the expected condition or functions, and attaining any other applicable metrics (e.g., acres).

(b) Performance standards must be based on attributes that are objective and verifiable. Ecological performance standards must be based on the best available science that can be measured or assessed in a practicable manner. Performance standards may be based on variables or measures of functional capacity or condition as described in assessment methodologies, measurements of hydrology or other aquatic resource characteristics, and/or comparisons to reference aquatic resources of similar type and landscape position. The use of reference aquatic resources to establish performance standards will help ensure that those performance standards are reasonably achievable, by reflecting the range of variability exhibited by the regional class of aquatic resources as a result of natural processes and anthropogenic disturbances. Performance standards based on measurements of hydrology should take into consideration the hydrologic variability exhibited by reference aquatic resources, especially wetlands. Where practicable, performance standards should take into account the expected stages of the aquatic resource development process, in order to allow early identification of potential problems and appropriate adaptive management.

§ 230.96 Monitoring.

(a) General.

(1) Monitoring the compensatory mitigation project site is necessary to determine if the project is meeting its performance standards, and to determine if measures are necessary to ensure that

the compensatory mitigation project is accomplishing its objectives. The submission of monitoring reports to assess the development and condition of the compensatory mitigation project is required, but the content and level of detail for those monitoring reports must be commensurate with the scale and scope of the compensatory mitigation project, as well as the compensatory mitigation project type. The mitigation plan must address the monitoring requirements for the compensatory mitigation project, including the parameters to be monitored, the length of the monitoring period, the party responsible for conducting the monitoring, the frequency for submitting monitoring reports to the permitting authority, and the party responsible for submitting those monitoring reports to the permitting authority.

(2) The permitting authority may conduct site inspections on a regular basis (e.g., annually) during the monitoring period to evaluate mitigation site performance.

(b) Monitoring period. The mitigation plan must provide for a monitoring period that is sufficient to demonstrate that the compensatory mitigation project has met performance standards, but not less than five years. A longer monitoring period must be required for aquatic resources with slow development rates (e.g., forested wetlands, bogs). Following project implementation, the permitting authority may reduce or waive the remaining monitoring requirements upon a determination that the compensatory mitigation project has achieved its performance standards. Conversely the permitting authority may extend the original monitoring period upon a determination that performance standards have not been met or the compensatory mitigation project is not on track to meet them. The permitting authority may also revise monitoring requirements when remediation and/or adaptive management is required.

(c) Monitoring reports.

(1) The permitting authority must determine the information to be included in monitoring reports. This information must be sufficient for the permitting authority to determine how the compensatory mitigation project is progressing towards meeting its performance standards, and may include plans (such as as-built plans), maps, and photographs to illustrate site conditions. Monitoring reports may also include the results of functional, condition, or other assessments used to provide quantitative or qualitative measures of the functions provided by the compensatory mitigation project site.

(2) The permittee or sponsor is responsible for submitting monitoring reports in accordance with the special conditions of the Order or the terms of the instrument. Failure to submit monitoring reports in a timely manner may result in compliance action by the permitting authority.

(3) Monitoring reports must be provided by the permitting authority to interested federal, tribal, state, and local resource agencies, and the public, upon request.

§ 230.97 Management.

(a) Site protection.

(1) The aquatic habitats, riparian areas, buffers, and uplands that comprise the overall compensatory mitigation project must be provided long-term protection through real estate instruments or other available mechanisms, as appropriate. Long-term protection may be provided through real estate instruments such as conservation easements held by entities such as federal, tribal, state, or local resource agencies, non-profit conservation organizations, or private land managers; the transfer of title to such entities; or by restrictive covenants. For government property, long-term protection may be provided through state or federal facility management plans or integrated natural resources management plans. When approving a method for long-term protection of non-government property other than transfer of title, the

permitting authority shall consider relevant legal constraints on the use of conservation easements and/or restrictive covenants in determining whether such mechanisms provide sufficient site protection. To provide sufficient site protection, a conservation easement or restrictive covenant should, where practicable, establish in an appropriate third party (e.g., governmental or non-profit resource management agency) the right to enforce site protections and provide the third party the resources necessary to monitor and enforce these site protections.

(2) The real estate instrument, management plan, or other mechenism providing long-term protection of the compensatory mitigation site must, to the extent appropriate and practicable, prohibit incompatible uses (e.g., clear cutting or minerel extraction) that might otherwise jeopardize the objectives of the compensatory mitigation project. Where appropriate, multiple instruments recognizing compatible uses (e.g., fishing or grazing rights) may be used.

(3) The real estate instrument, management plan, or other long-term protection mechanism must contain a provision requiring 60–day advance notification to the permitting authority before any action is taken to void or modify the instrument, management plan, or long-term protection mechanism, including transfer of title to, or establishment of any other legal claims over, the compensatory mitigation site.

(4) For compensatory mitigation projects on public lands, where state or Federal facility management plans or integrated natural resources management plans are used to provide long-term protection, and changes in statute, regulation, or agency needs or mission results in an incompatible use on public lands originally set aside for compensatory mitigation, the public egency authorizing the incompatible use is responsible for providing alternative compensatory mitigation that is acceptable to the permitting authority for any loss in functions resulting from the incompatible use.

(5) A real estate instrument, management plan, or other long-term protection mechanism used for site protection of permittee-responsible mitigation must be approved by the permitting authority in advance of, or concurrent with, the activity causing the authorized impacts.

(b) Sustainability. Compensatory mitigation projects shall be designed, to the maximum extent practicable, to be self-sustaining once performance standards have been achieved. This includes minimization of active engineering features (e.g., pumps) end appropriate siting to ensure that natural hydrology and landscape context will support long-term sustainability. Where active long-term management and maintenance are necessary to ensure long-term sustainability (e.g., prescribed burning, invasive species control, meintenance of water control structures, easement enforcement), the responsible party must provide for such management and maintenance. This includes the provision of long-term financing mechanisms where necessary. Where needed, the acquisition and protection of water rights must be secured and documented in the Order conditions or instrument.

(c) Adaptive management.

(1) If the compensatory mitigation project cannot be constructed in accordance with the approved mitigation plans, the permittee or sponsor must notify the permitting authority. A significant modification of the compensatory mitigation project requires approval from the permitting authority.

(2) If monitoring or other information indicates that the compensatory mitigation project is not progressing towards meeting its performance standards as anticipated, the responsible party must notify the permitting authority as soon as possible. The permitting authority will evaluate

and pursue measures to address deficiencies in the compensatory mitigation project. The permitting authority will consider whether the compensatory mitigation project is providing ecological benefits comparable to the original objectives of the compensatory mitigation project.

(3) The permitting authority, in consultation with the responsible party (and other federal, tribal, state, and local agencies, as appropriate), will determine the appropriate measures. The measures may include site modifications, design changes, revisions to maintenance requirements, and revised monitoring requirements. The measures must be designed to ensure that the modified compensatory mitigation project provides aquatic resource functions comparable to those described in the mitigation plan objectives.

(4) Performance standards may be revised in accordance with adaptive management to account for measures taken to address deficiencies in the compensatory mitigation project. Performance standards may also be revised to reflect changes in management strategies and objectives if the new standards provide for ecological benefits that are comparable or superior to the approved compensatory mitigation project. No other revisions to performance standards will be allowed except in the case of natural disasters.

(d) Long-term management.

(1) The Order conditions or instrument must identify the party responsible for ownership and all long-term management of the compensatory mitigation project. The Order conditions or instrument may contain provisions allowing the permittee or sponsor to transfer the long-term management responsibilities of the compensatory mitigation project site to a land stewardship entity, such as a public agency, non-governmental organization, or private land manager, after review and approval by the permitting authority. The land stewardship entity need not be identified in the original Order or instrument, as long as the future transfer of long-term management responsibility is approved by the permitting authority.

(2) A long-term management plan should include a description of long-term management needs, annual cost estimates for these needs, and identify the funding mechanism that will be used to meet those needs.

(3) Any provisions necessary for long-term financing must be addressed in the original Order or instrument. The permitting authority may require provisions to address inflationary adjustments and other contingencies, as appropriate. Appropriate long-term financing mechanisms include non-wasting endowments, trusts, contractual arrangements with future responsible parties, and other appropriate financial instruments. In cases where the long-term management entity is a public authority or government agency, that entity must provide a plan for the long-term financing of the site.

(4) For permittee-responsible mitigation, any long-term financing mechanisms must be approved in advance of the activity causing the authorized impacts.