CALIFORNIA AMPHIBIAN *and* REPTILE SPECIES *of* SPECIAL CONCERN

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CALIFORNIA DEPARTMENT OF FISH AND WILDLIFE

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FOREWORD

California boasts one of the most biologically diverse faunas in the United States, as well as one of the most threatened. One of the key elements of the state's efforts to protect its vertebrate fauna is through its Species of Special Concern program. The current volume, *California Amphibian and Reptile Species of Special Concern*, is an essential foundation upon which both biologists and state and federal agencies can manage the biological resources of the state. California has exceedingly sensitive species and ecosystems, many of which are at risk of extirpation or extinction as the state's environment changes at rates greater than at any time in history.

This book builds upon the shoulders of its predecessor from two decades ago (Jennings and Hayes 1994a), but it is not just a simple update. Jennings and Hayes surveyed an enormous number of experts to create a comprehensive publication on California's special concern amphibians and reptiles, and their volume was a key management tool for a generation of biologists. However, this new book goes several steps further, making it a necessary reference for wildlife and land managers, biologists, and nature lovers interested in amphibians and reptiles.

First, the maps generated for this book are stunning. They are literally beautiful enough to be framed, and detailed enough to guide resource managers. Second, there are color images of every taxon, generally taken in the field and highlighting the key features of each species. Third, the authors rely on the published literature to the maximum extent possible, pulling in the gray literature only when it is needed (which is often because many of these species are poorly known). But perhaps most importantly, the authors used multifactorial risk metrics that bring several measures of potential and actual threat into a single numeric score that captures the sensitivity of the species. The result is a tool that provides an important first pass at the difficult task of identifying those taxa that should be candidate Special Concern species.

Of course, there will always be important biological considerations that may argue against a strict interpretation of the metric scores, as the authors fully realize. For example, there are species on the Special Concern list that are so narrowly precinctive that the narrowness of their geographic range alone signals reason to be extra cautious about the species. The sandstone night lizard is one such taxon; its geographic range is much smaller than listed species such as black toad (*Bufo exsul*), and we know much less about the night lizard than we do about black toads. Regardless of the risk model score, this is a scary situation, and the narrowness of geographic range alone signals reason to be extremely cautious. Herpetologists are well aware of extinctions of entire species that were so narrowly precinctive that very subtle (sometimes unknown) environmental changes have caused those extinctions (e.g., the golden toad of Costa Rica, which had a geographic range the same size as that of the sandstone night lizard).

There are other species covered in this volume that will be challenging to manage for their protection in California. For example, the Gila monster (*Heloderma suspectum*) can be found in the extreme eastern part of the Mojave Desert in California (east of 116° longitude), where it has been recorded fewer than 30 times in the last 150 years. Within the distribution of Gila monsters in California, the pattern of rainfall includes winter rains and summer (monsoonal) rains; this biphasic pattern is typical in Utah, Nevada, and Arizona where Gila monsters are relatively more common. Throughout their geographic range, Gila monsters depend upon climate conditions conducive for reproduction by small mammals because neonatal small mammals are the principal prey for this species. However, climate is demonstrably changing in California to be warmer (especially in summers) and with increased frequencies of drought. These changes may not be mitigable at a local level, and this creates conservation challenges. Nevertheless, knowledge of both changes of climate and the biology of Gila monsters is meager, and this signals both that the Gila monster is clearly a reasonable candidate for SSC status and a need for additional research.

In keeping with this example, this volume calls for significantly increasing research and monitoring of these species. This is a recommendation that must be taken very seriously. Change to California wildlife is accelerating at a more rapid rate than ever before in history, and the best chance to protect California's Species of Special Concern from extirpation or extinction is increasing our knowledge of these poorly studied animals. Long-term monitoring of the status of populations is key, and contemporary methods such as population genomics can provide insights into population status and viability that were not possible just a few years ago.

As complete as it is, this volume should be considered a beginning, rather than a final set of definitive answers, for understanding ecologically sensitive amphibians and reptiles in California. It constitutes an enormously valuable benchmark, and also provides solid information about the biology and ecology of amphibian and reptile species in California. Now we need to pursue its recommendations so that we can facilitate the needed science that will help us protect California's biological resources. California needs to expand science and management of the state's precious biological resources so that our children and grandchildren, hopefully, will be able to experience no fewer species than are present in California today. This book is an important step in that direction.

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PREFACE

California's amphibians and reptiles are unique in the United States for the tremendous amount of evolutionary and ecological diversity that they represent. California is second only to Texas in terms of the number of native amphibians and reptiles found within a state and contains endemic species of all major groups except turtles and tortoises. The state is home to what might be the best-known example of ring speciation (in *Ensatina* salamanders), which provides a unique view into the process of species diversification. California is home to the tailed frog (*Ascaphus truei*), a species that is among the last surviving members of an ancient lineage that is the sister group to all other frogs on earth. It houses reptile and amphibian species with genetic- and temperature-dependent sex determination; species that lay eggs in the water, on land, or that are live-bearing; and species with a two-staged life cycle that undergo a profound metamorphosis, switching between distinctly different habitats in the process.

The California Department of Fish and Wildlife (formerly, California Department of Fish and Game) is the trustee agency for California's fish and wildlife resources. The challenges associated with effective management and conservation of these resources are formidable in California, where a large human population, diverse stakeholder interests, and extremely high biotic diversity must be jointly managed. Despite the challenge of implementing effective conservation in the state, doing so is an important and worthy goal given the vast diversity that the state supports. We have attempted to evaluate conservation status for the state's amphibians and reptiles openly and transparently, relying on both the best available science and the breadth of expert opinions relating to amphibian and reptile conservation in California. We have sought (and received) broad feedback from a wide range of interested parties including agency representatives, academic scientists, and avocational herpetologists and used this combined input to make informed recommendations about conservation risk and management needs for California's amphibians and reptiles. We have also highlighted where data are lacking and discussed how the community might fill these gaps in our knowledge. Our goal is for this volume to serve as both a summary of where we stand and a launching point for what we can achieve in the management and restoration of healthy amphibian and reptile populations in California.

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May 2015

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This project was a collaborative effort between the California Department of Fish and Wildlife, the Amphibian and Reptile Species of Special Concern Technical Advisory Committee, and contributors from government agencies, universities, nongovernmental organizations, and private citizens drawn from across California. Without the guidance, assistance, and advice from many individuals, this project would not have been possible. In particular, we thank Betsy Bolster for her help and support through all phases of this project. Betsy's deep experience has been indispensable in guiding and focusing the scope and aims for our efforts, and her advice has been an indispensable part of the success of the project. We also thank Kristi Cripe for her efforts and expertise in developing the maps for this document and her general advice on cartography. Sean Barry, Robert Fisher, and Hart Welsh, together comprising our Technical Advisory Committee, constituted a key resource at several phases of this project. Their advice directly shaped the development of the scoring metrics, nominee list, and the composition of the final Species of Special Concern list. They also provided extensive unpublished data and knowledge about California's amphibians and reptiles, answered countless questions, and served as a sounding board throughout the project.

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OVERVIEW

ABSTRACT

We provide a synthesis of the conservation risk faced by amphibians and reptiles in California that qualify as Species of Special Concern. After assembling a full list of the native amphibian and reptile taxa that are known to occur in the state, we developed a potential set of 73 nominee taxa that might qualify as Species of Special Concern. We developed eight metrics that capture key elements of declining and at-risk species, scored them for all 73 nominee taxa based on an extensive literature review, examined them on a case-by-case basis, and developed a final set of 45 Species of Special Concern. We then developed species accounts for each Species of Special Concern, documenting available information on their basic biology, known or hypothesized reasons for decline, and proposed management and future research needs. Overall, we sought to produce a clear, transparent document that explicitly states why decisions were made and supported with a summary of the best available science. We relied on peer-reviewed literature whenever possible to support those decisions.

Our evaluation resulted in 16 Species of Special Concern categorized as Priority 1 (those of greatest concern), 14 as Priority 2, 12 as Priority 3, and 3 which we could not prioritize based on available data. Our comparative analyses demonstrated that there were certain sets of organisms, geographic areas, and groups of ecological specialists in which species of greatest concern tended to be concentrated. Taxonomically, frogs, salamanders, and turtles all had higher average metric scores than lizards or snakes, mirroring the fraction of those taxa listed at the state and federal levels, and suggesting that these lineages are often of greatest conservation concern. There was also a strong trend for aquatic taxa to experience a greater conservation risk than terrestrial species. Geographically, southern California harbored more Species of Special Concern than central or northern California. This pattern was driven primarily by reptiles, which have a preponderance of at-risk species in the Southern California Coast, Southern California Mountains and Valleys, and the Mojave Desert ecoregions. Amphibian Species of Special Concern tended

to be more evenly distributed across northern and southern California ecoregions.

In a troublingly large number of cases, we found a striking lack of critical data for many aspects of the basic biology of amphibian and reptile species, and this lack of field ecology, natural history, and genetic data hindered our ability to make strong management recommendations. The solution to this lack of data is clear: California needs to launch a program that funds strong, peer-review quality analyses of basic ecology, combined with long-term monitoring studies to evaluate demographic trends at a set of sites for each species. Such studies need not be expensive and would make an enormous difference in our ability to manage many Species of Special Concern, hopefully precluding the need for future state and/or federal listing. Meaningful collaboration between the California Department of Fish and Wildlife and other research groups (be they other agencies, universities, nongovernmental organizations, or avocational groups) has helped to fill some of these gaps, particularly for federally listed species, and such collaborations for Species of Special Concern are the key to developing management plans into the future. We also found that in many cases population genetic approaches can help to fill critical gaps in our knowledge regarding species and subspecies boundaries, effective population sizes, corridors of likely habitat use, migration frequencies and pathways, and levels of hybridization with native and introduced species. These genetic measures should complement, rather than replace field studies, and they offer the opportunity to conduct relatively fast analyses that can and should provide critical early guidance for management decisions.

As critical basic biodiversity work in California continues, we are increasingly recognizing that the complex geology and changing environmental conditions in the state have led to the evolution of an amazing array of endemic taxa, many of which are extreme habitat specialists. To our knowledge, none of these sensitive species have been lost to extinction yet, although several are dangerously close. However, at least four taxa whose range limits historically entered the margins of the state may already be gone from California's boundaries, and some of the endemic species may be next. The identification of Species of Special Concern and the compilation of information, research needs, and management recommendations represents an important step to help California land managers prevent further declines, stabilize key populations, and potentially initiate recovery programs before formal listing is necessary.

INTRODUCTION

From a biodiversity perspective, California resides at one of the most important crossroads in the United States. The California Floristic Province is the only globally recognized biodiversity hot spot in North America north of Mexico, and one of three recognized in the north-temperate region (Myers et al. 2000). With a 2010 population of more than 37 million people, California accounts for roughly one-eighth of the human population of the United States (US Census Bureau 2013), has the largest agricultural production of any state in the country (USDA 2007), and has one of the highest average land values in the nation (Davis and Heathcote 2007). Conserving biodiversity in California is therefore both enormously important and extremely difficult from an economic and political standpoint and requires strong scientific guidance and the collective will of multiple stakeholder groups.

Formal species protection in California is accomplished via the California Endangered Species Act and/or the Federal Endangered Species Act. The California Department of Fish and Wildlife (CDFW) is responsible for implementing the latter. As of January 2014, over 150 animals in our state were listed as threatened or endangered under either one or both acts. To help preclude the need to list additional species, the CDFW administratively designates Species of Special Concern. The intent of designating Species of Special Concern is to (1) focus attention on animals at conservation risk by the CDFW, other state, local, and federal governmental entities, regulators, land managers, planners, consulting biologists, and others; (2) stimulate needed research on poorly known species; and (3) achieve conservation and recovery of these animals before they meet California Endangered Species Act criteria for listing as threatened or endangered. Species of Special Concern carry no formal legal status but are widely viewed as one of the important front lines in species conservation planning and management. Regardless of the stakeholder group involved, whether members of the conservation, agricultural, or urban development communities, it is in everyone's best interest to maintain stable populations of Species of Special Concern to avoid the need for formal listing.

The Species of Special Concern designation is used to promote conservation in various ways by the CDFW, land managers, and others to promote conservation. For example, Species of Special Concern are considered "Species of Greatest Conservation Need" in California's Wildlife Action Plan (Bunn et al. 2007, http://www.wildlife.ca.gov/SWAP). State Wildlife Action Plans outline the steps needed to conserve these taxa before they become rarer and more costly to protect and provide access to funds for this purpose. Species of Special Concern are also considered when evaluating environmental impacts under the California Environmental Quality Act (California Public Resources Code Sections 21000-21177). The California Environmental Quality Act requires state agencies, local governments, and special districts to evaluate and disclose impacts to wildlife and habitat from proposed projects. Specifically, Species of Special Concern may meet the definitions of endangered, rare, and/or threatened in Section 15380 of the California Environmental Quality Act guidelines. Also, Section 15065 relates to the standards under which the lead agency determines if impacts to biological resources should be considered significant. Impacts to Species of Special Concern are generally considered significant if they are based on factors such as population-level effects, proportion of the taxon's range affected by the project, and effects on habitat. Environmental impact reports that analyze and evaluate the potential impacts on Species of Special Concern caused by the proposed project must be prepared before planned projects can move forward. Large-scale planning efforts, such as Habitat Conservation Plans and Natural Community Conservation Plans, also may include conservation measures for non-listed, at-risk species including Species of Special Concern. In addition, Species of Special Concern are tracked by the California Natural Diversity Database (http://www.dfg.ca.gov/biogeodata/cnddb), an important source of information on species distribution. Federal land management agencies like the Bureau of Land Management and US Forest Service often add Species of Special Concern to their sensitive species lists to focus attention on these taxa. In all, the Species of Special Concern designation results in a greater depth of knowledge about species as well as proactive conservation aimed at maintaining or restoring populations to avoid the need for future, formal listing.

In this volume, we update and evaluate the original Species of Special Concern document for amphibians and reptiles (Jennings and Hayes 1994a). The first Species of Special Concern document compiled was for birds (Remsen 1978). Over the following three decades, documents have been published or updated for birds (Shuford and Gardali 2008), mammals (Williams 1986; Bolster 1998), and fishes (Moyle et al. 1989, Moyle et al. 1995). As these documents have matured and been revised, so too have the methods by which Species of Special Concern have been identified from the potential pool of candidate taxa. With the exception of the 2008 bird publication, previous iterations of these assessments were largely based on expert opinion. A list of native California taxa was assembled, screened for risk potential, and evaluated by a small team of experts (usually in consultation with many additional experts throughout the state). The most at-risk taxa not already listed under the California Endangered Species Act were then selected as Species of Special Concern.

The Species of Special Concern assessment process changed profoundly with the 2008 bird publication (Shuford and Gardali 2008). A key change, and one that we also follow here, was to formalize the criteria by which species receive this designation. Following Shuford and Gardali (2008) and current CDFW standards (http://www.dfg.ca.gov/wildlife/nongame/ssc/index.html), we created a set of eight metrics that capture the extent to which an amphibian or reptile species is at risk of extinction in California. We used this system to increase transparency, facilitate clear feedback from a broad group of individuals on our scoring, and enhance the ability of the CDFW and other agencies to replicate this process in the future. We then ranked all species by their summed metric scores, presented that ranking to a wide-ranging group of experts, and determined inclusion or exclusion from the special concern list. This approach provided a clear connection between data and ranking, and an explicit description of the most important factors contributing to ongoing declines. It also provided a strong connection between the evaluation process for different taxonomic groups and therefore greater uniformity in the methodology used among all CDFW Species of Special Concern publications.

The current volume is divided into two sections. In Part I (this section), we provide a detailed description of our methods, including the metrics and their scoring, outreach strategies for public input, locality mapping, and the roles of different contributors in producing the set of Species of Special Concern taxa. Following this is an overview of the results of our review and several quantitative descriptions of geographical, ecological, and taxonomic patterns of Species of Special Concern. We end with a discussion of the results and present recommendations for the conservation of amphibian and reptile Species of Special Concern in California. Throughout, we emphasize immediate research needs, both for particular species and for broader assemblages and landscapes within the state. Part II consists of a series of species accounts that provide a synopsis of information for each Species of Special Concern. Each account also includes a map documenting localities where the species has been collected or observed along with a depiction of its current range.

Throughout this document, we have used the peer-reviewed literature as our primary source of information and have included unpublished reports, web sites, and data from the field notes of professional and avocational herpetologists to fill in gaps in the primary literature. We rely primarily on the peer-reviewed literature because it has been evaluated by independent experts and deemed admissible into the scientific literature. However, we also recognize that the published literature for many species is sparse, and in those cases we also evaluated and included a large amount of unpublished information. Finally, we particularly emphasized the more recent, post-1990 literature, given the extensive review by Jennings and Hayes (1994a) of the earlier literature.

METHODS

Overview of Project Design and Process

The process of developing this document involved cooperation among several groups. The initial study design was developed collaboratively between the CDFW and the authors (Thomson, Wright, and Shaffer). We then assembled a Technical Advisory Committee comprising members with broad geographical and taxonomic expertise in California's amphibian and reptile fauna. This group developed the set of metrics used in evaluating potential Species of Special Concern, as well as a standardized format for species accounts. We then reached out to all segments of the herpetology community, including academics, land

and resource managers, avocational herpetologists, and the interested public for further information, feedback, and review at various points in the process. Our goal throughout was to keep our actions and decisions transparent and accessible to anyone with an interest in herpetological conservation in California.

We began by developing a current list of all native amphibian and reptile species and subspecies known to occur in the state (Appendix 1). Based on the broad knowledge of field herpetology represented by the authors and the Technical Advisory Committee, we used this list to develop a set of Special Concern nominees. Our goal was to include in this nominee list all taxa that anyone felt were declining or in need of protection in the state. The authors conducted preliminary reviews of each of these taxa, searching the literature and interviewing experts, and used these data to produce a set of preliminary scores for each of the nominees using the risk metrics. These scores were reviewed and refined by the Technical Advisory Committee and then further reviewed and refined based on input from the herpetological community at large. The authors and Technical Advisory Committee used the metric scores, as explained later in this document, to construct a set of taxa for inclusion as Species of Special Concern. After the list was finalized, we produced species accounts for each of the Species of Special Concern.

During this evaluative process, we compiled locality information for each taxon, which we then combined with data from the California Natural Diversity Database and Biogeographic Observation and Information System to produce distribution maps for each nominee species. The Technical Advisory Committee, the CDFW, and other experts reviewed these range maps, resulting in the maps in this document.

Species List, Taxonomy, and Units of Conservation

We developed our species list by compiling information from existing taxonomic lists and recent taxonomic literature. We included all recognized or proposed species, subspecies, and distinct population lineages that have been identified. We generally used the most recent revisionary studies, although we sometimes made decisions based on the degree to which the scientific community had accepted proposed changes and the quality and strength of data informing proposed revisions. Little consensus exists on taxonomy for certain groups (e.g., California mountain kingsnake, *Lampropeltis zonata*), and we tried to strike a balance between incorporating the most current, reliable information while also maintaining taxonomic stability in the face of current uncertainty. For example, Frost et al. (2006a) proposed a large number of taxonomic changes for California amphibians, often shifting species into new generic name combinations (e.g., the western toad, *Bufo boreas*, changes to *Anaxyrus boreas* under this scheme). These changes have been vigorously debated (Crother et al. 2009, Frost et al. 2009a, Pauly et al. 2009), and we have taken the conservative approach of retaining the traditional nomenclature.

We focused our evaluation primarily at the species level, although we also considered subspecies and (rarely) parts of an otherwise stable species range that appeared to be in decline. This follows most similar efforts to date in recognizing species as the fundamental units of conservation, while still acknowledging that significant diversity exists and should be maintained within species. This also allowed us to limit the extent to which taxonomic controversy might negatively impact important conservation efforts. For example, if we were to consider only species (or formally described subspecies), we would fail to consider currently unnamed populations in need of conservation action. The southern populations of the common garter snake (*Thamnophis sirtalis*) are an example of such a population, as are the southern populations of the Coast Range newt (*Taricha torosa*). Throughout this document we use the term “taxa” to refer to species, subspecies, or distinct populations.

Development of the Nominee List

The first stage in the process was to develop a list of nominee Species of Special Concern from the comprehensive list of taxa that occur in the state. We included all taxa from the previous amphibian and reptile Species of Special Concern document (Jennings and Hayes 1994a), those that were recently extirpated or possibly extirpated from the state, and all taxa currently listed under the Federal Endangered Species Act. We excluded any taxa that were already legally designated by the state (i.e., Endangered or Threatened under the California Endangered Species Act), because Species of Special Concern status would provide no further state-level protections. Although federally listed taxa also experience a higher level of protection than Species of Special Concern, we still considered them in the evaluation process because federal status could potentially be the result of conservation needs from parts of the species' range outside of California. Because of this, an assessment of each species focusing on its California range provides information about its status within the state.

We included additional nominee taxa that members of the Technical Advisory Committee identified as potentially at risk based on their experience with that taxon in the field. If at least one member of the committee suspected that a taxon might qualify as a Species of Special Concern, we included it for evaluation. Additional taxa were added through consultation with experts on specific species or larger taxonomic groups and by suggestion during the public comment phase of the project (see below). We then evaluated these taxa with the risk metrics and used the resulting scores as our primary basis for Species of Special Concern determination (see below).

Definition of Species of Special Concern

We define a Species of Special Concern as any native species, subspecies, or distinct population of amphibian or reptile occurring in the state that currently meets one or more of the following criteria (see also Comrack et al. 2008):

- Is extirpated from the state within the recent past;
- Is listed as federally, but not state, Threatened or Endangered and/or meets the state definition of Threatened or Endangered but has not formally been listed;
- Is experiencing, or formerly experienced, serious, noncyclical, population declines or range retractions that, if continued or resumed, could qualify it for state Threatened or Endangered status;
- Has naturally small populations and/or range size and exhibits high susceptibility to risk from any factor(s) that, if realized, could lead to declines that would qualify it for state Threatened or Endangered status.

We developed a set of risk metrics to address the latter two criteria. Taxa scoring high on these risk metrics were then judged to be prime candidates for inclusion on the list. Taxa meeting the first two criteria were included automatically. All taxa were scored for the risk metrics and included in our quantitative analyses.

Risk Metrics

Working with the Technical Advisory Committee and using CDFW criteria (http://www.dfg.ca.gov/wildlife/nongame/ssc/index.html), we developed a set of conservation risk metrics to quantify the level of threat to California's at-risk amphibians and reptiles. Although quantification of conservation risk is necessarily approximate, the metric approach allows for improved repeatability between Species of Special Concern updates and a framework for discussion and revision. Earlier Species of Special Concern documents were based largely on expert opinion and the use of risk metrics does not completely eliminate this important element of the assessment process. Rather, the risk metrics place expert opinion, as well as data, within a standardized framework that makes decisions more transparent. For example, our ecological tolerance metric provides a clear definition of how we quantified the ecological specialization of each taxon and how it relates to conservation risk. If, at a later time, additional data become available or other workers disagree with our interpretation of the existing data, there now exists a clear way in which this new information can be incorporated into the overall score for any species.

The possible score for each metric ranged from 0 (little or no risk) up to a maximum of 25 (high risk), reflecting the relative importance of the risk quantified by that metric. We weighted metrics that measure *documented* conservation concerns, such as declines in abundance, more highly than other metrics that focused on *potential* conservation concerns, such as life history factors that contribute to sensitivity. We did this for two reasons. First, our weighting reflects the emphasis on these factors in the definition of Species of Special Concern. Second, documented conservation concerns usually require more immediate management action and are likely more serious threats to survival than potential conservation concerns. The result of this decision is that some metrics, such as those measuring declines in distribution or abundance, affected the overall risk metric score more than, for example, a naturally small range size. The eight risk metrics are as follows.

I. *Range Size*

The range size metric estimates the percentage of California that each taxon occupies. Though this measure could be treated as continuous, we have approximated it with discrete categories for two reasons. First, we have little biological reason to believe that a taxon that occupies, for example, 35% of California is under any greater conservation risk than a taxon that occupies 42%. Both of these hypothetical taxa occupy moderate portions of the state and probably experience similar risk arising from the size of their range. Second, there is inherent uncertainty in many amphibian and reptile range predictions as portrayed in range maps, and we felt that it was more appropriate to broadly categorize ranges rather than attempt to precisely estimate them. We therefore categorize range size as *small*, which includes those taxa that are at immediate risk from relatively small scale disturbances; *medium*, which includes taxa that occupy a portion of the state that is big enough so that a single large catastrophic event would be unlikely to affect the entire range; and *large*, which includes those taxa that occupy such a large portion of the state that range size itself is unlikely to have any significant impact on threat. Patchiness and ecological specialization of species that limit range on a local scale are quantified in other metrics. Our aim for this metric is only to estimate the actual size of the species range. In the few cases where the known range is strictly limited by habitat specialization or limitation (e.g., desert populations of the regal ring-necked snake, *Diadophis punctatus regalis*, or the Gila monster, *Heloderma suspectum*) and the taxon almost certainly does not occur between isolated habitat patches, we treated the known populations as individual polygons in scoring this metric.

(I) RANGE SIZE (% OF CALIFORNIA OCCUPIED)	SCORE
Small (<10%)	10
Medium (10–50%)	5
Large (>50%)	0

II. *Distribution Trend*

The distribution trend metric aims to quantify documented decreases in the overall range of each taxon based on extirpation of previously known localities. The total score for this metric comes from two sources. First, we attempted to quantify the extent of known range reductions, scoring them using the categories below. We classified the extent of range reduction into discrete categories for similar reasons as range size. We then added an additional 5 points if the documented reduction in range appears to have been ongoing since the last Species of Special Concern document was published (Jennings and Hayes 1994a) and has not yet stabilized or reversed. We did this to increase the weight of declines that are continuing at present, and which therefore are likely to continue in the immediate future. As a result, a species might attain a particular score through either a documented reduction or a less severe reduction that is ongoing. In scoring this metric, we used peer-reviewed published data whenever possible. The best data for this metric came from repeated field surveys of habitat through time, and we used them whenever they were available. However, datasets of this type are, at present, uncommonly available for amphibian and reptiles of California.

(II) DISTRIBUTION TREND	SCORE
Severely (>80%) reduced	20
Greatly (>40–80%) reduced	15
Moderately (>20–40%) reduced	10
Slightly (<20%) reduced or suspected of having been reduced but trend unknown	5
Stable (~0% reduced) or increasing	0

Add 5 additional points if negative trend is ongoing for a total of 25 points possible for this metric.

III. Population Concentration/Migration

This metric focuses on whether features of the life history of individual taxa, such as migration events or aggregations, make them naturally vulnerable to decline or extirpation. For instance, taxa that migrate to breed in ponds are exposed to additional risk during the migration itself (e.g., road crossings) as well as increased risk while concentrated in the breeding habitat. This latter risk could come about if a catastrophic event occurs during the breeding concentration (e.g., if a toxic spill or group of predators killed the breeding animals) or because the actual breeding site is destroyed (e.g., draining of the aquatic breeding habitat). We score this trait either *present* or *absent* based on the available life history data for each taxon.

(III) POPULATION CONCENTRATION/MIGRATION	SCORE
Vulnerable life stages present	10
No vulnerable life stages	0

IV. Endemism

The endemism metric captures the percentage of a species' entire range that occurs in California. Endemism determines the extent to which conservation actions in California are likely to impact the taxon's persistence range-wide. From another perspective, this is a way of measuring California's responsibility to conserve individual species. Taxa whose range is completely, or nearly completely, contained within California's borders are in need of greater conservation consideration from our state than taxa whose range only extends peripherally into California. We recognize that this presumes appropriate conservation measures are also being implemented in other areas of North America (including Mexico and Canada). We again made this measure discrete in recognition of the inherent uncertainty in our knowledge of range limits.

(IV) ENDEMISM (% OF ENTIRE RANGE IN CALIFORNIA)	SCORE
100% (endemic)	10
>66–99%	7
33–66%	3
<33%	0

V. Ecological Tolerance

This metric measures ecological specialization. Species that are narrow specialists on specific ecological resources (such as habitat, prey, temperature regimes) are inherently more sensitive to ecological disturbance than species that can tolerate a wider range of ecological conditions. In addition to the degree of specialization, we also considered the extent to which the resource that each taxon specializes on is common or rare. For instance, several saxicolous (rock loving) lizard species (e.g., the leaf-toed gecko, *Phyllodactylus nocticolus*) use rocky habitats that occur throughout extensive areas of the species' total range. We scored cases like this as specialists on a common resource. Conversely, vernal pool breeding amphibians (e.g., Couch's spadefoot, *Scaphiopus couchii*) require temporary aquatic pools that are rare throughout their range for successful breeding. We scored these taxa as specialists on a rare resource. We adjusted the rareness of the resource with respect to its availability within the species' range, rather than its availability within the state.

(V) ECOLOGICAL TOLERANCE	SCORE
Narrow ecological specialist on a rare resource	10
Narrow ecological specialist on a common resource	7
Moderate ecological specialist	3
Broad ecological tolerance	0

VI. *Population Trend*

The population trend metric captures changes in abundance at localized, population-level sites. This is distinct from the distribution trend, which measures extirpation of localities; population trend captures declining abundances at localities that are not extirpated. In many cases, distributional declines as measured by distribution trend will be associated with earlier declines as measured by population trend. This raises the potential of scoring taxa twice for the same decline. To avoid this, we scored population declines that have led to extirpation under the distribution trend metric. We gave those same taxa high scores for the population trend metric only if additional population declines have been documented at currently extant sites. We scored population trend in the same way as distribution trend, first scoring the extent of the decline and then adding an additional 5 points if evidence suggests that the trend is ongoing. As a result, a species might attain a particular score through either a documented reduction or a less severe reduction that is ongoing.

(VI) POPULATION TREND	SCORE
Severe declines (>80% reduced)	20
Great declines (>40–80% reduced)	15
Moderate declines (20–40%)	10
Slight (<20%) or suspected declines	5
Stable (~0% reduced) or increasing	0
Add 5 additional points if declines are ongoing.	

VII. *Vulnerability to Climate Change*

The climate change metric measures a taxon's sensitivity to the projected effects of climate change. We scored this metric using the projected impacts on California landscapes based on the California Climate Action Team assessments (Cayan et al. 2008a), followed by our interpretations of how these impacts are likely to affect each taxon based on life history and habitat requirements. For example, climate projections suggest that snowpack in the Sierra Nevada is likely to decrease by 30–90% (depending on carbon emissions and the climate model used) over the next 100 years, leading to a narrower window of time over which the spring snowmelt will occur (Maurer and Duffy 2005, Cayan et al. 2006, Maurer 2007). This is likely to have an impact on the snowmelt-dependent aquatic habitats that many Sierran amphibians use for one or more life stages, and may also reduce the time period over which moist microhabitats will occur in forest ecosystems. Other impacts that we considered for this metric included changing hydrology (amount and variation of precipitation), temperature, wildfire frequency and intensity, and changes in the extent of habitat and vegetation types. Given our imprecise knowledge of both future climate change effects and their impacts on species, we discretized this impact into four broad categories.

(VII) VULNERABILITY TO CLIMATE CHANGE	SCORE
Highly sensitive	10
Moderately sensitive	7
Slightly sensitive	3
Unlikely to be sensitive	0

viii. Projected Impacts

The projected impacts metric estimates the effect that future threats may have on each species over the near term (20 years). It does not incorporate threats arising from changing climate, because these are captured in a separate metric. This includes impacts stemming from known threats, such as planned or projected habitat loss and, to a lesser extent, impacts from irregularly occurring threats, such as disease outbreaks. Given the potential for these risks to be reduced by management, plus the inherent uncertainty associated with complex projections, we considered potential threats to be of relatively less importance than documented threats such as population declines.

(VIII) PROJECTED IMPACTS (OF THREATS OVER THE NEXT 20 YEARS)	SCORE
Serious	10
Moderate	7
Slight	3
No substantial impact	0

Scoring Nominee Taxa

We scored all of the nominee taxa for each of the eight metrics based on the best available evidence. To begin with, the primary authors produced a brief summary of the state of conservation knowledge for each nominee taxon and used these summaries to perform a preliminary scoring assessment. In making these assessments, we included the peer-reviewed literature, unpublished reports, survey data, field notes, and the opinions of knowledgeable biologists. In several cases, few data were available to make assessments for a given metric. In these cases, if the data appeared to be strong enough to clearly indicate that a threat was present, we scored that taxon using the most precise estimate that we were able to make. In cases where no data were available or the limited data were ambiguous, we scored taxa as "data deficient" for that metric. Following these preliminary assessments, we circulated all of the scores and taxon summaries to the Technical Advisory Committee for review and further input. In the rare cases of substantial disagreement, we discussed the issue and evaluated the data as a group, and reached a consensus on the most reasonable score for a given taxon.

After this preliminary scoring process was complete, we created an overall score for each taxon by summing its metric scores and dividing by the total score possible for that taxon (Total Score/Total Possible). Using the ratio of total score to total possible score allowed us to normalize the scores across varying levels of data deficiencies. For example, in cases where a taxon was scored as data deficient for one or more metrics, the total possible score was lower than would be the case if all metrics had been scored. This would result in a lower risk assessment due to uncertainty as opposed to data, and we used standardization by the Total Possible score in order to focus on documented risks.

Public Comment

After the scoring assessments were complete, we opened a 60-day public comment period by posting all of our initial findings on the project's website and sought input widely on herpetological and conservation-oriented email lists and websites (Appendix 2). We requested comments and feedback on the initial set of scores, additional data that could inform the scoring (particularly for the metrics that had been scored as data deficient), and feedback on the process to date. When individuals suggested changes to the metric scores, we asked for a short explanation of what should be changed and why, along with any data and/or field notes that were available to support the proposed change. At the close of the public comment period, we compiled and evaluated all of the information that we received (see Results, Public comment). We evaluated each proposed change on a case-by-case basis, usually making the change if it was reasonable, supported by information (in the form of unpublished reports, data, or field notes), and not in strong conflict with other existing data. In cases where a suggested change was in strong conflict with other data, we asked that the contributor supply additional data justifying their viewpoint and made a decision on the final resolution of any conflicting information.

We also asked that contributors send additional data that could be incorporated into the locality maps (see below). To facilitate this process, we supplied a standardized data sheet similar to that used for data submission to the California Natural Diversity Database. These localities were added to the California Natural Diversity Database and to our set of existing localities, and they were used in developing range maps.

Ranking and Determination of Species of Special Concern Status

After incorporating the information received during the public comment period, we worked with the Technical Advisory Committee to develop the set of Species of Special Concern taxa. Taxa with the highest scores were included on the list, while those with intermediate scores were evaluated on a case-by-case basis; this combined approach was similar to that used in the Bird Species of Special Concern (Shuford and Gardali 2008). Specifically, taxa that had intermediate scores but had a combination of exceedingly small range size, extreme ecological specialization, and high projected impacts were included as Species of Special Concern. In essence, this approach weights the combination of these factors more heavily in order to meet the last of the four criteria for inclusion as a Species of Special Concern, "small populations and/or range size and exhibits high susceptibility to risk from any factor(s), that if realized, could lead to declines that would qualify it for state Threatened or Endangered status" (Comrack et al. 2008).

We further ranked Species of Special Concern into three priority categories based on the severity and immediacy of threats affecting each taxon. Priority 1 Species of Special Concern are those taxa that are likely to experience severe future declines and/or extirpation without immediate conservation actions. Priority 2 Species of Special Concern require substantial conservation and management actions, although the threats facing them are less immediate and severe than those in Priority 1. Finally, Priority 3 Species of Special Concern are clearly at risk but likely are not experiencing a substantial and immediate threat of extirpation, although the potential for this threat to develop exists if no management actions are undertaken. One of the primary goals of the Species of Special Concern designation is to identify taxa for which managers can undertake relatively small scale and achievable conservation actions that will negate the need for more costly and serious listings at a later date. Priority 3 taxa are prime candidates for such efforts.

Watch List and Additional Taxa in Need of Research and Monitoring

Taxa that were previously considered Species of Special Concern but are no longer included comprise a Watch List (Appendix 3). Appendix 3 includes an explanation for each taxon's change in status and discusses future conservation concerns regarding Watch List taxa. In Appendix 4, we discuss several other taxa in need of research and monitoring that did not warrant inclusion as Species of Special Concern. Some of these were taxa that had scores indicating a lower, but still substantial, amount of risk. Although we decided that they were at a lower priority than the Priority 3 Species of Special Concern and therefore should not be so designated, they formed a group of species to reevaluate in the future. We were also missing important information for some taxa that would have allowed us to make more informed judgments about conservation status. We devote a paragraph to each of these additional taxa in need of research and monitoring in Appendix 4, briefly describing the threats facing each and outlining research and management needs.

Species Accounts

We prepared a species account for each Species of Special Concern that summarized our findings and the relevant aspects of the taxon's biology. We also provided management and research recommendations for each taxon. These accounts follow a standardized format containing each of the following sections.

Status summary. The status summary is a short explanation of each animal's current and former status as a California Species of Special Concern, including its priority level. In the first version of the Amphibian and Reptile Species of Special Concern monograph, Jennings and Hayes (1994a) categorized each taxon according to whether they felt it was a Species of Special Concern or met the criteria for listing as Threatened or Endangered under the California Endangered Species Act. However, this strategy led to some potential confusion because the Jennings and Hayes (1994a) Threatened and Endangered categories did not correspond to actual state listing categories, nor had taxa they described as Threatened or Endangered undergone the rigorous status evaluation required to assess status under the California Endangered Species Act. To avoid this confusion, we used Priority categories (1, 2, or 3) to convey similar information on relative severity of threat as represented in the ranking of Species of Special Concern. This section also contains the overall metric score.

Identification. The identification section summarizes and explains the diagnostic characters for each animal, providing a guide for identifying it in the field. This section also explains how to differentiate each taxon from similar species with which it may be confused. Several taxa within the state are members of morphologically similar species complexes that have been identified primarily based on molecular data. In some of these cases, accurate identifications using morphological characters alone are difficult or impossible, and we generally recommend that biologists rely on geographic range. We also provide references to the taxonomic literature to guide the reader to the more thorough and technical descriptions of morphology that are beyond the scope of this document.

Taxonomic relationships. In addition to identification information, we provide a summary of the taxonomic status of each animal. This section contains information on current controversies over scientific names, at either the species or higher taxonomic levels. It also summarizes our current understanding of phylogenetic relationships, intraspecific variation, and species boundaries among closely related taxa.

Life history. This section summarizes the current state of knowledge for each taxon's life history, which broadly includes ecology, natural history, and breeding biology. As an exhaustive review of life history information would be enormous for some taxa, we focused on information that is most relevant to current and future management actions and to the risk metrics. Specifically, we concentrated on information that relates to timing and duration of reproductive activity, daily and seasonal activity, and dietary information. Because management efforts for many taxa could be greatly enhanced by a better understanding of life history, we attempted to point out the areas that require further study rather than speculating about the details of life history where the data are weak. We emphasized data from California populations, but used data from other areas of the range or similar species when those were the best available data. We note when we used data from non-California populations and why we believed that the data could be accurately applied.

Habitat requirements. This section focused on the current state of knowledge concerning habitat use, preferences, and requirements. We attempted to distinguish between habitat *preferences*, the habitats in which the taxon is most frequently found, and habitat *requirements*, which are the characteristics of the habitat that the taxon requires for survival over long timescales.

Distribution. This section describes each animal's current distribution and makes an assessment of changes that have occurred throughout its documented history in the state. We focused primarily on the known distribution within the state, although we also discussed the distribution outside of California if applicable. Finally, to stimulate additional fieldwork, we point out areas where the distribution is poorly known or there is a high probability of significant new localities being discovered.

Trends in abundance. This section reviews information relating to changes in abundance throughout each taxon's documented history. For current population status, we used quantitative population-level analyses where available. However, these kinds of data are rare. Historical data tend to be spotty and incomplete for amphibians and reptiles, and much of the historical information comes from nonquantitative sources, including field reports and personal communications from experienced field biologists.

Nature and degree of threat. This section contains a detailed description of the principal threats that each taxon faces. We highlighted both the nature and severity of different threat sources, while discussing any uncertainty and conflicting data in the literature associated with these threats. We evaluated the weight of evidence and discussed what threats might be playing the largest role(s) in causing declines.

Status determination. This section connects the information on different sources of threat to the metric scores and Species of Special Concern priority categories. We explained the rationale for our determination and the seriousness of the different major threats facing each taxon.

Management recommendations. This section makes recommendations aimed at achieving sound, biologically based management and status improvement for each Species of Special Concern. Wherever possible, we made these recommendations both taxon-specific and action-oriented to allow conservation resources to be put directly into management efforts, rather than into further development of management strategies. We did, however, recommend further research and strategy development as a prerequisite to effective management for taxa that lacked necessary data.

Monitoring, research, and survey needs. This section outlines the additional information necessary to achieve effective management and status improvement. In general, information needed to inform management actions falls into the general areas of monitoring, research, or surveys, and we discuss each as appropriate.

Maps. We developed locality maps to complement the distribution information in the text for each taxon by compiling data from museum collections, state agency databases (e.g., California Natural Diversity Database), and other online databases (e.g., North American Field Herping Association) (Table 1). Data from the CDFW's California Natural Diversity Database and the Biogeographic Information and Observation System were assessed up through April 2012. Museum locality data from HerpNet and the Global Biodiversity Information Facility were assessed through February 2012. Our goal was to develop a set of annotated and geo-referenced localities that accurately describe each taxon's range. Records that appeared to be possibly erroneous (i.e., those that occurred in unexpected areas) were checked individually and excluded in those instances where no supporting information could be found or where the specimens were misidentified (see individual species accounts). We attempted to verify all records coming from online databases and the public by requesting, minimally, photo vouchers or detailed field notes to substantiate the record. The California Natural Diversity Database contains localities that lack this information, so we followed up on questionable records by attempting to contact the individual(s) that initially reported the record. We submitted most new localities that we gathered to the California Natural Diversity Database to make them available for future workers. In a few cases, we could not obtain permission to include localities in the database, so these were included in the maps in this volume, but

TABLE 1

List of museum collections and other data sources that were queried for locality records

Museum Collections	Other Sources
American Museum of Natural History	Cal Photos
Arizona State University	California Biogeographic Information and Observation System
Brigham Young University	California Natural Diversity Database
California Academy of Sciences	Field Notes
California Academy of Sciences, Stanford University Collection	Literature Records
California State University, Chico	Mendocino Redwood Company
Carnegie Museum of Natural History	North American Field Herping Association
Cincinnati Museum Center	Our Own Surveys
Cornell University Museum of Vertebrates	Public Input/Personal Communications
Humboldt State University	US Forest Service
Los Angeles County Museum	US Geological Survey
Museum of Comparative Zoology, Harvard University	
Museum of Vertebrate Zoology, University of California, Berkeley	
National Museum of Natural History	
Royal Ontario Museum	
San Diego Natural History Museum	
Santa Barbara Museum of Natural History	
Slater Museum of Natural History	
Sternberg Museum of Natural History	
University of Alberta Museum of Zoology	
University of Arizona Museum of Natural History	
University of California, Davis – Zoology Collection	
University of California, Santa Barbara	
University of Colorado Museum of Natural History	
University of Michigan Museum of Zoology	
University of Nevada Reno	
University of Texas at El Paso	
Yale Peabody Museum	
Zoological Institute of the Russian Academy of Sciences	

excluded from the database. The complete geospatial dataset and associated metadata from this project are accessioned in the CDFW's Biogeographic Information and Observation System (BIOSds644).

After removing erroneous and questionable records from the data, we developed point locality maps with our CDFW Geographic Information System specialist by projecting all localities for each taxon to the California (Teale) Albers projection (figure 1). We used the California Wildlife Habitat Relationships (http://www.dfg.ca.gov/biogeodata/cwhr) mapping protocol to develop range maps for each taxon using these localities. California Wildlife Habitat Relationships is a comprehensive information system for the state's terrestrial vertebrates that seeks to integrate data on species life history, habitat needs, and ranges.

To develop species range estimates, we selected the full set of US Department of Agriculture ecoregion subsections that contained at least one locality and used these as a starting point for range maps (figure 2). We then overlaid existing range maps from California Wildlife Habitat Relationships, as well as data layers for habitat types, watersheds, elevation, land use, and urbanization. Using these draft maps,

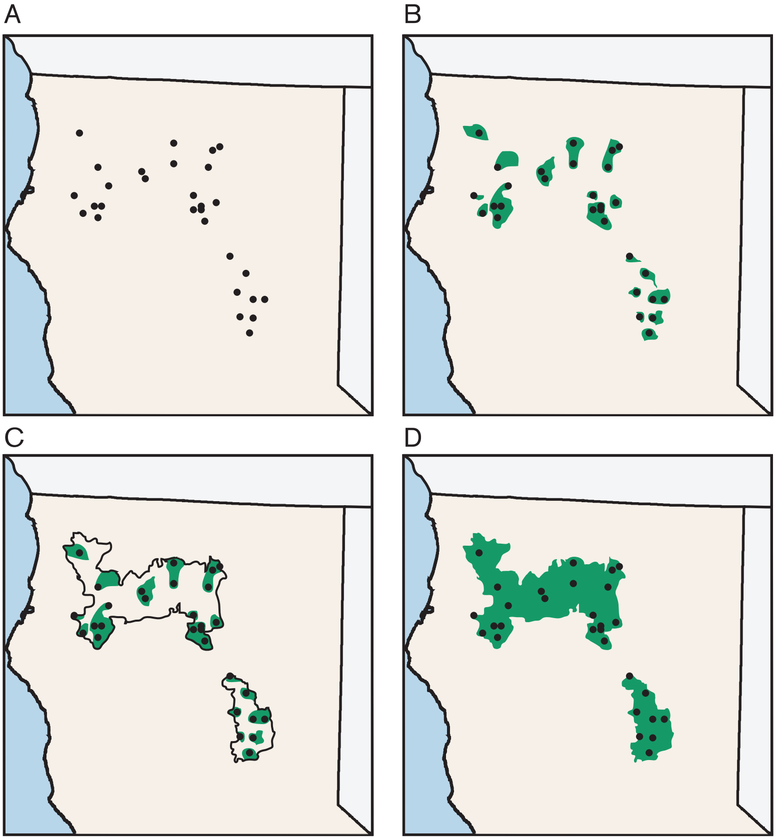


FIGURE 1 Development of range maps for each species. We began by plotting localities on a base map in a geographic information system (A). We then selected the intersection of these localities with an objective geographic object such as US Department of Agriculture (USDA) Ecoregion subsection boundaries, elevational boundaries, or watershed boundaries (in this example, watershed boundaries were used). The particular geographic object that we used varied according to the biology of the taxon (e.g., watershed boundaries for stream-dwelling amphibians, elevation for high-elevation taxa) (B). We then interpolated between the geographic objects that had known localities using expert opinion to develop an approximate range boundary (C). The approximate range boundary and known localities were then drawn together to produce a map for this document (D).

we restricted range boundaries based on ecoregion subsection, watersheds, and other data layers to a more biologically realistic species range. In accordance with the California Wildlife Habitat Relationships guidelines, our goal was to define the current maximum geographic extent of the species within the state, where maximum geographic extent is defined as the area within the range boundary where the species can potentially be expected to occur given suitable habitat conditions. We delineated the range boundaries to minimize errors of omission, even to the extent of allowing some commission error. For certain species, significant fractions of the range are potentially extirpated (see the species accounts for additional detail). No range shading is included for the species that are presumed extirpated in California (see individual species accounts).

In most cases, we defined the edges of species ranges by selecting meaningful landscape characteristics to set a boundary, such as elevation, rivers, or watershed boundaries. Our goal was to identify specific places on the landscape

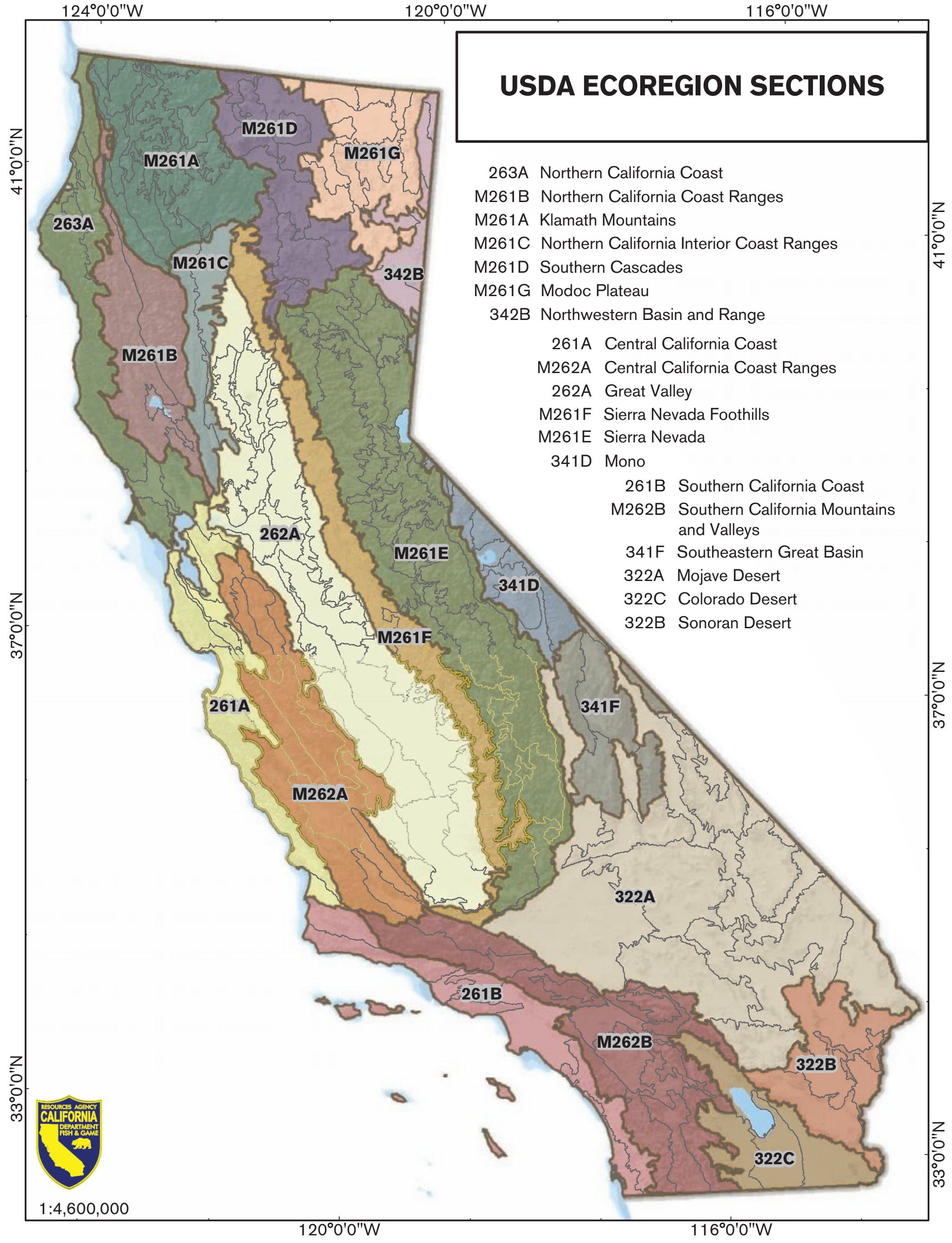


FIGURE 2 United States Department of Agriculture (USDA) Ecoregion subsections which were used in developing range maps.

where future surveys could be conducted to further characterize the species' range. Range maps that lack specific and objective boundaries provide only generalized starting points for such surveys. In total, our range maps present comprehensive estimates based on currently available species locality data and represent our best effort to use these data to approximate a species range, fully recognizing that such ranges are hypotheses to be tested rather than fixed entities.

Review Process

All phases of this project were reviewed by the three authors, the Technical Advisory Committee, and the CDFW. Most parts of the project were also subject to a wider review from members of the herpetological conservation community. For each taxon, we asked at least two experts to review the species account, including the maps and any appendix information. Finally, the Technical Advisory Committee, the CDFW, biologists from state and federal land management agencies, and other interested parties reviewed the finished manuscript as a whole.

RESULTS

Status Lists

We identified 217 native species, subspecies, and distinct population segments that are, or are suspected to be, present in California (Appendix 1). Seventy-three of these taxa were considered nominee Species of Special Concern and underwent evaluations using the risk metrics. Four additional taxa were initially considered for evaluation but were subsequently state listed and removed from further consideration (see Watch List). Of the 73 candidates, we determined that 28 did not merit special status at this time and 45 met our criteria for Species of Special Concern status (figures 3 and 4 and Table 2). Three of these species qualified for Species of Special Concern status by definition because they were listed under the Federal, but not the California, Endangered Species Act (the arroyo toad, *Bufo californicus*; the California red-legged frog, *Rana draytonii*; and the Yosemite toad, *B. canorus*). We conducted the scoring separately for the two subspecies of the western pond turtle (*Emys marmorata marmorata* and *E. m. pallida*) because the severity of threats facing one population appeared to be larger than those facing the other. However, both populations merited inclusion as Species of Special Concern, resulting in a single species account where threats to each population are discussed separately.

We ranked the Species of Special Concern taxa according to the magnitude of risks that they face, with the two pond turtle populations receiving separate Priority scores. This resulted in 16 taxa categorized as Priority 1, 14 as Priority 2, and 12 as Priority 3. Three additional species clearly qualify as Species of Special Concern, although the scarcity of field records precludes their accurate prioritization at this time: the regal ring-necked snake (*Diadophis punctatus regalis*), Cope's leopard lizard (*Gambelia copeii*), and the Gila monster (*Heloderma suspectum*). In these three cases, we have not assigned a priority score pending additional fieldwork.

Performance of Metrics

Spearman's rank correlations among the eight risk metrics indicated that approximately two-thirds (18/28) of the possible pairwise correlations among metrics were significant (Table 3). Some metrics were not highly correlated with other metrics (e.g., endemism was not correlated with any other metrics), while other metrics were correlated with four or five other metrics (e.g., distribution trend, population concentration/migration, and population trend). Some pairs of correlations indicated that there was considerable overlap in the scores received across taxa. The strongest correlation among metric scores was between distribution trend and population trend ($\rho = 0.66$, $p < 0.001$), indicating that animals that have

TABLE 2

List of California amphibian and reptile Species of Special Concern and priority designations

Three species qualify as Species of Special Concern, although the scarcity of data precludes their accurate prioritization at this time (see text for further discussion)

Scientific Name	Common Name	Priority
Ambystoma macrodactylum sigillatum	Southern long-toed salamander	2
Aneides flavipunctatus niger	Santa Cruz black salamander	3
Anniella pulchra	California legless lizard	2
Arizona elegans occidentalis	California glossy snake	1
Ascaphus truei	Coastal tailed frog	2
Aspidoscelis tigris stejnegeri	Coastal whiptail	2
Batrachoseps campi	Inyo Mountains salamander	3
Batrachoseps minor	Lesser slender salamander	1
Batrachoseps relictus	Relictual slender salamander	1
Bufo alvarius	Sonoran Desert toad	1
Bufo californicus	Arroyo toad	1
Bufo canorus	Yosemite toad	1
Coleonyx variegatus abbotti	San Diego banded gecko	3
Crotalus ruber	Red diamond rattlesnake	3
Diadophis punctatus regalis	Regal ring-necked snake	Undefined
Dicamptodon ensatus	California giant salamander	3
Elgaria panamintina	Panamint alligator lizard	3
Emys marmorata marmorata	Northern western pond turtle	3
Emys marmorata pallida	Southern western pond turtle	1
Gambelia copeii	Cope's leopard lizard	Undefined
Heloderma suspectum	Gila monster	Undefined
Kinosternon sonoriense	Sonora mud turtle	1
Masticophis flagellum ruddocki	San Joaquin coachwhip	2
Masticophis fuliginosus	Baja California coachwhip	3
Phrynosoma blainvillii	Coast horned lizard	2
Phrynosoma mcallii	Flat-tailed horned lizard	2
Rana aurora	Northern red-legged frog	2
Rana boylii	Foothill yellow-legged frog	1
Rana cascadae	Cascades frog	2
Rana draytonii	California red-legged frog	1
Rana pipiens	Northern leopard frog	1
Rana pretiosa	Oregon spotted frog	1
Rana yavapaiensis	Lowland leopard frog	1
Rhyacotriton variegatus	Southern torrent salamander	1
Salvadora hexalepis virgultea	Coast patch-nosed snake	2
Scaphiopus couchii	Couch's spadefoot	3
Spea hammondii	Western spadefoot	1
Taricha rivularis	Red-bellied newt	2

Taricha torosa, Southern populations	Coast range newt	2
Thamnophis hammondii	Two-striped garter snake	2
Thamnophis sirtalis, Southern populations	Common garter snake	1
Uma notata	Colorado Desert fringe-toed lizard	2
Uma scoparia	Mojave fringe-toed lizard	3
Xantusia gracilis	Sandstone night lizard	3
Xantusia vigilis sierrae	Sierra night lizard	3

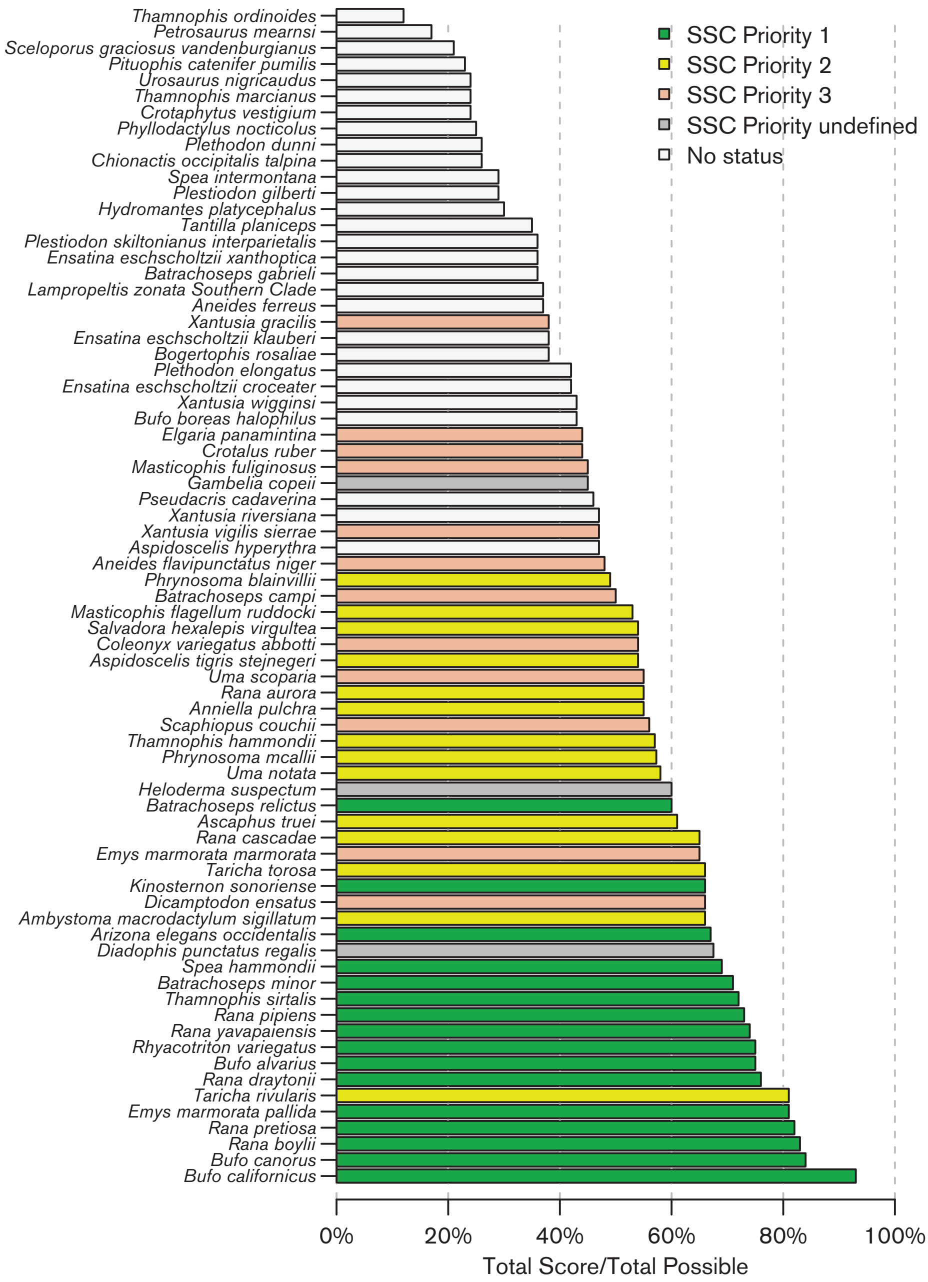


FIGURE 3 Total Score/Total Possible for 73 taxa evaluated for Species of Special Concern status.

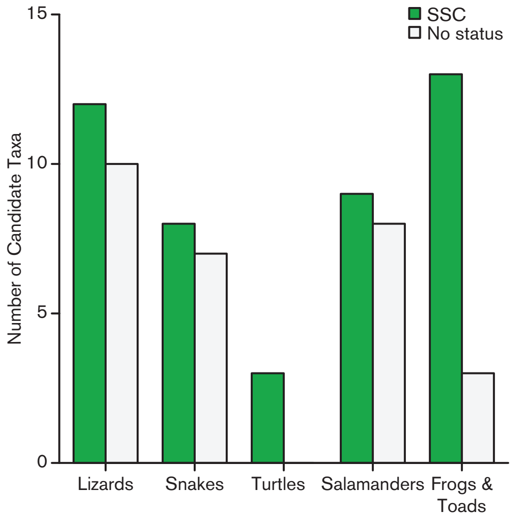


FIGURE 4 Number of taxa in each status category among the 73 nominee taxa by taxonomic group. Species of Special Concern (SSC) are represented by filled bars. Open bars are nominee taxa that did not receive SSC status.

been extirpated from historic localities tended to also be undergoing declines in abundance in currently occupied sites. Taxa experiencing high levels of extirpation also tended to have vulnerable life stages (correlation between distribution trend and population concentration/migration) and be more at risk from future threats (correlation between distribution trend and projected impacts). Those with vulnerable life stages also tended to be declining in abundance (correlation between population concentration/migration and population trend) and were more vulnerable to climate change (correlation between population concentration/migration and vulnerability to climate change).

All but two metrics (range size and endemism) were significantly positively correlated with Total Score/Total Possible (Table 3). Distribution trend and population trend were a priori given the greatest weight (each had a maximum score of 25 vs. a maximum score of 10 for all other metrics), and they were also the most highly correlated with Total Score/Total Possible ($\rho = 0.77$ and 0.87, respectively). Projected impacts, population concentration/migration, and vulnerability to climate change also stood out as contributing to risk, although the relationships were not as strong ($\rho = 0.57$–0.68).

Principal components analysis of the metric scores for the 73 evaluated taxa showed that the first two principal component axes accounted for about half (54%) of the total variation. Distribution trend, population trend, and projected impact of threats loaded most strongly on the first principal component axis, and Species of Special Concern taxa tended to have positive values for this axis (80% of Species of Special Concern taxa positive; figures 5 and 6). Ecological tolerance and range size loaded most strongly on the second PC axis. However, there is little correlation with special concern status along this axis (figure 6).

Patterns in the Metric Scores

The Total Score/Total Possible ratios for the Species of Special Concern taxa were normally distributed with a mean of 63%, ranging from 38% to 93% (Shapiro–Wilk test for normality, $W = 0.98$, $p = 0.58$). Three of the Species of Special Concern taxa are also federally listed as endangered or threatened, and all of these taxa (California red-legged frog, *Rana draytonii*; arroyo toad, *Bufo californicus*; Yosemite toad, *B. canorus*) had a Total Score/Total Possible greater than 75%, occurring in roughly the top 20% of Species of Special Concern (figure 3). The top 20% of taxa were amphibians, with the exception of the western pond turtle (*Emys marmorata pallida*) (figure 3). In contrast, the lowest scoring 20% of Species of Special Concern taxa were all reptiles with the exception of the Santa Cruz black salamander (*Aneides flavipunctatus niger*) (figure 3). On average, turtles and frogs and toads had the highest scores among the five major taxonomic groups (frogs and toads, salamanders, lizards, snakes, and turtles; figure 7).

We were unable to score certain metrics due to a lack of data. Population trend had the largest number of deficiencies with 26% (19/73). Distribution trend was data deficient for 8% of

TABLE 3

Spearman's rank correlations (ρ) among the eight ranking criteria scores

Values below the diagonal are for the 73 candidate taxa. Values above the diagonal are for the 45 Species of Special Concern taxa

	RS	DT	PCM	EN	ET	PT	CC	PI	TS/TP
Range Size (RS)	—	−0.31*	−0.06	−0.16	0.29	−0.24	0.02	−0.07	−0.04
Distribution Trend (DT)	−0.27*	—	0.30	−0.29	−0.41**	0.46**	0.05	0.28	0.56***
Population Concentration/Migration (PCM)	−0.27*	0.41***	—	−0.12	−0.08	0.49**	0.39*	0.00	0.73***
Endemism (EN)	−0.13	−0.10	−0.04	—	0.26	−0.14	−0.15	−0.40**	0.02
Ecological Tolerance (ET)	0.21	0.00	0.09	0.15	—	−0.41*	0.12	−0.25	0.03
Population Trend (PT)	−0.31*	0.66***	0.57***	−0.01	0.02	—	0.39*	0.33*	0.79***
Vulnerability to Climate Change (CC)	−0.09	0.22	0.50***	0.01	0.30*	0.40**	—	0.08	0.47**
Projected Impact of Threats (PI)	−0.07	0.61***	0.23	−0.15	0.26*	0.65***	0.25*	—	0.25
Total Score/Total Possible (TS/TP)	−0.12	0.77***	0.66***	0.14	0.39***	0.87***	0.57***	0.68***	—

*$p < 0.05$; **$p < 0.01$; ***$p < 0.001$.

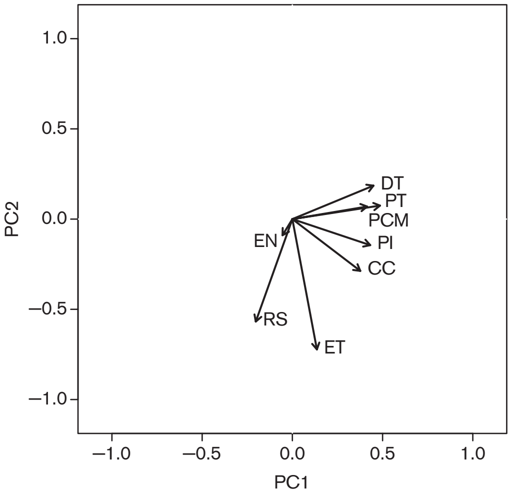


FIGURE 5 Vectors of PCA loading coefficients on first two PC axes. These two axes explain approximately half of the variation in metric score among the 73 nominee taxa. Distribution trend (DT), population trend (PT), population concentration/migration (PCM) loaded strongly onto PC1. Range size (RS) and ecological tolerance (ET) loaded strongly onto PC2. Climate change (CC) loaded equally and moderately on both axes, and endemism (EN) did not load strongly onto either axis.

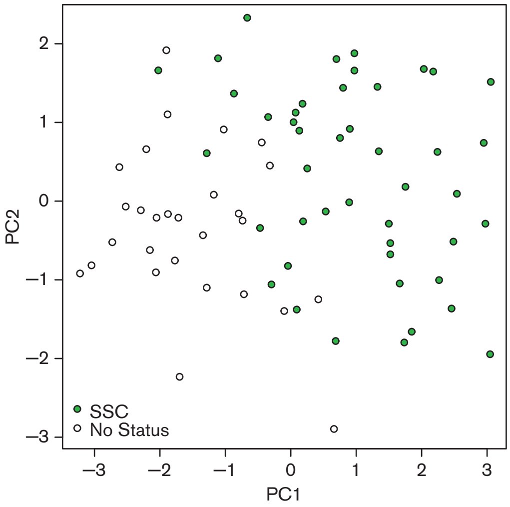


FIGURE 6 Distribution of all 73 taxa evaluated for Species of Special Concern (SSC) status along PCA axes 1 and 2. Most SSC taxa are positive for PC1 and most taxa with "No Status" are negative for PC1. There is little separation among taxa along PC2. SSC are represented by filled symbols. Open symbols are nominee taxa that did not receive SSC status.

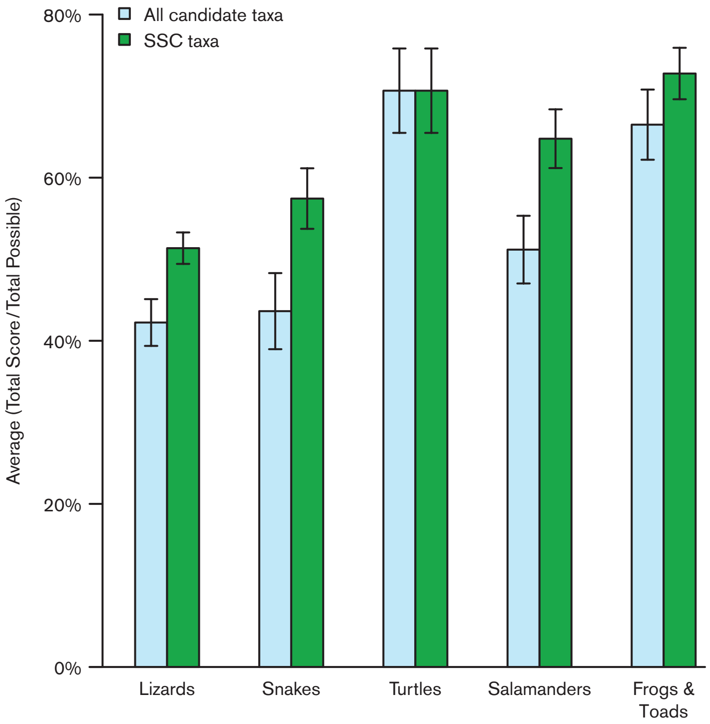


FIGURE 7 Average Total Score/Total Possible by taxonomic group. Filled bars are averages across the Species of Special Concern (SSC) taxa. Open bars are averages across all 73 nominee taxa. Error bars are standard errors.

taxa (6/73), and only a few taxa lacked data on vulnerability to climate change (2/73), projected impacts (3/73), and population concentration/migration (2/73). Among the Species of Special Concern, nine species were data deficient for the critically important population trend metric: Cope's leopard lizard (*Gambelia copeii*), coast patch-nosed snake (*Salvadora hexalepis virgultea*), regal ring-necked snake (*Diadophis punctatus regalis*), California giant salamander (*Dicamptodon ensatus*), Gila monster (*Heloderma suspectum*), Sonora mud turtle (*Kinosternon sonoriense*), lowland leopard frog (*R. yavapaiensis*), Sonoran Desert toad (*B. alvarius*), and red-bellied newt (*Taricha rivularis*). Southern populations of the common garter snake (*Thamnophis sirtalis*) were data deficient for population concentration/migration, and the Oregon spotted frog (*R. pretiosa*) was data deficient for vulnerability to climate change. The Gila monster (*H. suspectum*) was data deficient for three metrics (distribution trend, population trend, and projected impacts), and the regal ring-necked snake was data deficient for the same three metrics plus population concentration/migration.

Certain geographic areas of the state emerged as experiencing a high degree of conservation risk, measured by the number of Species of Special Concern contained within them. At least two important geographic trends emerged from our analysis (figure 8). First, California ecoregions north of San Francisco Bay tended to have far fewer at-risk taxa than those from southern California (figure 8). In particular, the Southern California Coast, Southern California Mountains and Valleys, and the Mojave Desert ecoregions all contained a large number of Species of Special Concern (figures 2 and 8). Second, the geographic pattern of risk varied between amphibians and reptiles. Overall, reptiles experienced the highest risk in the three previously mentioned ecoregions as well as the Colorado Desert, while the northern ecoregions generally had only a single reptile Species of Special Concern (western pond turtle, *E. m. marmorata*). However, amphibian Species of Special Concern taxa were more evenly distributed among ecoregions across the state, with a slight peak in the mountains surrounding the Central Valley and in northern coastal California (generally 7–8 species) and a slight drop-off in the southern ecoregions (generally 5–6 species; figure 8).

To assess possible correlations between habitat type and conservation risk, we scored all 73 nominee taxa as predominantly terrestrial or aquatic, based largely on where reproduction takes place. Our categorization of aquatic versus terrestrial was not identical to that used in Jennings and Hayes (1994a), although it is broadly similar. We categorized amphibians based on their breeding biology—those that lay aquatic eggs and have free-living aquatic larvae were considered aquatic, whereas those with terrestrial eggs and direct development were considered to be terrestrial. Under these criteria, all frogs and toads were scored as aquatic, as well as the salamander genera *Ambystoma, Dicamptodon, Rhyacotriton,* and *Taricha*. Terrestrial salamander genera were all from the family Plethodontidae, and included *Aneides, Batrachoseps, Ensatina, Hydromantes,* and *Plethodon*. All lizards and snakes, including the semiaquatic garter snakes (*Thamnophis*) were considered terrestrial, since all either lay terrestrial shelled eggs or are live-bearing, and all spend the majority of their time on land. All of the turtles were considered to be aquatic since they spend the vast proportion of their lives, including all feeding and mating activities, in freshwater aquatic habitats. Categorizing taxa in this manner shows that there is an overall effect of habitat on Total Score/Total Possible (One-way Anova, $p < 0.0001$; figure 9). The same pattern was true for aquatic versus terrestrial salamanders (figure 9).

Public Comment

The formal public comment period lasted for 60 days over the summer of 2009, although we continued to solicit and incorporate feedback after this period closed. During the public comment phase of the project, the website was visited 886 times by visitors from 17 countries. The majority

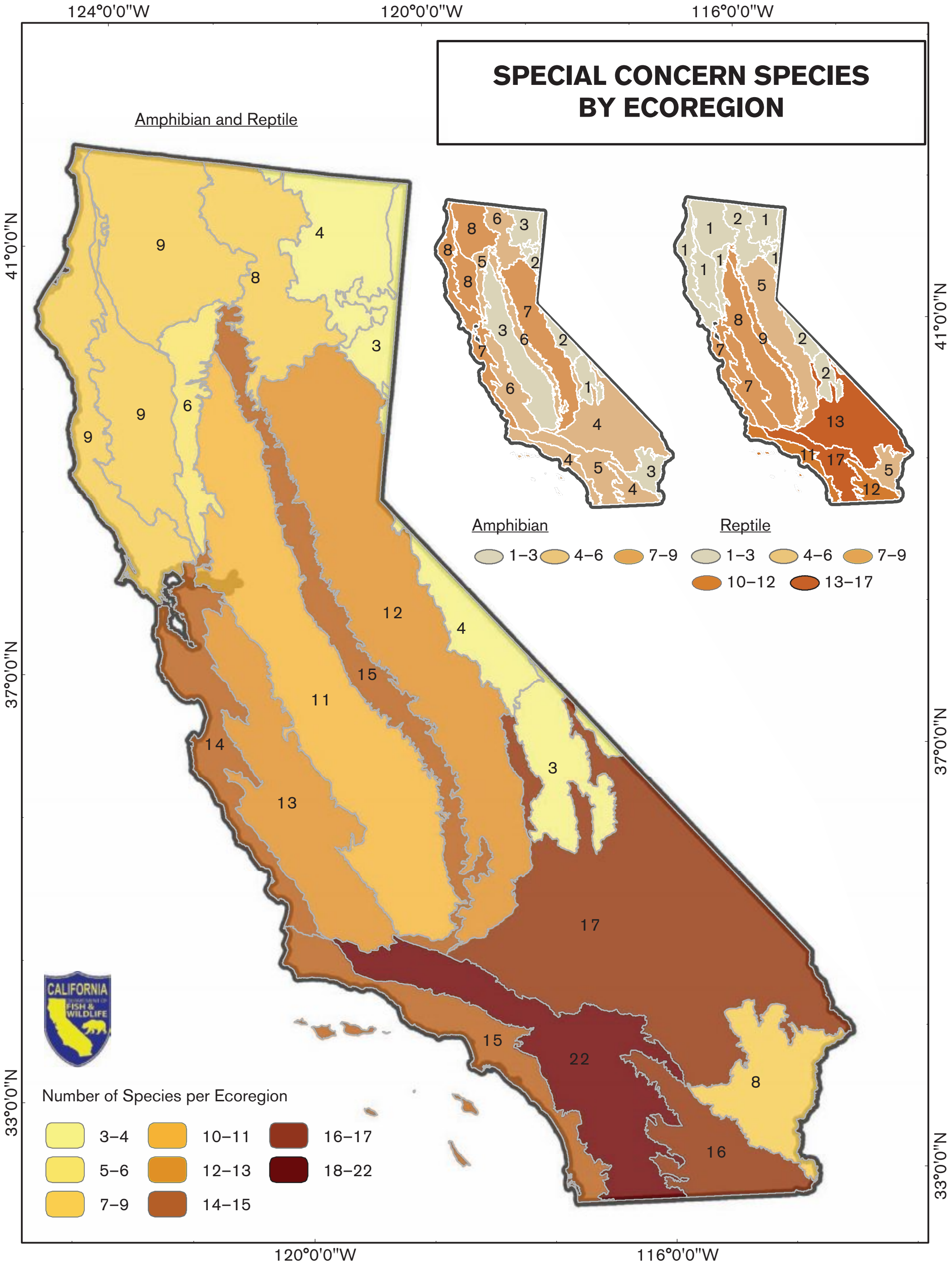


FIGURE 8 The number of Species of Special Concern that occurs within each US Department of Agriculture (USDA) Ecoregion section.

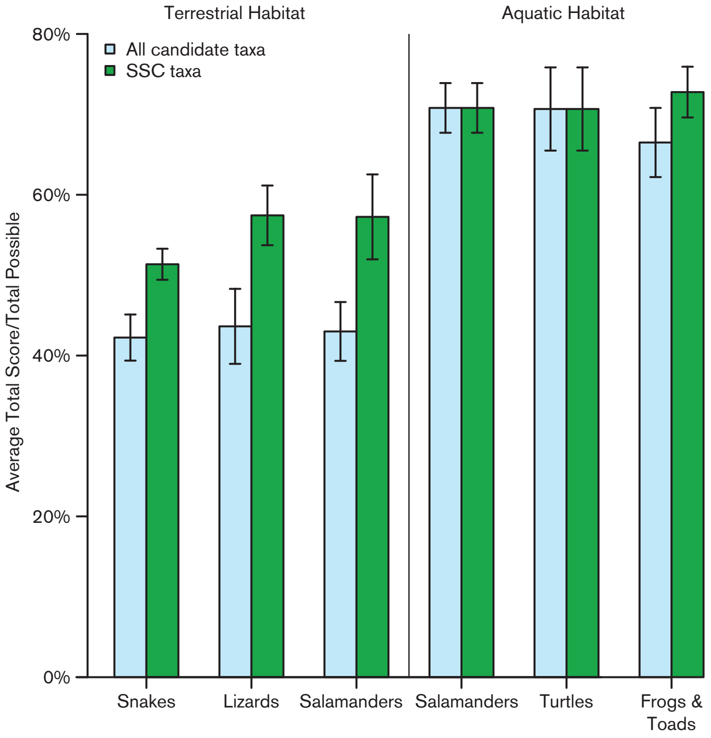


FIGURE 9 Average Total Score/Total Possible by aquatic or terrestrial habitat type. Filled bars are averages across the Species of Special Concern (SSC) taxa. Open bars are averages across all 73 nominee taxa. Error bars are standard errors.

of visitors (575) were from California, followed by visitors from neighboring states (Washington: 32; Oregon: 28; Arizona: 26). We received feedback from a wide variety of conservation professionals, academics, and enthusiasts. Because much of this feedback came from informal conversations on the telephone or at workshops, meetings, and conferences, we cannot precisely quantify the number of data contributors to this project. However, we received substantial contributions in the form of unpublished data, reprints, field notes, and/or localities during the public comment period from approximately 45 individuals (see Acknowledgments).

DISCUSSION

Risk Metrics

Overall, the metrics performed well, successfully identifying taxa that herpetologists generally consider to be at risk across the state, such as ranid frogs. Similarly, scores for the Species of Special Concern that are federally listed suggested that the metrics were performing well. Evaluating all taxa within a metric framework also facilitated identification of patterns among the metric scores that revealed insights into the geographic and ecological factors associated with declines. As emphasized by Shuford and Gardali (2008) for birds, no single set of metrics can capture the intricacies of the natural world fully. The strengths of our approach were that the eight metrics covered a wide range of factors that indicate declines and established a repeatable and transparent baseline for the evaluation of Species of Special Concern. During the initial public input phase of the project, we observed firsthand how a metric-based framework facilitated incorporation of feedback into conservation decisions, regardless of disagreements over the particular metrics used.

That is, when disagreements arose, the metrics allowed us to discuss conflicting scores for individual taxa, focusing discussions on specific issues and questions.

Our metrics covered four basic categories that spanned the diversity of conservation issues faced by any species: geography of declines, changes in population biology over time, key ecological attributes associated with risk, and estimates of future impacts. Metric scores within these categories were often correlated, capturing real patterns in how declines occur. For example, the high correlation between distribution trend and population trend reflects the fact that populations tend to become smaller and smaller as they become isolated and fragmented over time. This general shrinking of populations for many taxa with naturally extensive metapopulations will lead to a high score for population trend. However, as this trend continues over time, those isolated, declining populations experience much greater demographic stochasticity (Lande 1988), leading to more frequent extirpations of local populations and thus high scores for distribution trends. Thus, although these two metrics could be decoupled in principle, our assessments indicate that they tend to be associated in natural systems, and the metrics reflect this association rather than a redundancy in the approach. They also highlight the importance of measuring population connectivity as a research goal and of maintaining or reestablishing it as a management objective.

The correlation among metric scores may help explain why the rankings were robust to data deficiencies. This feature of the rankings is critical when evaluating reptile and amphibian taxa that can be cryptic, rare, and for which survey data are often lacking. We ranked taxa using the ratio of the total score to the total possible, rather than just the total score, to account for the different possible total scores for each species arising from data deficiencies. An implication of this approach is that each species' score is based on the data available and that the metrics differentially influenced scores depending on data availability. For example, population size is difficult to estimate with precision and generally requires extensive multiyear field studies. As a result, we could not score population trend for eight Species of Special Concern. If such data deficiencies were biasing our results, then this would be reflected in a different distribution of Priority 1, 2, 3 and Undefined scores for data-deficient taxa compared to the overall set of Species of Special Concern, but this was not the case ($\chi^2 = 5.4$, df = 3, $p = 0.14$). We acknowledge that data deficiencies in key metrics, such as distribution and/or population trend, could allow for taxa to achieve high Total Score/Total Possible ratios based on having only moderate scores for the remaining metrics. Although this was rarely an issue in our analyses, we also believe that this captures a realistic axis of risk. Taxa that have life histories indicating some amount of risk, particularly small range size and high ecological specialization, but for which we have no data on trends in abundance or distribution, are prime candidates both for unnoticed declines and for further research or monitoring. By scoring them as data deficient and basing their overall score only on available data, we explicitly upweight the importance of those metrics for which we do have information, appropriately bringing them to the attention of biologists and resource managers.

The metric scores were informative for broadly categorizing risk, with generally accepted high-risk taxa receiving the highest scores (e.g., arroyo toad, *Bufo californicus*) and clearly low-risk taxa receiving the lowest scores (e.g., northwestern garter snake, *Thamnophis ordinoides*). If a few strongly correlated risk metrics were uniformly high for at-risk taxa, this could have produced a sharp break point in overall score for Special Concern taxa, but this was not the case. Instead, the risk metric scores formed a smooth continuum from very low to extremely high Total Score/Possible Score values, indicating that a wide variety of combinations of metric scores characterized different taxa (figure 3). This smooth continuum in scores made it difficult to use metric scores alone to decide on special concern status, particularly for the lower-ranking taxa. It also forced us to focus on the specific biology of taxa with lower metric scores in evaluating whether they should or should not be Species of Special Concern. For example, the yellow-blotched ensatina (*Ensatina eschscholtzii croceater*) has much of its small range on private land, and concerns regarding the management and development of that land was a primary motivation for its previous designation as a Species of Special Concern (Jennings and Hayes 1994a). However, more recent planning efforts have emphasized the importance of retaining much of the yellow-blotched ensatina's habitat as unfragmented space (e.g., Tejon Ranch Conservancy 2008). This shift to regional conservation planning addressed the concerns about habitat loss for this species as described in the previous amphibian and reptile Species of Special Concern document (Jennings and Hayes 1994a), so we placed it on the Watch List. However, we identified the sandstone night lizard (*Xantusia gracilis*), which has a lower metric score, as a Species of Special Concern because of its tiny range size and associated potential for extinction.

The same was generally true for assigning priority rankings to individual taxa. Once again, there are no clear cut-offs in ranking scores among Species of Special Concern taxa in figure 3, making the identification of unambiguous criteria for priority score difficult. If the correlation between ranking and priority were perfect (or if we defined priority based solely on ranking), then all Priority 1 (green) taxa would be at the bottom of figure 3, Priority 2 (yellow) would be next, Priority 3 (peach) next, followed by taxa with No Status (white) at the top of the figure. This is close to, but not identical with, our priority ranking scheme.

We could have simply imposed priority-level cut-offs using the metric scores themselves rather than trying to add information that goes beyond a ranking based entirely on metrics. We did not do so because we felt that this would amount to a statement that all relevant biological information for each species was captured in the metric data. For example, the red-bellied newt (*Taricha rivularis*) ranked in the top 20% of taxa but is considered a Priority 2 Species of Special Concern. This decision was made because the ecological and population size data for this taxon are limited in scope, such that it was not possible to conclude that severe future declines and/or extirpation are likely without immediate conservation actions. Overall, we view the metrics as a useful but necessarily approximate guide for informing conservation decisions, not a complete replacement for careful consideration of the biology of each taxon on a case-by-case basis.

Taxonomic Patterns in Metric Scores

Taxonomic patterns among the Species of Special Concern can be measured as the total number of taxa, the fraction of the total number of species in the state that are Species of Special Concern, or as the average numerical metric score (Total Score/Total Possible) for different taxonomic groups. Each is informative, and together they provide a more complete overall picture of the status of the amphibian and reptile fauna of California than does any single measure.

When viewed in the context of all 217 taxa that are known to naturally occur in California (Appendix 1), turtles and amphibians are the most at-risk taxonomic groups. Among the candidate taxa, turtles and frogs had similar average metric scores (71% and 67%, respectively; figure 7), and many of these taxa are Species of Special Concern. All of California's nonmarine turtles are at risk at the Species of Special Concern or State Threatened level (figure 10). This pattern mimics the situation for turtles and tortoises globally; according to the IUCN, turtles have the highest fraction of Red List taxa among any major group (39% of all species and 62% of the currently evaluated species; Rhodin et al. 2010). While very few turtle species occur in the state, half of California's frogs and toads

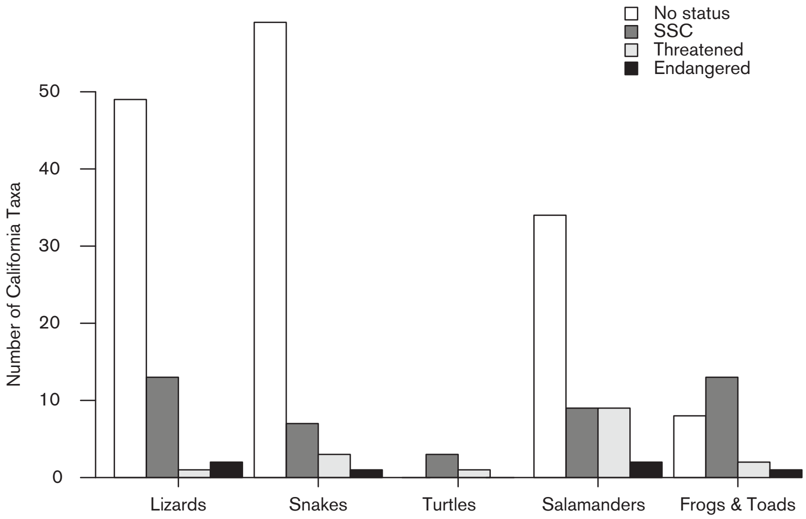


FIGURE 10 Percent of California reptile and amphibian taxa (*n* = 217 by state protected status: Endangered, Threatened, Species of Special Concern [SSC], No Status).

are included as Species of Special Concern. The state's other amphibian group, salamanders, has the next highest fraction of included taxa, with squamates (lizards and snakes) being least at risk at the state level (figure 10). These patterns are consistent with global concerns about amphibian declines in recent decades (Lannoo 2005). No frogs or toads were included in the additional taxa in need of research and monitoring category (Appendix 4), which confirms that a disproportionately large research effort has focused on this globally imperiled group compared to other taxa (Stuart et al. 2004).

Ecological Patterns in Metric Scores

Although taxa can be categorized along a variety of ecological axes, one clear distinction is between aquatic and terrestrial primary habitat requirements. The most striking overall pattern is the higher Total Score/Total Possible scores for aquatic (all frogs and toads, aquatic salamanders, turtles) compared to terrestrial (terrestrial salamanders, lizards and snakes) taxa. Jennings and Hayes (1994a) suggested that taxa having aquatic life stages were more extinction prone than terrestrial taxa, and our analysis supports this conclusion. However, phylogenetic and ecological patterns are confounded in this analysis because all frogs and turtles that we scored are also aquatic and all of the lizards and snakes were terrestrial. Thus, it is not clear whether frogs, toads, and turtles as taxonomic groups are at risk or whether obligatorily aquatic taxa are at risk. Salamanders provide some insight into this issue, as both aquatic and terrestrial taxa occur in California. The Total Score/Total Possible metric scores for Species of Special Concern in these two groups are strikingly different (terrestrial salamanders 57%, aquatic salamanders 71%) and consistent with the interpretation that aquatic taxa are, on average, at greater risk than terrestrial ones. Even within salamanders, however, phylogeny is still a confounding variable because all salamanders in the family Plethodontidae are terrestrial, whereas all of the other California salamanders are aquatic. While the overall pattern of higher scores for aquatic taxa is clear, it is not possible to infer causality from this analysis.

Concluding Thoughts on Metric Score Patterns

Two general conclusions emerge from our analyses of metric scores across taxa. First, regardless of whether the pattern is driven by evolutionary relatedness or some intrinsic feature of aquatic ecosystems, aquatic species are at greater risk than terrestrial ones. Second, amphibians overall are at greater risk than reptiles. Both of these conclusions may stem from the ecology of aquatic and terrestrial taxa, particularly in the relatively arid landscape that dominates much of California. Although amphibians have been characterized in the past as harbingers of habitat deterioration due to their permeable skin and sensitivity to environmental chemicals, recent work suggests that this may be less of a general conclusion than was previously thought (Kerby et al. 2010). However, what is clear is that water is a limiting resource over most of California, and climate change predictions for the next 50–100 years indicate that this limitation will only increase in the future. Aquatic habitats in California have also been particularly negatively impacted by nonnative fish, amphibian, and invertebrate introductions (see discussion below), and managing and preventing future introductions is a major challenge to conserving aquatic habitats. Aquatic invasive predators, combined with water modification and overutilization, have led taxa that rely on water, be it a mountain stream or vernal pool, to more precipitous declines than purely terrestrial taxa.

The fact that aquatic taxa are more at risk does not, however, indicate that terrestrial taxa are uniformly secure, now or in the future. The greatest biodiversity hot spot for terrestrial lizards and snakes in the state is in southern California (Parisi 2003; figure 8). Much of this region has experienced heavy development which has lead to major conservation concerns. Coastal taxa that are diurnally active and highly mobile (e.g., coast patch-nosed snake, *Salvadora hexalepis virgultea*; coastal whiptail, *Aspidoscelis tigris stejnegeri*) are particularly at risk, in part because habitat fragmentation and heavy road traffic, interactions with humans, their commensals (e.g., raccoons, skunks, rats, crows), and pets (dogs and cats), as well as general problems with fragmented habitat and a loss of metapopulation dynamics. In addition, some of the greatest areas of urban growth in California are in the relatively sparsely populated inland xeric regions, where remote conditions and lack of easily developed water and infrastructure have thus far protected many species. As these regions become more heavily populated and more fragmented by roads and urban centers, we predict a shift in endangerment patterns over the next several decades.

To help avoid future population declines, listings, and extinctions, amphibian and reptile Species of Special Concern are sometimes considered in both urban and large-scale planning efforts. Large-scale efforts originate at both the state (Natural Community Conservation Plan [NCCP]) and federal (Habitat Conservation Plan [HCP]) levels and involve cooperation between the two jurisdictions and other public and private partners. For example, five amphibian or reptile Species of Special Concern are included in the heavily populated planning area covered by the San Diego Multiple Species Habitat Conservation Plan (http://www.wildlife.ca.gov/Conservation/Planning/NCCP/Plans/San-Diego-MSCP). As of December 2013, nine approved NCCPs were being implemented, some of which include amphibian and reptile taxa, and 16 NCCPs were in the planning phase. Of the nine plans undergoing implementation, 1.5 million acres (0.6 million hectares) have been committed to reserve lands. The total planning area for the 25 NCCPs covers over 33 million acres (13.3 million hectares) (http://www.dfg.ca.gov/habcon/nccp/). As of 25 June 2014, there are 147 approved Federal HCPs in California (http://ecos.fws.gov/conserv_plans/). HCPs are primarily focused on federally listed species, so any benefit to ARSSC taxa is typically incidental to the plan.

Other large-scale wildlife planning efforts include a statewide assessment of essential habitat connectivity sponsored by the CDFW and the California Department of Transportation. The effort identified large remaining blocks of intact habitat or natural landscape and linkages between them that need to be maintained, particularly as corridors for wildlife (http://www.dfg.ca.gov/habcon/connectivity/).

Peripheral Populations and Endemic Taxa

At least 10 of the 45 Species of Special Concern are best considered peripheral in California. For these species, the bulk of their range occurs outside of the state, where they may be abundant and in little danger (e.g., Couch's spadefoot, *Scaphiopus couchii*), of relatively uncertain status (e.g., regal ring-necked snake, *Diadophis punctatus regalis*), or declining and protected (e.g., Oregon spotted frog, *Rana pretiosa*). Particularly for those taxa that are common rangewide, a reasonable question to ask is whether they should be protected in California, where they may occur in marginal habitat at the edge of their ranges. From a biological perspective, conditions beyond the state's borders are clearly relevant to range-wide conservation risk. However, from a political and jurisdictional perspective, managing populations outside of California is not the state's responsibility. We consider peripheral taxa as valid Species of Special Concern because the CDFW's mission is to "maintain native fish, wildlife, plant species and natural communities for their intrinsic and ecological value and their benefits to people [...] include[ing] habitat protection and maintenance in a sufficient amount and quality to ensure the survival of all species and natural communities" that naturally occur in California (http://www.dfg.ca.gov/about). Therefore, peripheral populations are similar to taxa whose entire range occurs within the state in that they are established, natural components of the biodiversity of California; whether they require special conservation measures should be based on their current status in the state. Two of our metrics, range size and endemism, take the peripheral nature of populations into account, at least indirectly. Range size generally upweights these populations, since they have small ranges within the state. Countering this, endemism measures the fraction of the species' overall range that occurs in California, which tends to downweight such taxa. Each had a maximum score of 10, so they had equal impacts in the total score for each taxon.

Endemic taxa, by contrast, are clearly one of the state's most important conservation responsibilities (Table 4). Because they occur nowhere else, these taxa make up a critical component of California's unique amphibian and reptile fauna, so conservation successes or failures within the state are likely to have much larger impacts on these species than taxa that range more widely.

Geographic Patterns in Species of Special Concern

Range maps are an important resource in delimiting changes in the distribution of taxa. However, range is also difficult to determine precisely for many reptiles and amphibians due to their naturally low population densities, cryptic natural history, and the paucity of survey data. In constructing these range maps, we included, rather than excluded, regions where the likelihood of occurrence was high but no specimens have been documented to date. Our reasons for doing so were twofold. First, by setting boundaries that may be too large, we hope to encourage field researchers to expand their geographical horizons when searching for new localities. Second, since the taxa are at-risk, we want to err on the side of potential habitat inclusion for conservation purposes. We used previously established units (watershed boundaries, ecoregions, etc.) rather than arbitrary polygons around localities to provide objective boundaries from which future surveys can work. For instance, where we drew a species as being present in one watershed but absent in the next, this provides a very straightforward way to focus additional surveys. Surveyors can ask the question, "Is the taxon present in the adjacent watershed?," and focused efforts can answer that question, refining range boundaries in an organized, efficient manner.

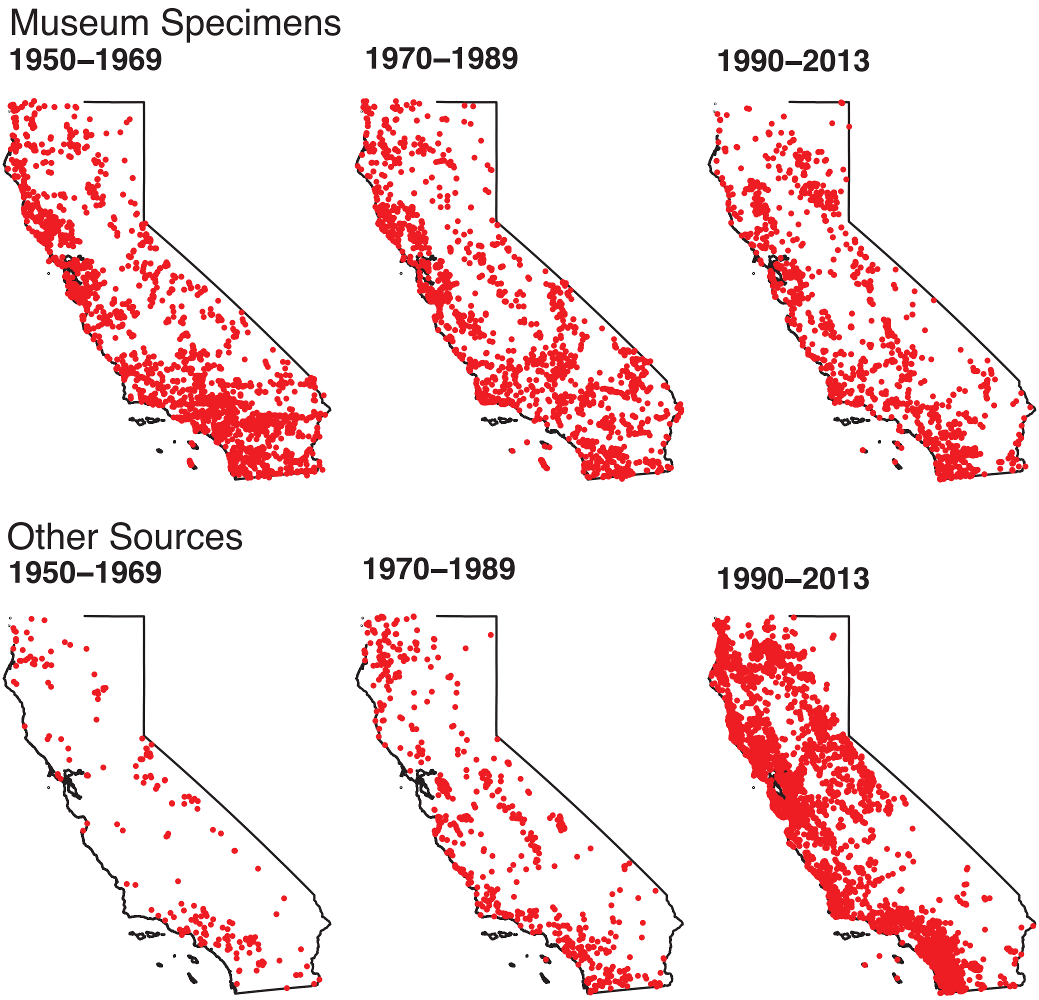
These maps also highlight an important, frequently overlooked point: we need a mecha-

TABLE 4

Endemic and Near Endemic Species of Special Concern

Endemic	
Aneides flavipunctatus niger	Santa Cruz black salamander
Batrachoseps campi	Inyo Mountains salamander
Batrachoseps minor	Lesser slender salamander
Batrachoseps relictus	Relictual slender salamander
Bufo canorus	Yosemite toad
Dicamptodon ensatus	California giant salamander
Elgaria panamintina	Panamint alligator lizard
Masticophis flagellum ruddocki	San Joaquin coachwhip
Taricha rivularis	Red-bellied newt
Taricha torosa, Southern populations	Coast Range newt
Thamnophis sirtalis, Southern populations	Common garter snake
Xantusia gracilis	Sandstone night lizard
Xantusia vigilis sierrae	Sierra night lizard
Near endemic	
Aniella pulchra	California legless lizard
Bufo californicus	Arroyo toad
Emys marmorata marmorata	Northern western pond turtle
Emys marmorata pallida	Southern western pond turtle
Phrynosoma blainvillii	Coast horned lizard
Rana boylii	Foothill yellow-legged frog
Rana draytonii	California red-legged frog
Spea hammondii	Western spadefoot
Uma scoparia	Mojave fringe-toed lizard

nism, including a curated database, that tracks documented absence as well as documented presence data. Documenting, and even defining, absence is often a very difficult problem, but these efforts can be helped by collating survey results (including both positive and negative occurrence data) into a publically available and easily accessible format. Locality data from the past couple of decades tend to come from sight records, survey data, and other field research that does not result in the collection of museum specimens (figure 11). While museums are increasingly making their data accessible through online databases, there is currently no centralized way to collate locality data from other sources across all California reptiles and amphibians. The California Natural Diversity Database is an important means by which the state collates status and location information for Species of Special Concern and those listed under the federal and California Endangered Species Acts. Currently, this resource does not document absence data for sites where only negative surveys have occurred and focuses solely on those taxa on California's Special Animals list. Expanding the scope of this



Museum Specimens

1950–1969

1970–1989

1990–2013

Other Sources

1950–1969

1970–1989

1990–2013

FIGURE 11 Distribution of Species of Special Concern locality records over time. Data from other sources include records from the California Natural Diversity Database and the Biogeographic Information and Observation System, both of which contain some museum records, though the majority of records plotted are from survey data.

database or adding an additional database to capture negative occurrence data, as well as survey data from other taxa, would help the state's efforts to improve estimates of species ranges.

When we plot the number of at-risk species contained within each ecoregion, geographic patterns in conservation risk emerge (figure 8). The southern California coast and mountains and the Mojave Desert have the largest number of at-risk species overall, although this pattern is due largely to trends among reptiles. This important area of conservation risk is driven along the south coast by habitat loss and degradation arising from the massive land use changes that this area has experienced over the last century. The Mojave Desert, conversely, is often viewed as being less disturbed and protected by reserves, parks, and military reservations. Our analysis highlights that this is not entirely true. The Mojave Desert has experienced some degree of habitat degradation and loss, although, to date, not as strongly as that which has occurred along the coast where extensive urbanization has effectively removed large areas of habitat. However, the Mojave

Desert, as well as the Great Basin, Colorado, and Sonoran Deserts, and some of the southern Sierra Nevada and associated foothills constitute the 22.5 million acre planning area for future renewable energy development (wind, solar) in southern California. In addition, many of the at-risk species in the Mojave Desert use specialized and rare resources that have experienced a disproportionate amount of habitat degradation relative to other areas of the desert. For example, the fringe-toed lizards of the genus *Uma* exclusively use sand dune habitats, which also disproportionately attract off-highway vehicle use even in some protected areas (see species accounts for additional details). The Mojave Desert is also home to a large number of narrowly distributed or rare taxa that may exist at the edge of their physiological tolerance and persist in small, often isolated areas (e.g., Gila monster, *Heloderma suspectum*). These species may be at particular risk of further declines as climate change occurs. Importantly, it is not the case that all desert species are declining equally, since the Great Basin and Sonoran ecoregions have relatively few at-risk reptiles, while an intermediate number occur in the Colorado Desert.

For amphibians, the areas of largest conservation risk are the mountainous areas surrounding the Central Valley and the forested regions of central and northern California (figure 8). These areas have not experienced massive land use change per se, although they have experienced considerable habitat fragmentation and modification stemming from water diversions, timber harvest, and nonnative species (Bunn et al. 2007, http://www.wildlife.ca.gov/SWAP). Some studies indicate that agriculture in the Central Valley has had an impact on some species in the Sierra Nevada and Cascades Range via increased exposure to pesticide drift from the Central Valley (e.g., Davidson et al. 2002, Davidson 2004, Lind 2005). In addition, many of these regions are heavily exploited for timber harvest, and this has also had an impact on both stream-dwelling and terrestrial amphibians (e.g., Olson et al. 2007, Welsh and Hodgson 2008). An emerging threat in northern California is marijuana cultivation, which can degrade both terrestrial and aquatic amphibian habitat (CDFW 2013). Increased sedimentation, dewatering of headwater streams, and application of agricultural chemicals are all potential negative effects of marijuana growing, and these effects should be monitored and potentially regulated. High elevation mountainous areas are expected to experience large impacts from climate change through the altered timing and amount of snowmelt (Cayan et al. 2008b), and this future risk probably affects amphibians to a greater extent than co-occurring reptiles (figure 8). Increasing temperatures associated with climate change may also lead to phenological shifts in several species, which could interact with several of the existing threats (Todd et al. 2011). This pattern in both amphibians and reptiles is driven to some extent by species richness of the respective groups. Southern California and the deserts have the highest richness of reptile diversity, whereas the Sierra Nevada and northern Coast Ranges are home to greater amphibian species richness (Parisi 2003, Stebbins 2003).

Finally, for all taxa we note that the distribution of locality data is uneven and patchy across the state (figure 12). At first glance, it appears that the areas with the greatest human impacts and populations (southern coastal California, the Bay Area) are also the areas with the greatest number of locality records, and it may be that these are simply the areas that have received the greatest efforts from field biologists. Unfortunately, we cannot unambiguously say whether the sparse locality records, for example, from the Mojave Desert reflect sparse fieldwork, underreporting of data, or a genuine low density of animals in the region. Our sense is that all of these factors are contributing to the distribution of locality records. That is, it is almost certainly the case that there has been much more intensive sampling effort, and consequently a larger number of records, in San Diego County than in the eastern Mojave

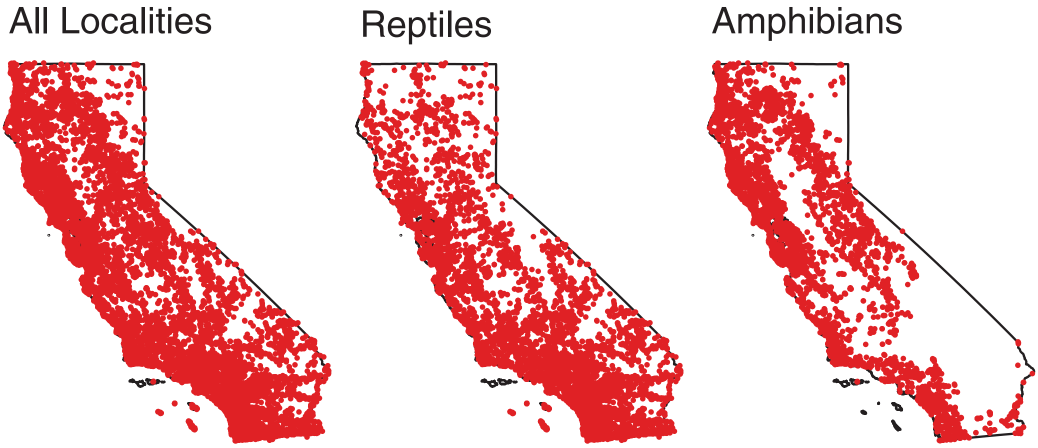


FIGURE 12 Occurrence of Species of Special Concern taxa locality records throughout the state. Regions with few occurrence records may represent areas with few SSC taxa, low sampling effort, or both.

(figure 11). However, it is also true that both reptiles and amphibians are sparsely represented in the eastern Mojave (compare reptile and amphibian maps in figure 12), even though this is a region of high abundance and species richness for reptiles. However, the large number of sensitive species (figure 8) and the recent, intensive development in San Diego County cause many environmental impact assessments to be undertaken under the California Environmental Quality Act, and this has likely contributed to the larger number of records compared to the deserts of southern California.

Differences between This Document and Jennings and Hayes (1994a)

Species priority assessments for conservation purposes are subject to revision over time as factors that affect risk, including habitat protection, invasive species, and scientific knowledge change. Although the number of species identified as being of concern was similar (49 vs. 45), a number of differences exist between the current and previous assessments. Jennings and Hayes (1994a) based their assessment on a combination of their own knowledge and that of a large group of leading experts on individual species; we follow a similar procedure here but summarize the available data using a metric-based approach. Jennings and Hayes (1994a, p.183) felt that for no species of amphibian or reptile was there compelling evidence to "downgrade" status from more threatened to less threatened, whereas we removed several taxa from the Species of Special Concern designation. In total, 34 taxa occur on both lists; we added 11 taxa that were not included by Jennings and Hayes (1994a) and excluded 15 taxa that were previously included (Table 5, Appendix 3).

The status of 43% (26/60) of Species of Special Concern taxa has changed between 1994 and the present. Of the 26 species that changed status, approximately half were upgraded and half were downgraded: 58% (15/26) of the taxa were on the previous list but not the current one, and 42% (11/26) were upgraded from having no formal status to Species of Special Concern (Table 5). These changes reflect differences in approach between these two compilations, insights gained from an additional 20 years of field and systematic research, and real changes that have occurred in the abundance of species. However, on face value, it appears that the past two decades have not been a completely negative period for amphibian and reptile biodiversity in California.

Several factors contribute to these changes. In Table 5, we broadly categorized reasons for changes into three categories. "Listing status"

TABLE 5

Comparison of Species of Special Concern between this publication and Jennings and Hayes (1994a)

Gray cells denote species designated by both publications (see text for additional details)

Taxon	Jennings and Hayes	Thomson et al.	Reason
Ambystoma californiense	X		Listing status
Ambystoma macrodactylum sigillatum		X	New data
Aneides flavipunctatus niger		X	New data
Aniella pulchra	X	X	
Arizona elegans occidentalis		X	New data
Ascaphus truei	X	X	
Aspidoscelis hyperythra[1]	X		New data
Aspidoscelis tigris stejnegeri		X	New data
Batrachoseps sp. "*Breckenridge*"[2]	X	X	
Batrachoseps campi	X	X	
Batrachoseps minor		X	Taxonomy
Batrachoseps relictus	X	X	
Bogertophis rosaliae[3]	X		New data
Bufo alvarius	X	X	
Bufo californicus[4]	X	X	
Bufo canorus	X	X	
Coleonyx variegatus abbotti		X	New data
Crotalus ruber	X	X	
Diadophis punctatus regalis		X	New data
Dicamptodon ensatus		X	New data
Elgaria panamintina	X	X	
Emys marmorata marmorata[5]	X	X	
Emys marmorata pallida[6]	X	X	
Ensatina eschscholtzii croceater	X		New data
Ensatina eschscholtzii klauberi	X		New data
Gambelia copeii		X	New data
Heloderma suspectum	X	X	
Hydromantes platycephalus	X		New data
Hydromantes sp. "Owens Valley"	X		Taxonomy
Kinosternon sonoriense	X	X	
Lampropeltis zonata parvirubra	X		New data
Lampropeltis zonata pulchra	X		New data
Masticophis flagellum ruddocki	X	X	
Masticophis fuliginosus		X	Taxonomy
Phrynosoma blainvillii[7]	X	X	
Phrynosoma mcallii	X	X	

(continued)

TABLE 5 *(continued)*

Taxon	Jennings and Hayes	Thomson et al.	Reason
Pituophis catenifer pumilis[8]	X		New data
Plestiodon skiltonianus interparietalis[9]	X		New data
Plethodon elongatus	X		New data
Rana aurora	X	X	
Rana boylii	X	X	
Rana cascadae	X	X	
Rana draytonii[10]	X	X	
Rana muscosa	X		Listing status
Rana pipiens	X	X	
Rana pretiosa	X	X	
Rana sierrae[11]	X		Listing status
Rana yavapaiensis	X	X	
Rhyacotriton variegatus	X	X	
Salvadora hexalepis virgultea	X	X	
Scaphiopus couchii	X	X	
Spea hammondii[12]	X	X	
Taricha rivularis		X	New data
Taricha torosa (Southern populations)	X	X	
Thamnophis hammondii	X	X	
Thamnophis sirtalis ssp.	X	X	
Uma notata	X	X	
Uma scoparia	X	X	
Xantusia gracilis	X	X	
Xantusia vigilis sierrae	X	X	

1. Evaluated under the name *Cnemidophorus hyperythrus beldingi* in Jennings and Hayes (1994a).
2. Now included within *Batrachoseps relictus*.
3. Evaluated under the name *Elaphe rosaliae* in Jennings and Hayes (1994a).
4. Evaluated under the name *Bufo microscaphus californicus* in Jennings and Hayes (1994a).
5. Evaluated as a single species, *Clemmys marmorata*, in Jennings and Hayes (1994a).
6. Evaluated as a single species, *Clemmys marmorata*, in Jennings and Hayes (1994a).
7. Evaluated as two subspecies, *Phrynosoma coronatum blainvillii* and *Phrynosoma coronatum frontale* in Jennings and Hayes (1994a).
8. Evaluated under the name *Pituophis melanoleucus pumilis* in Jennings and Hayes (1994a).
9. Evaluated under the name *Eumeces skiltonianus interparietalis* in Jennings and Hayes (1994a).
10. Evaluated under the name *Rana aurora draytonii* in Jennings and Hayes (1994a).
11. Evaluated as part of *Rana muscosa* in Jennings and Hayes (1994a).
12. Evaluated under the name *Scaphiopus hammondii* in Jennings and Hayes (1994a).

applies to a few taxa, like the California tiger salamander (*Ambystoma californiense*), that are no longer considered Species of Special Concern because they were listed under the California Endangered Species Act between 1994 and 2014. These taxa are still considered to be at risk, but their state listing precludes inclusion as a Species of Special Concern. “Taxonomy” is more difficult to categorize because many taxa have had name changes between the two lists.

However, in Table 5 we highlight taxonomic changes that led to either the recognition of a new at-risk taxon or the elimination of a previously recognized taxon that is no longer considered valid. An example of the former is the Baja California coachwhip (*Masticophis fuliginosus*), which was considered a part of the widespread and relatively common coachwhip (*M. flagellum*) in 1994, but has since become more widely recognized as a distinct species (Grismer 2002). We note taxonomic changes in Table 5 that did not impact special concern status, like the elevation of the arroyo southwestern toad (*Bufo microscaphus californicus*) (Jennings and Hayes 1994a) to the arroyo toad (*B. californicus*) (current document) as footnotes. The remaining taxa changed special concern status because of new data. This category covers a variety of factors, ranging from better and more extensive field survey data which has revised our understanding of the severity of threats (e.g., the Mount Lyell salamander, *Hydromantes platycephalus*) to new threats that have been identified since 1994 (e.g., predation by introduced fishes for the southern long-toed salamander, *Ambystoma macrodactylum sigillatum*). Some of the difference in threat evaluation stems from our choice of metrics. For example, climate change is currently a particularly important aspect of conservation risk that was not previously considered. In some cases, the availability of suitable habitat has changed, either positively or negatively. Habitat may be set aside for conservation (e.g., Tejon Ranch appears to be setting aside considerable land that will benefit the yellow-blotched ensatina, *Ensatina eschscholtzii croceater*) but is usually lost (e.g., coastal scrub habitat for the California glossy snake, *Arizona elegans occidentalis*). Finally, we note that the factors listed in Table 5 are an over-simplification of the reasons behind our decisions. An explanation for each of the 15 taxa that appeared on the previous list but not on the new list is also included in Appendix 3.

Ultimately, the comparison of the two Species of Special Concern documents emphasizes what can be learned by periodically updating and evaluating the conservation status of taxa on a regular basis. For the 34 taxa that have remained Species of Special Concern, we can and should ask what more can be done to improve their status. Some of the taxa that are no longer Species of Special Concern may inform the kinds of positive changes that can be brought about by management, research, or both. For example, additional surveys and taxonomic research on the Mount Lyell salamander (*H. platycephalus*) have shown that the species is more widespread than previously thought and clarified the taxonomic status of populations in Owens Valley, which were previously suspected of being distinct and of conservation concern. Finally, the challenges of incompletely known taxonomy that were emphasized by Jennings and Hayes (1994a) still pose a major challenge to effective management; if we do not have a complete catalogue of the taxa that occur in California, we cannot even enumerate what may need protection to maintain biodiversity.

Management Recommendations for California Amphibians and Reptiles

While effective management of the Species of Special Concern will generally require development of specific management strategies tailored to the biology of individual taxa, several general recommendations have emerged from this document.

1. Protect aquatic habitats. The metric scores indicate that aquatic species are at greater risk than terrestrial ones, suggesting that remaining aquatic habitats with native amphibian and turtle populations should be high conservation priorities. California's aquatic habitats have been highly modified from a faunal perspective. As of 2002, there were 51 nonnative freshwater fishes in California, the majority of which were deliberately introduced to enhance recreational fisheries (Moyle 2002). Nonnative fishes now predominate in many California waterways, raising concerns about increased competition, predation, habitat interference, disease, and hybridization with native species (CDFG 2008). A large body of ecological research has demonstrated a negative effect of introduced fishes and bullfrogs (*Rana catesbeiana*) on California's native anurans (e.g., Hayes and Jennings 1986, Tyler et al. 1998, Knapp and Matthews 2000, Vredenburg 2004, Knapp 2005, Leyse 2005, Welsh et al. 2006, Pope et al. 2008). As a result, predatory salmonids, centrarchids, catfishes, and other nonnative species should be eradicated wherever feasible and should not be introduced into remaining native amphibian or reptile habitat. Maintaining appropriate water flow regimes for stream-dwelling taxa is also critical, as are broad riparian buffers to maintain lotic habitats and reduce siltation (e.g., Lind et al. 1996, Yarnell 2005, Hancock 2009).

Specific management recommendations include the following:

- Control, or eliminate where possible, invasive aquatic species, particularly predatory fishes, crayfish, and bullfrogs. For widespread, established invasives, plans should be developed with actions that reflect those identified in the California Aquatic Invasive Species Management Plan (CDFG 2008). For bullfrogs in particular, plan Objectives 5 and 6 apply: Education and Outreach and Long-Term Control and Management. Invasive species in the early stages of colonization (e.g., *Nerodia fasciata*, *N. sipedon* and *N. rhombifer*) should be eradicated as soon as possible to prevent further spread. Known to be present in California since the 1990s, coordinated efforts have yet to effectively coalesce to make significant progress toward eradicating *Nerodia*, though educational (http://biology.unm.edu/mmfuller/WebDocs/HTMLfiles/nerodia.html) and occasional agency efforts occur.
- Eliminate, limit, or mitigate effects of dams, water diversions, and other hydrological disturbances to breeding streams whenever possible, and particularly during breeding seasons.
- When biologically appropriate, enhance connectivity and continuity of streams to allow free movement of aquatic species. Conversely, the potential for increasing connectivity to facilitate the spread of invasive species or disease should be considered on a species-by-species basis.
- Maintain riparian vegetation buffers and adjacent upland habitat.
- Eliminate roads within buffer zones and mitigate their effects in high-use amphibian migration areas whenever possible to avoid siltation and road mortality.
- Restrict use of heavy equipment on dirt roads and upland habitats, particularly during the breeding season when eggs and small larvae may be most affected by siltation.
- Maintain culverts under roads adjacent to breeding streams to reduce siltation.

2. *Protect integrity and connectivity of large terrestrial habitat patches.* The size of habitat patches necessary to support healthy populations of most species may be larger than previously recognized (Prugh et al. 2008). The amount and configuration of habitat clearly has a strong impact on the overall extirpation and recolonization dynamics of adjacent populations, and ultimately, of entire species. Besides the general conclusion that more intact habitat is always desirable, specific requirements will always need some level of study on a species-by-species basis. For example, ongoing work on the state and federally endangered California tiger salamander (*Ambystoma californiense*) suggests that this species routinely moves long distances (up to 2 km) away from breeding ponds, suggesting that the extent and quality of upland habitat is likely to have a strong impact on the species' long-term persistence (Trenham and Shaffer 2005, Searcy and Shaffer 2008, Searcy and Shaffer 2011). Several diurnally active and wide-ranging reptile species in southern California appear to be sensitive to habitat fragmentation and disappear from patches of small suit-

able habitat (e.g., coastal whiptail, *Aspidoscelis tigris stejnegeri*; coast patch-nosed snake, *Salvadora hexalepis virgultea*). Habitat fragmentation is a strong driver of declines for many species, and we recommend that land managers pay particular attention to preserving extensive habitat blocks where possible (see Mitrovich et al. 2009, for a well-worked example).

Although the individual conservation needs of species vary, formal conservation planning occurs on a broader scale that considers large areas of habitat for many species simultaneously. Because of many aspects of their shared biology, amphibians and reptiles are often considered as a group, and some excellent, general guidelines for their management have been developed (see, e.g., the Partners in Amphibian and Reptile Conservation habitat management guidelines http://www.parcplace.org/parcplace/publications/habitat-management-guidelines.html). In addition, the biology of amphibian and reptile species needs to be jointly considered within the framework of larger conservation initiatives. The California Natural Community Conservation Planning program is one such initiative that takes an area-wide approach to conservation planning, simultaneously considering conservation of many plant and animal species as well as potential land use activities (see Fish and Game Code Section 2800-2840). These broadscale, integrative approaches to conservation planning promise to be among the more effective strategies for achieving habitat protection and should become an increasingly central mechanism for conservation planning in California. Preserving linkages between adjacent habitat patches is also a key priority in these landscape-level conservation initiatives. Biologically, these linkages maintain metapopulation connectivity and habitat corridors that are often essential for long-term conservation. The California Essential Habitat Connectivity Project seeks to identify corridors between large remaining blocks of intact habitat and is one step in this direction (Spencer et al. 2010). Projects such as these are critically important for maintaining gene flow and migration among localized populations and should continue to be considered as landscape-level conservation initiatives move forward in the state.

Specific management recommendations include the following:

- All Species of Special Concern and the taxa discussed in Appendices 3 and 4 should be considered in Habitat Conservation Plans, Natural Community Conservation Plans, and other local and regional habitat management planning efforts.
- Develop species-specific ecological and landscape genetic datasets to determine the most important habitat corridors for protection and management of amphibian and reptile Species of Special Concern on specific landscapes.
- Identify and either eliminate or mitigate land uses that interrupt connectivity across habitat blocks that have been set aside for conservation. These might include roads, grazing, mining, timber harvest, and many other land uses and activities.

3. Mitigate the effects of roads as a source of mortality and habitat fragmentation. Roads have two primary effects: mortality and fragmentation (Fahrig et al. 1995, Gibbs and Shriver 2002, Mazerolle 2004, Gibbs and Shriver 2005; see also review in Andrews et al. 2008). The overall impact of road mortality on amphibian and reptile populations varies across road types, from species to species, geographically, temporally, and seasonally, and road-associated mortality levels interact with the movement patterns and seasonal migrations of individual taxa. In other parts of the country, roads have been documented to significantly contribute to fragmentation and reduced gene flow, interrupting normal metapopulation dynamics (Fahrig et al. 1995, Hels and Buchwald 2001, Langen et al. 2009, Clark et al. 2010, Sutherland et al. 2010), and the same presumably occurs in California. For example, surveys of 21 roads for migrating, federally endangered

California tiger salamander (*A. californiense*) in Sonoma County suggest widespread mortality that has increased over time as traffic volume has increased. For surveys of one 1200-ft section of Stony Point Road conducted from 2001 to 2010, 160 of 262 salamanders (61%) found were road mortalities, suggesting that vehicular traffic is a substantial form of death in this extremely endangered species (D. Cook, unpublished data). Langen et al. (2009) identified predictors of hot spots of amphibian and reptile road mortality for use when planning roads or when conducting surveys on existing roads to locate priority areas for mitigation.

Although they have been employed infrequently in California, tunnels that assist amphibian and reptile movements can be an effective management tool that should be more actively investigated (for a comprehensive summary of published and unpublished literature, see Caltrans 2012). Two important aspects of migration tunnels are that they must have some capacity to funnel individuals into the tunnels (drift fences, concrete walls, or other similar structures), and they must be actively maintained. Without regular, scheduled maintenance, tunnels fill with debris, drift fences become covered with leaves, runoff soil, trash, and woody debris, and the tunnel quickly ceases to function. Tunnels may also play a role in the deserts of southern and eastern California, particularly as vehicular traffic increases, and roads fragment previously contiguous habitat. For additional recommendations regarding herpetofauna and roads, see Schmidt and Zumbach (2008).

Specific management recommendations include the following:

- Limit traffic, and consider road closures, during amphibian breeding migrations on sensitive public lands.
- Use signage (e.g., “Newt Crossing” warning signs) to warn vehicular traffic that they are in key migration areas.
- Develop standards for and install, maintain, and monitor usage of tunnels, underpasses or other passage mechanisms to reduce road-related mortality.
- Use various media resources for public education campaigns.

4. *Translocate animals only when biologically appropriate*. A general management strategy, variously referred to as relocation, repatriation, or translocation (Germano and Bishop 2009), is the practice of moving animals across landscapes, often from a site destined for development to a protected site. These efforts have become increasingly common as partial or complete mitigation for development projects that affect amphibians and reptiles. Several key biological issues need to be considered before animals are translocated. Disease transmission is an important problem that has had devastating consequences for several species (Jacobson 1993). The well-known upper respiratory tract infection that has decimated desert tortoise (*Gopherus agassizii*) populations is thought to be derived from released captive animals (Jacobson 1993). Genetic consequences of relocation programs should also be considered. Increasingly, genetic data are allowing researchers to elucidate fine-scaled genetic structure among populations, and the insights gained from nonlethal genetic sampling allow insight into biological parameters that are relevant for conservation including population subdivision, gene flow, migration corridors, and population sizes. However, the overall extent and functional consequence of this variation is still poorly understood for most organisms.

Moving individuals around the landscape has the potential for deleterious effects, either by diluting or eliminating unique historical lineages or by disrupting genetic variation that may be an important component of local adaptation. As emphasized in a recent review (Germano and Bishop 2009), homing and poor habitat quality are two of the primary reasons why translocation efforts may fail, and they should be carefully studied on a case-by-case basis. A recent document providing guidelines for translocations for the California tiger salamander (Shaffer et al. 2009) may serve as a model for some other taxa as well. It emphasizes that translocations should only be attempted into unoccupied habitat, and only after the threats that caused the initial declines have been effectively removed. It also emphasizes that sufficient research must have been conducted to provide compelling evidence that the potential damages that can be done to existing conspecific and heterospecific taxa do not outweigh the potential gains to the animals and populations being relocated.

In some cases, headstarting programs may represent a suitable alternative to repatriation or translocation, particularly if the headstarting is done under seminatural conditions. Many species experience the most severe mortality during early life stages. Raising individuals in captivity from a given site to the size or age where they are past this initial peak of mortality and then releasing them at the site where they were initially collected may avoid many of the potential issues associated with translocations while also providing a temporary boost to populations that are in decline. Headstarting is only appropriate, however, where suitable unoccupied habitat exists, or where introduction of individuals will not create problems for existing species at the introduction site.

Specific management recommendations include the following:

- Only translocate animals when other alternatives do not exist.
- Only translocate animals into situations where other animals at the translocation site will not be adversely affected by the introduced animals.
- Only translocate animals when the ecological requirements of the species exist in the new habitat.
- Utilize methods to increase the likelihood that translocations will be successful. These potentially include "soft" translocations (i.e., moving young animals rather than adults with established home ranges) and moving a sufficiently large number of individuals to ensure that a successful breeding population can establish (Germano and Bishop 2009).

Research, Survey, and Monitoring Needs

Both new research and continuing, long-term monitoring are integral parts of the science-driven protection and recovery of sensitive species. For California amphibians and reptiles, our level of basic knowledge on natural history is frequently so fragmentary that even rudimentary information is lacking, and increasing our understanding of these animals is critical for effective management. Many of the particular research needs are discussed in individual species accounts under the "Monitoring, research, and survey needs" section; here, we highlight several basic research and monitoring needs that are common to virtually all taxa.

Distribution

A statewide survey for all amphibians and reptiles is essential to establish baseline data for ongoing status determination and monitoring. Survey efforts are particularly needed for those Special Concern taxa whose population status or range size are a high priority for clarification. These surveys should employ standardized and repeatable methods, with the data emerging from these efforts made widely and easily accessible (Heyer et al. 1994). The Partners in Amphibian and Reptile Conservation Inventory and Monitoring guide (Graeter et al. 2013) serves as an important resource in the detailed design of these distributional surveys. Greatest need taxa include (1) those that may be recently extirpated, but for which comprehensive surveys have yet to be conducted (e.g., the Sonora mud turtle, *Kinosternon sonoriense*); (2) recently discovered taxa that are currently known from relatively small ranges, which may also be tied to specific narrow habitat types, that have yet to be thoroughly surveyed (e.g., the regal ring-necked snake, *Diadophis punctatus regalis*); (3) at-risk taxa that are difficult to detect or that have ranges that are poorly understood because they occur in remote, difficult-to-survey areas (e.g., the Gila monster, *Heloderma suspectum*); and (4) taxa that may occur only on private land where gaining access can be challenging (e.g., the Oregon spotted frog, *Rana pretiosa* or the western spadefoot, *Spea hammondii*). In addition, surveys of virtually all Species of Special Concern, particularly at their hypothesized range edges, would greatly enhance our knowledge of range boundaries for most taxa.

Natural History

Basic natural history and ecology information is the foundation for effective management, and for most amphibian and reptile Species of Special Concern, it is either fragmentary or completely lacking. Home range sizes, habitat suitability analyses, food habits, the effects of invasive plants and animals, compatibility with grazing and agriculture, the effects of human activities including forestry, recreation, and water diversions are unknown for many of the taxa considered here. For some questions and species, this probably is not a pressing problem—calling the southern long-toed salamander (*Ambystoma macrodactylum sigillatum*) a "generalist predator" is, to the best of our knowledge, correct, and filling in the precise details of which invertebrates are the most important prey in specific situations may not be an urgent management issue. However, in other cases, filling in at least some of this basic ecology is absolutely critical. For example, of the 19 species of pond/stream breeding Species of Special Concern amphibians, we do not have a well-tested, clearly understood model of terrestrial habitat use for a single taxon. For example, we have little idea of whether the southern long-toed salamander (*A. m. sigillatum*) requires 10, 1000, or 10,000 m radius habitat patches around breeding ponds. Filling in these fundamental information gaps, hopefully across a range of habitat types, constitutes the highest priority conservation-related research need for Species of Special Concern.

Climate Change

Climate change is likely to have a number of effects on the California landscape that are relevant to amphibian and reptile conservation. While the impact that climate change has on California's landscape is undergoing extensive study (reviewed by Cayan et al. 2008a) and is a CDFW focus (http://www.wildlife.ca.gov/Conservation/Climate-Science), the impact that these effects will have on amphibian and reptile species requires additional study. The Association of Fish and Wildlife Agencies has initiated work on this problem in the southeastern United States, and the CDFW, in collaboration with the Southwest Climate Science Center, initiated a detailed investigation of future climate impacts on amphibians and reptiles across California (Wright et al. 2013). A major focus of these projects, and one that requires additional research effort, is to model a full range of future climate change predictions and their impacts on both common and rare amphibian and reptile taxa.

Importantly, the interplay between conservation risks that climate change presents and competing factors that will arise needs careful examination. For example, many climate projections forecast a decrease in the snowpack in the Sierra Nevada, as well as a shift in the speed and timing of snowmelt to be both more rapid and earlier in the year (Maurer and Duffy 2005, California Climate Action Team 2006, Maurer 2007). Even for the lowest carbon emissions scenarios and relatively conservative estimates of increasing temperatures, current models predict a 30–60% decrease in Sierra Nevada snowpack (Cayan et al. 2006). This is likely to have important, direct impacts on amphibians that rely on snowmelt-fed streams and lakes for their breeding habitat. In addition, it is likely to further stress California's already overburdened water resources, setting the stage for further conflicts between the ecological needs of at-risk species and municipal and agricultural demands for increasingly limited water.

The combined impacts of changes in climate on biological diversity are likely to be strong. Several studies have documented ongoing (Walther et al. 2002, Parmesan and Yohe 2003, Root et al. 2003, Root et al. 2005, Parmesan 2006, Pounds et al. 2006) and expected (Hughes 2000) implications of climate change, with some estimates predicting 35% or more (Harte et al. 2004, Thomas et al. 2004) of species being "committed to extinction" under mid-range warming scenarios. These effects will likely be especially pronounced for amphibians, which generally exhibit limited dispersal and are already experiencing severe declines (Stuart et al. 2004, Lawler et al. 2010). The uncertainties involved with estimating specific effects that will occur on landscapes, species' responses to these changes, and the interplay of factors that will result from climate change (e.g., agricultural and municipal water needs vs. amphibian breeding habitat needs, alternative energy development in the desert vs. reptile habitat needs) clearly indicate that this topic requires further study. An important step in this direction is a recent initiative by the US Fish and Wildlife Service (USFWS) to fund the California Landscape Conservation Cooperative, an interdisciplinary program to facilitate research and planning across scientific and management agencies in the state (http://californialcc.org/about-us). Results of the CDFW and Southwest Climate Science Center collaboration mentioned above should be integrated into the California Landscape Conservation Cooperative process.

Threats from Disease

Diseases in amphibian and reptile populations have become an issue of global significance. In particular, the pathogenic chytrid fungus, *Batrachochytrium dendrobatidis* (*Bd*), has been linked to precipitous declines in several amphibian species in the state (e.g., the Sierra Nevada yellow-legged frog, *Rana sierrae*) and globally (Stuart et al. 2004). At the present time, no broadly effective management strategies for controlling or mitigating the effects of this pathogen are known, and this is a critical, active research area. Proposed management strategies that would benefit from further study include altering population dynamics to minimize disease outbreaks, treating individual amphibians and habitats to control the prevalence or spread of disease, biological control of *Bd* using the zooplankter *Daphnia magna*, and in the most dire cases, maintenance of captive assurance colonies followed by repatriation with assisted selection (Buck et al. 2011, reviewed by Woodhams et al. 2011). Efforts to develop management strategies should not focus exclusively on strategies for the short term, such as direct control of *Bd* in the wild or captive breeding. Rather, management strategies that allow susceptible amphibians to persist in the wild in the presence of *Bd* are needed for long-term conservation of sensitive species (Woodhams et al. 2011).

The extent and type of interactions that *Bd* may have with other threats, such as climate change, pesticide exposure, or other pathogens, are also key research needs. A growing body of work on *Bd* indicates that it has negative consequences on at-risk species of amphibians in California (Davidson et al. 2007, Morgan et al. 2007, Andre et al. 2008, Lacan et al. 2008, Padgett-Flohr 2008, Briggs et al. 2010), that synergistic interactions with pesticides may have strong biological effects (Davidson et al. 2007), and that terrestrial amphibians may serve as vectors for the disease (Schloegel et al. 2009, Weinstein 2009). Other emerging diseases, particularly those that have their origins in human pets or are a result of human-mediated movements and relocations of animals are also high-priority research targets. Important examples include ranaviruses and iridoviruses, both of which have also been linked to amphibian declines (e.g., Picco et al. 2007, Schloegel et al. 2009).

Phylogeography and Landscape Genetics

Another important research need, and one that may be easier to fill than comprehensive ecological studies, is genetic analyses for most species. Some limited phylogeographic and landscape genetic work has been completed for a few California amphibians and reptiles (or their close relatives), and these analyses have provided key insights into the importance of drainages on stream- and pool-breeding amphibians and reptiles (Shaffer et al. 2000, Spinks and Shaffer 2005, Dever 2007, Wang 2009b, Lind et al. 2011), corridors of land use (Wang et al. 2009), the importance of environmental variables in structuring populations (Savage et al. 2010), and a variety of other problems (e.g., the provenance of introduced populations; Johnson et al. 2010). At a broader, regional-to-range-wide scale, phylogeographic studies have been conducted for several Species of Special Concern, in many cases indicating either that previous subspecies (which often serve as proxies for genetic lineages) are nondiagnosable and correspond poorly to genetic patterns (Rodríguez-Robles et al. 1999b) or that unappreciated lineage diversity is stronger than previously suspected (Shaffer et al. 2004, Leavitt et al. 2007, Parham and Papenfuss 2009). We are aware of phylogeographic work for roughly half of the Species of Special Concern (although many of those studies rely on a single mitochondrial gene and need data from additional nuclear gene analyses), and we strongly encourage the research community to gather these data for the remaining taxa.

Monitoring

To establish that a species or population is declining or recovering requires long-term monitoring. Such efforts can take many forms, each with strengths and weaknesses. Ideally, monitoring data would be generated by intensive, multiyear mark–recapture-based studies that follow the fate of individuals through time, leading to a detailed inventory of population increases and decreases (Heyer et al. 1994). Such monitoring is not difficult conceptually, but it requires time, effort, and often substantial financial resources. However, this is also an area that is undergoing renewed methodological development. Monitoring methods now exist that require less recapture effort and that can incorporate detection probabilities in a rigorous manner, both of which can help to effectively monitor rare and/or cryptic taxa (reviewed by Mazerolle et al. 2007). One such example is the emerging techniques to monitor rare or cryptic taxa via detection of persistent DNA in environmental samples (Ficetola et al. 2008, Dejean et al. 2011).

Techniques to survey amphibians and reptiles vary, depending on the taxon, habitat, and life stage involved. Although standardized survey protocols are essential to proper inventory and monitoring, relatively few have been developed, representing an ongoing research need, particularly for rare taxa or taxa that are difficult to detect. Some of this standardization is beginning to take place and a few excellent resources are available or forthcoming (Heyer et al. 1994; the ongoing Amphibian Research and Monitoring Initiative being undertaken by the US Geological Survey, and the Partners in Amphibian and Reptile Conservation Inventory and Monitoring guides are such examples). In the absence of detailed, multiyear monitoring, we advocate at least two potential approaches that have received relatively little attention to date for amphibian and reptile taxa. The first is single-pass monitoring via population surveys conducted on public lands. Such surveys can be incredibly informative, yet only require a few field days per year to monitor a large number of species and sites (e.g., Thomson et al. 2010). A recent example for 75 ponds from the East Bay Regional Park District provided multiyear data for five species of pond-breeding amphibians and two species of semiaquatic garter snakes, and demonstrates the kind of data that can be collected even with very cursory efforts for each site (S. Bobzien, unpublished data; M. Ryan, unpublished data). A critical goal of such monitoring efforts should be to publish the results in the peer-reviewed literature and/or deposit in a publically available, curated dataset. Our sense is that a great deal of valuable monitoring data exists, but is not easily accessible because it has never been published or made publically accessible. Another type of single-pass "monitoring" can be genetic monitoring. By collecting nondestructive, but vouchered, tissue samples, reasonable estimates of the effective population size (Wang 2009a, Wang et al. 2011), historical population increases or decreases (Piry et al. 1999), and ongoing movement between existing populations (Wilson and Rannala, 2003) can be applied to many populations and species. Although each of these genetic approaches has its own set of assumptions and caveats, together they form a powerful addition to traditional field-based studies of population monitoring.

A second approach to monitoring falls under the more general category of "citizen science" (Bonney et al. 2009, Dickinson et al. 2010). Although often less rigorous and more error prone than more formal monitoring, the interested public comprises a large network of knowledgeable, committed individuals who will often willingly contribute to overall monitoring efforts. These efforts can help identify general patterns of population increases and decreases, as has been amply demonstrated by the very successful Breeding Bird Surveys (Sauer et al. 2011) and Christmas Bird Counts (National Audubon Society 2011) conducted for North American birds. Several programs for citizen-science-based frog and toad monitoring programs are in place in other parts of the United States (e.g., the FrogWatch USA program, http://www.aza.org/frogwatch), and they have provided valuable data on breeding time, duration, and population sizes for frogs and toads based on their audible calls at breeding sites. Road surveys (Coleman et al. 2008) can also provide valuable data on population sizes, although the confounding effects of mortality induced by vehicular traffic is always a concern in such studies. That said, documentation of road mortality, particularly during key migration seasons, is an ideal topic of additional citizen science initiatives. California has recently initiated at least two citizen-science web-based projects focusing on southern California reptiles and amphibians (RASCals; see http://www.nhm.org/site/activities-programs/citizen-science/rascals, and the California chapter of the Field Herp Forum http://www.fieldherpforum.com), both of which seek to increase communication and the dissemination of distributional information on California amphibians and reptiles.

Finally, because monitoring provides the basic information upon which much of conservation rests, a temptation naturally arises to "over-monitor." By this, we mean that additional monitoring becomes favored over the implementation of management actions. Monitoring efforts constitute the most important strategy for measuring the effectiveness of conservation actions. However, monitoring also carries a cost, because these efforts require valuable conservation resources that otherwise might be spent on direct management efforts. Monitoring efforts should have clearly defined goals and well-characterized statistical power, including an assessment of the added benefit to be gained from future monitoring efforts. Monitoring efforts should be clearly documented, and results should be readily accessible. In some cases, the optimal strategy may be limited, but consistent, monitoring combined with direct conservation actions, rather than ever-more detailed monitoring with fewer actions. The implementation of effective management in the face of imperfect knowledge about the status of populations is one of the greatest challenges facing the conservation of many amphibian and reptile species.

Species of Special Concern Conservation Recommendations

To promote the conservation of amphibian and reptile Species of Special Concern in California, we make the following recommendations:

- Maintain a Species of Special Concern Technical Advisory Committee with explicit expertise covering the taxonomic and geographic scope of taxa in California. We recommend that membership on this committee be of relatively limited term (e.g., 10 years) to ensure that new voices and fresh problem-solving strategies are available. We especially encourage that committee composition include some early career scientists, particularly those with strong statistical and technical skills. This group should meet periodically in order to update and revise the status information on the Species of Special Concern.

- Develop and implement a web-based mechanism whereby the Species of Special Concern document can be more easily updated and improved, creating a “living document” that is responsive to changing conditions and new data.
- In conjunction with efforts to facilitate future revisions of this document, support the development of a database that collates species occurrence data. This database should house information on both positive and negative occurrence data and not be limited to species that are already designated as Species of Special Concern.
- Increase wildlife agency capacities to address management needs of California’s amphibians and reptiles, as funding and staffing allow.
- Establish both a priority list and a funding stream for critical research needs for Species of Special Concern.
- Continue to promote strong collaborations between wildlife agencies and the university/research communities throughout California to ensure that the strongest possible science is brought to bear on important management needs and that the state’s research priorities are being pursued.
- Use forthcoming analyses of predicted road usage and construction as a management guide for conservation planning for Species of Special Concern. Included in this analysis should be ways to use tunnels or other constructs to minimize the effects of new and existing roads on Species of Special Concern.
- Create a coordination network for localities, voucher specimens, and tissue samples for amphibian and reptiles from throughout California. Roadkill specimens are a particularly valuable source of information, since they represent vouchered specimens and, in some cases, sources of DNA for genetic research and life history data (diet, body condition, etc.) for ecological studies.
- Create a mechanism by which both professional biologists and concerned citizens can contribute locality, natural history, and other data types that might help detect or quantify conservation risk for Species of Special Concern. Improve data sharing and communication among wildlife agencies, amphibian and reptile conservation groups, and organizations in the avocational herpetological community.
- To facilitate data collection, streamline the process for appropriate permitting for research by professionals, and in the case of citizen science projects, the public.
- Encourage publication of data arising from these efforts in the peer-reviewed literature to increase access to management-relevant findings, particularly for government reports and studies conducted by private consultants.
- Integrate information from this document, as appropriate, with that of an upcoming analysis of the existing regulatory situation for all of California’s amphibians and reptiles and their general conservation needs.

SPECIES ACCOUNTS

COUNTIES AND ISLANDS OF CALIFORNIA

COASTAL TAILED FROG

Ascaphus truei Stejneger 1899

Status Summary

Ascaphus truei is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 61% (67/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Ascaphus truei is a small (2.5–5.0 cm SVL) dark frog with an olive, brown, gray, or reddish dorsum and lighter colored ventral surface. Other color characters include a pale triangular blotch on the snout and a dark eye stripe. This species has rough, granular skin, and the outermost toes on the hind feet are broad. Males have a unique tail-like copulatory organ that is unmistakable. This frog is nocturnal and adults have vertical pupils (Stebbins 2003).

Larvae grow up to 6.0 cm in TL and are adapted to life in fast-flowing streams. They have dorsoventrally flattened bodies and large sucking mouthparts that extend nearly halfway down their head-body on the ventral surface. These morphological traits allow larvae to attach to rock substrates (Altig and Brodie 1972, Nussbaum et al. 1983, Welsh and Hodgson 2011). Larvae often have a light-colored tail tip with a proximal dark band (Stebbins 2003).

Coastal Tailed Frog: 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)	10
vi. Population trend (25)	15
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	67
Total Possible	110
Total Score/Total Possible	0.61

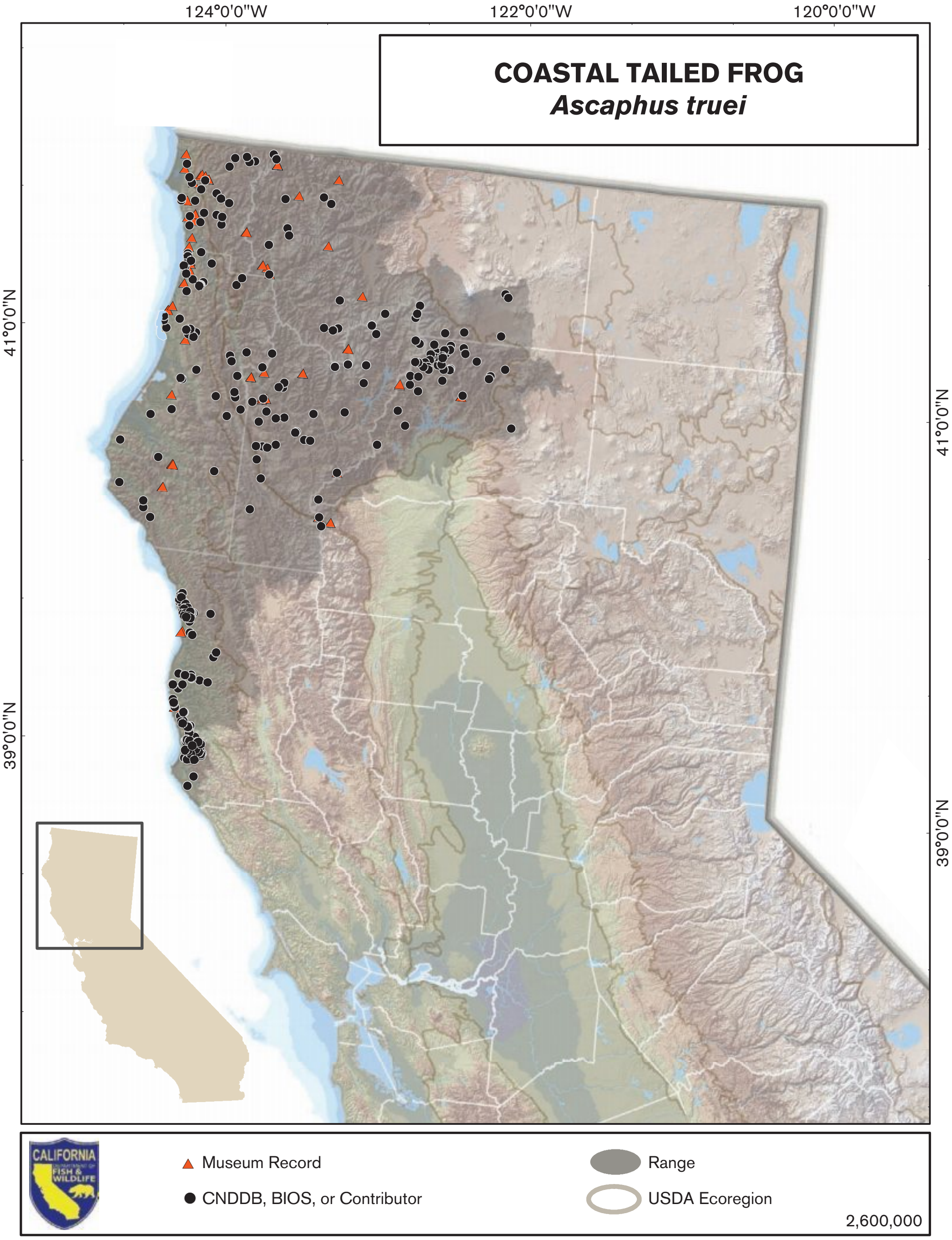


PHOTO ON PREVIOUS PAGE: Coastal tailed frog, Del Norte County, California. Courtesy of Rob Schell Photography.

In California, metamorphosed *A. truei* may be confused with co-occurring foothill yellow-legged frogs (*Rana boylii*). *Rana boylii* have horizontal pupils, more robust hind legs, and males lack enlarged toes and "tails" (Stebbins 2003). In addition, the enlarged mouthparts of *A. truei* tadpoles are distinctive.

Taxonomic Relationships

The formerly monotypic genus *Ascaphus* was recently split into a coastal (*A. truei*) and an inland species (*A. montanus*), but California populations remain *A. truei* (Nielson et al. 2001, Nielson et al. 2006). The two species of *Ascaphus* comprise the family Ascaphidae. This family forms the sister group to all other anurans either alone or in combination with the New Zealand endemic Leiopelmatidae (Roelants et al. 2007). In either case, it is from one of the oldest and most phylogenetically distinctive extant anuran lineages.

Life History

Ascaphus truei exhibits substantial geographic variation in life history. Here, we focus on data from California populations where possible. Breeding occurs primarily in the spring and summer in coastal populations (Sever et al. 2001, Burkholder and Diller 2007), but there are reports from Trinity County of animals found in breeding condition in the fall (J. Garwood, pers. comm., in Burkholder and Diller 2007). Females likely breed in alternate years (Burkholder and Diller 2007) and can store viable sperm for up to a year (Nussbaum et al. 1983, Sever et al. 2001). Eggs begin developing in the fall, and oviposition occurs the following summer between July and September in California populations (Sever et al. 2001, Karraker et al. 2006). Egg diameter is 4 mm on average (Brown 1977), and clutch size averages around 40 for the species with a range of 28–89 eggs per clutch documented in California populations (Karraker et al. 2006). Egg masses can be difficult to find in the field (Karraker et al. 2006). Recent surveys in coastal California have found single and multiple clutches, with the timing of the surveys (late August–early September) likely the most important factor for detecting eggs (R. Bourque, pers. comm.). Clutches are pearl-like strings of eggs and have been found attached to the underside of cobble or boulder substrates in riffles and pools (Karraker et al. 2006).

Time to metamorphosis in lowland coastal California populations (elevation <200 m) is 1–2 years (Wallace and Diller 1998, Bury and Adams 1999). Longer developmental times have been observed in montane populations (e.g., 4 years to metamorphose in a Washington population at ~1500 m elevation; Brown 1990). In a population in Humboldt County, California, females reached sexual maturity 2.5–3 years after metamorphosis, while males were sexually mature 1.5–2 years after metamorphosis (Burkholder and Diller 2007). Post-metamorphic frogs grow year-round, with growth rates fastest in the summer (Burkholder and Diller 2007).

Adults and post-metamorphic juveniles are generalist invertebrate predators (Bury 1970b). Larvae are generalist grazers and scrapers, consuming diatoms and other periphyton (observations from *A. montanus*; Metter 1964).

Landscape genetic studies have detected different patterns of connectivity among populations in California and Washington. In four watersheds in Mendocino County at the southern range limit of the species, high population structure among watersheds suggested limited long-distance gene flow, and movements within watersheds were inferred to occur along waterways (Aguilar et al. 2013). By contrast, a study in Washington concluded that some animals engage in long-distance dispersal through terrestrial habitats, and these movements do not rely on stream connectivity (Spear and Storfer 2008). These differences may be due to regional variation in climate and forest type, though additional studies are needed.

Habitat Requirements

Ascaphus truei requires cold, permanent, swift-flowing streams with coarse (e.g., cobble, boulder, bedrock) substrates. Some populations may persist in streams that occasionally dry depending on the length of the larval period (Wallace and Diller 1998). *Ascaphus truei* tends to be more common in mature and old-growth forest relative to younger stands, in terms of both presence and abundance (Bury and Corn 1988, Corn and Bury 1989, Welsh 1990, Gomez and Anthony 1996, Welsh and Lind 2002, Welsh et al. 2005, Ashton et al. 2006).

Several studies have examined the relationship between *A. truei* presence and abundance and environmental variables at different scales. Larvae are positively associated with low stream temperatures, high water velocity, steep gradients, and the presence of riffles, waterfalls, and cobble and boulder substrates (Hawkins et al. 1988, Bury et al. 1991, Welsh and Ollivier 1998, Diller and Wallace 1999, Adams and Bury 2002, Welsh and Lind 2002, Wahbe and Bunnell 2003). Larvae are negatively associated with fine sediment load (i.e., embeddedness), pools, and slow-flowing stream habitat (Hawkins et al. 1988, Corn and Bury 1989, Welsh and Ollivier 1998, Diller and Wallace 1999, Welsh and Hodgson 2008). Steep gradients allow for flushing of fine sediments, although gradient effects may be more pronounced in harvested compared to primary forest habitat (Corn and Bury 1989). Adults are positively associated with high rainfall, moist forest habitats, and pool habitat, and negatively associated with fine sediment loads (Welsh and Lind 2002, Ashton et al. 2006). Adults and larvae in the Mattole Watershed were restricted to headwater channels, and canopy closure was the best single predictor of *A. truei* presence (Welsh and Hodgson 2011). *Ascaphus truei* were never detected in streams where canopy closure was less than 83% (Welsh and Hodgson 2011).

Some researchers have suggested a positive association between *A. truei* and the presence of harder, more consolidated parent geologies because they produce less sediment (Diller and Wallace 1999, Dupuis et al. 2000, Wilkins and Peterson 2000). However, *A. truei* does occur in streams with unconsolidated geologies, such as those derived from marine sediments, particularly in areas not subjected to recent or historical anthropogenic disturbance (e.g., Adams and Bury 2002, Welsh and Lind 2002, Ashton et al. 2006). The absence of *A. truei* from some streams with unconsolidated geologies may be because the presence of easily erodable substrates exacerbates the impacts of habitat disturbance, which can have long-lasting effects (Adams and Bury 2002, Welsh and Lind 2002, Ashton et al. 2006).

Ascaphus truei is extremely sensitive to warm temperatures at all life stages. Eggs have a temperature tolerance range from 5°C to 18.5°C (Brown 1975a). The critical thermal maximum range for larvae is 28.9–30.1°C, and larvae avoided temperatures above 22°C in laboratory trials (de Vlaming and Bury 1970). First-year larvae collected from Del Norte County selected temperatures below 10°C along a thermal gradient in the laboratory, while second-year larvae selected temperatures closer to 15°C (de Vlaming and Bury 1970). The critical thermal maxima for adults ranged on average from 27.6°C to 29.6°C (data from *A. montanus*; Claussen 1973). Field temperatures at occupied sites are usually well below these limits, with larvae occurring in streams with a mean of 11.6°C (range 5.7–15.8°C; Welsh and Hodgson 2008).

In addition to narrow thermal tolerances, *A. truei* is also extremely sensitive to desiccation (Brattstrom 1963), which may limit adult use of upland habitat to periods of wet weather conditions (Nussbaum et al. 1983). One mark-recapture study in Humboldt County documented movements of only 0–30 m along the stream channel over a two-year period (Burkholder and Diller 2007). However, recapture probabilities were low, and some animals may have moved beyond the study area. Longer distance movements have been documented from populations outside of California, from tens of meters up to 400 m into upland habitat (McComb et al. 1993, Gomez and Anthony 1996, Vesely 1996, Wahbe et al. 2004, Matsuda and Richardson 2005). Seasonal variation in adult location in managed forests in Washington was hypothesized to be a localized breeding migration, with downstream movements for oviposition and a return upstream in late summer (Hayes et al. 2006). It is unknown whether similar movements also occur in older, less disturbed forests in the area. In an *A. montanus* population in Montana, seasonal movements may be due to behavioral thermoregulation (Adams and Frissell 2001).

Distribution (Past and Present)

Ascaphus truei ranges from British Columbia to northern California, mostly west of the Cascades Mountains (Stebbins 2003). California is the southern limit of the range, with *A. truei* occurring south from the Oregon border along the coast to Mendocino County and east to Shasta County (Grinnell and Camp 1917, Mittleman and Myers 1949, Salt 1952, Bury et al. 1969, Welsh 1985). *Ascaphus truei* ranges from near sea level in Humboldt County up to 2150 m in the Trinity Alps (J. Garwood, pers. comm.).

Random sampling of streams has documented higher occupancy rates for *A. truei* in unmanaged or older forests compared to managed or younger stands (Welsh 1990). We therefore assume that some historically occupied localities are no longer occupied due to disturbance. In one study in the Mattole Watershed in Mendocino and Humboldt counties, *A. truei* was present in 71% of streams in old and mature forests, but was not found in second growth forests (Welsh et al. 2005). Further studies in the Mattole Watershed have found *A. truei* in 67% (14/21) of streams in unmanaged forests, but only in 4% (1/28) of streams in managed stands (H. Welsh and G. Hodgson, unpublished data). Streams with mixed harvest histories in the South Fork of the Trinity River had an intermediate level of occupancy, with 28% of streams occupied (17/60; Welsh et al. 2010). Studies from outside of California also indicate that *A. truei* is present in a greater proportion of streams in unmanaged forests (Bury and Corn 1988, Corn and Bury 1989, Hayes et al. 2006). A survey of streams in private, managed timber lands all less than 80 years old along the northern California coast found stream occupancy rates of 37% (18/49) at the level of 30 m sampling reaches and 76% (54/72) at the level of entire stream reaches (Diller and Wallace 1999). The relatively high occupancy rates in these young forests are thought to be due to the ameliorating effect of maritime climate, as most sites were within 30 km of the coast (Bury 1968, Diller and Wallace 1999).

Trends in Abundance

Ascaphus truei tends to be lower in abundance in managed compared to unmanaged forest stands (Bury and Corn 1988, Corn and Bury 1989, Welsh 1990, Gomez and Anthony 1996, Welsh and Lind 2002, Ashton et al. 2006). Clear-cuts can have immediate effects on abundance. Larval densities were higher in late-succession and old-growth forests compared to adjacent clear-cuts lacking streamside buffers in Oregon and British Columbia (Dupuis and Steventon 1999, Biek et al. 2002). Upland pitfall trapping in clear-cuts and mature forests in British Columbia found similar total numbers of *A. truei* in both forest types, but very few adults in clear-cuts, suggesting that immature frogs in clear-cuts are transients or incur high mortality rates (Matsuda and Richardson 2005). Several researchers have predicted declines or continuing declines if anthropogenic disturbances continue (e.g., Corn and Bury 1989, Dupuis and Steventon 1999, Welsh and Lind 2002, Ashton et al. 2006, Olson et al. 2007).

Nature and Degree of Threat

Declines and local extirpations to date are largely due to land management including timber harvesting and road construction (Welsh and Ollivier 1998, Welsh et al. 2005). Marijuana cultivation and climate change are also emerging as potential threats to this taxon.

The mechanisms underlying declines and extirpations due to timber harvesting and road construction are primarily increased sedimentation, increased stream temperatures, and fragmentation. While the initial impacts of road construction may be relatively short-lived, longer-term impacts are caused by sedimentation due to runoff from poorly maintained dirt and gravel roads (L. Diller, pers. comm.). Reduced canopy cover does not seem to increase temperatures as much at high-elevation sites, and *Ascaphus truei* may be more resilient to timber harvesting in areas where stream temperature is cooler due to overall climate (e.g., Diller and Wallace 1999, Wahbe and Bunnell 2003). Reductions in canopy or riparian vegetation that result in increased light levels may cause shifts in the algal community (i.e., from diatoms to filamentous green algae) that negatively affect the quality and abundance of larval food (L. Diller, pers. comm.). Landscape genetic studies in Washington suggest that significant overland dispersal occurs through terrestrial habitat, with gene flow detected between populations on a scale of 25–30 km (Spear and Storfer 2008). While timber harvests have some initial effect on gene flow, it may take multiple generations before the effects of fragmentation on population genetic structure can be detected.

An emerging threat to *A. truei* is large-scale marijuana cultivation, though little data is currently available due to limited accessibility of private lands. Similar to timber harvesting, marijuana cultivation requires clearing land and building roads which can increase sedimentation. Contamination from pesticides used on marijuana grows has been documented to negatively affect mammals in the field (Thompson et al. 2014), and amphibians are likely to be susceptible as well because of their permeable skin. Of particular concern for headwater amphibians like *A. truei* is the dewatering of waterways that are diverted for irrigation (CDFG 2013).

Climate change poses potential risks to *A. truei* 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). The frequency of extremely hot days is projected to increase, with roughly nine additional days over 32.2°C (Bell et al. 2004). Such temperatures exceed the critical thermal maxima for all life stages of *A. truei*, though 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 areas 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, some modest decreases, and some 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 and may lead to a mismatch between the timing of breeding and appropriate stream conditions. 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 (Lenihan et al. 2008). Westerling et al. (2011) projected a 100% increase in area burned in northwestern California under some scenarios. Direct mortality of adults and larvae due to fire has been documented in *A. montanus* populations (P. Van Eimeren, pers. comm., in Pilliod et al. 2003, Hossack et al. 2006). Short-term impacts of fire may be due to warmer temperatures and/or increased ammonia levels or other changes to water chemistry (Pilliod et al. 2003), but long-term impacts are understudied. 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 *A. truei* because stream conditions and forest age seem to be more important indicators of habitat quality than forest type.

Status Determination

Ascaphus truei is a specialist of cold, headwater stream habitats in old and mature forests, a habitat type that incurs substantial disturbance from land management activities. Declines in distribution and abundance have been documented in response to anthropogenic disturbances, and climate change has the potential to further negatively impact this species. These factors all contribute to a Priority 2 designation for this species.

Management Recommendations

Remaining old and mature forest habitats should be protected, with a focus on managing 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 preserve stream network processes (Olson et al. 2007). 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). 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 in the California range of *Ascaphus truei*.

Construction of new roads should be minimized or avoided in areas where protecting *A. truei* 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.

Ascaphus truei management would benefit from greater legal clarity regarding state and federal law on marijuana cultivation in California. Currently, some cultivation is legal under state law but prohibited under federal law, which may be hampering regulation of cultivation sites. Greater enforcement of existing environmental and land use laws is needed, and development of additional regulations should consider environmental impacts on *A. truei*.

Monitoring, Research, and Survey Needs

The presence of uncut streamside buffers on the entire channel network can ameliorate the impacts of land management on *Ascaphus truei* populations, but more research is needed into optimum buffer widths as they relate to different life history requirements and different portions of the catchment network. Studies from *A. truei* populations in British Columbia and Oregon have found positive effects of buffers 5–60 m wide (Bull and Carter 1996, Dupuis and Steventon 1999, Stoddard and Hayes 2005, Pollett et al. 2010). Experiments to determine optimal buffer widths in California habitats are needed. We recommend, at a minimum, that comparative data from coastal Mendocino County (the southern limit of the species range), coastal Humboldt/Del Norte Counties (the northern limit of the species range in California), and inland Trinity County are needed to assess the minimum forest buffer on industrial timber lands to retain key temperature and stream clarity conditions for *A. truei*.

Much of the research on *A. truei* has focused on stream-breeding habitat and presence/absence studies. While more difficult, monitoring efforts to document abundance and population dynamics are needed to gain insight into declines that cannot be inferred from presence/

absence surveys (Welsh 2011). Such studies could also determine which life history stages limit population growth in this species. When possible, population estimates in managed forests should be compared to *A. truei* abundance in nearby undisturbed mature forest stands (i.e., reference populations) to assess the impacts of disturbance (Welsh 2011).

More studies are needed on use of upland habitats by adults and dispersing animals. Such studies should be targeted at identifying terrestrial habitat corridors, if present, which can then be protected to maintain connectivity among populations (Olson et al. 2007, Olson and Burnett 2009). Landscape genetic analyses from replicate California populations may be particularly informative, given that recent studies from different parts of the range reach different conclusions about population connectivity (Spear and Storfer 2008, Spear and Storfer 2010, Aguilar et al. 2013).

Field research on impacts of marijuana cultivation on amphibian populations would contribute to development of environmental regulations for this growing industry and inform management strategies in cultivated areas.

SONORAN DESERT TOAD

Bufo alvarius Girard 1859

Status Summary

Bufo alvarius is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 75% (64/85). During the previous evaluation, it was also designated as a Species of Special Concern (Jennings and Hayes 1994a). The species has not been found in California since 1955 (but see the "Distribution" section).

Identification

Bufo alvarius is a large (10.1–19.0 cm SVL) olive, brown, or gray toad with prominent cranial crests and large elongate paratoid glands (Stebbins 2003). The skin is smoother than in other North American toads, with few warts along the dorsum. *Bufo alvarius* has large warts on the hind limbs and prominent white warts at the corners of the mouth (Stebbins 2003). The call is a low-pitched bleat or screech (Elliott et al. 2009).

This species is unlikely to be confused with any other anuran within its California range. All other true toads (family Bufonidae) in the region have extensive warts over the entire dorsal surface and lack large warts on the hind legs. The spadefoots (*Scaphiopus* and *Spea*, family Scaphiopodidae) are much smaller as adults

Sonoran Desert Toad: 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)	Data deficient
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	10
Total Score	64
Total Possible	85
Total Score/Total Possible	0.75

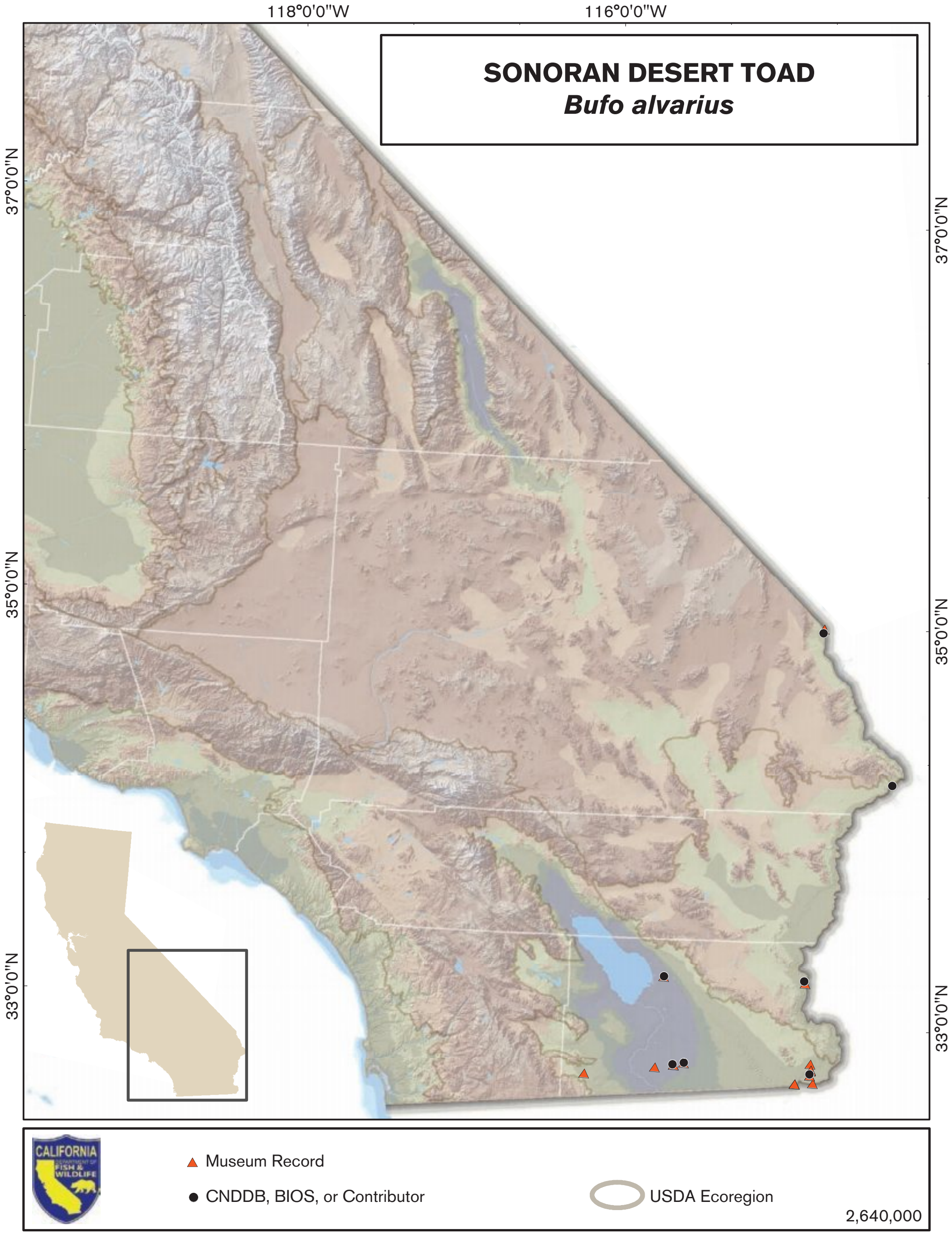


PHOTO ON PREVIOUS PAGE: Sonoran Desert toad, Cochise County, Arizona. Courtesy of Rob Schell Photography.

and have a conspicuous black keratinized spade on the ventral surface of the rear feet.

Taxonomic Relationships

The validity of this taxon has never been questioned, although confusion about the type specimen and locality has been discussed (Fouquette 1968). Osteological and genetic data, as well as call characteristics, suggest that it is related to Central American bufonids (Tihen 1962, Martin 1972, Sullivan and Malmos 1994, Pauly et al. 2004, Frost et al. 2006a).

Frost et al. (2006a) suggested a taxonomic revision that placed this species in the genus *Cranopsis*. Based on subsequent discoveries of older available names, the genus name for this taxon was later revised to *Ollotis* (Frost et al. 2006b) and then *Incilius* (Frost et al. 2009b, Pauly et al. 2009). We retain the older taxonomy both for taxonomic stability and because the analyses supporting the original rearrangement are controversial (Crother 2009, Frost et al. 2009a, Pauly et al. 2009).

Life History

The life history of this species in California is unknown, and we base the following discussion on observations from other areas. *Bufo alvarius* spends much of the year underground, presumably in rodent burrows (Degenhardt et al. 1996). *Bufo alvarius* is primarily nocturnal and becomes active before summer rains. It is more strongly aquatic than most North American toads (Stebbins 1951). Breeding behavior appears to be generally associated with summer rains (Sullivan and Malmos 1994), although amplexus has been reported in stock ponds before rains have occurred (Degenhardt et al. 1996). Several years may pass between breeding events depending on the presence of sufficient rainfall (Sullivan and Fernandez 1999). The species sometimes congregates in large numbers for breeding, with nearly all reproduction of a local breeding population occurring in a single night (Degenhardt et al. 1996). The time required for hatching and metamorphosis is unknown but may be less than a month (notes of Thornber, reported in Ruthven 1907 and Storer 1925). This species appears to be a dietary generalist, feeding on any live arthropod or small vertebrate prey that it can successfully capture (Stebbins 1951, Cole 1962). Poison secreted by the skin and paratoid glands is particularly toxic and has caused death and paralysis in dogs and is a potent hallucinogen in humans (Musgrave 1930, Stebbins 1951, Stebbins 2003).

Habitat Requirements

The habitat requirements for *Bufo alvarius* in California are unknown. In arid habitats of Arizona and New Mexico, the species can be found in and around a variety of water sources used for breeding, including springs, stock ponds, washes, river bottoms, and irrigation ditches (Stebbins 1951, Stebbins 1972), though it is occasionally found at great distances (>1 mi) from water (Slevin 1928). Upland habitat surrounding known aquatic breeding localities elsewhere in the range include mesquite-creosote desert lowland, arid grassland, rocky riparian zones, oak–sycamore–walnut assemblages in mountain canyons, and montane pine–oak–juniper plant communities (Stebbins 2003, Fouquette et al. 2005).

Distribution (Past and Present)

There are no known extant populations in California. Historically, the species ranged in California along the Lower Colorado River and into the Imperial Valley (Grinnell and Camp 1917, Stebbins 1951, Jennings and Hayes 1994a), likely ranging as far north as the southern tip of Nevada (Cooper 1869). It is not known if records in the Imperial Valley are a natural part of the historic range or whether they represent recent range expansion following the development of irrigation (Stebbins 1951).

The last verified record (LACM 87044) from California dates to 31 July 1955, 7 km north of Winterhaven. More recent surveys have failed to detect the species (King and Robbins 1991, Jennings and Hayes 1994b). Sporadic records continue to be reported on the Arizona side of the Colorado River, however. Several individuals were found near the Cibola National Wildlife Refuge in 1980 (Anderson and Ohmart 1982; B. Anderson, pers. comm.), and a single individual was found at the refuge itself in 1986 (J. Rorabaugh, pers. comm.). On 1 July 2004, a large individual was found "by the golf course on the Parker Strip," La Paz County, Arizona (J. Rorabaugh, pers. comm.). On 29 July 2009, an amplexing pair was found along the Bill Williams River at Planet Ranch, Mohave County, Arizona, and the species is reportedly "fairly common" 24–32 km above the confluence of the Gila and Colorado Rivers, Yuma County, Arizona (J. Rorabaugh, pers. comm.). A single, unverified record of a calling *Bufo alvarius* was reported near Bard, California, in the spring of 2007 or 2008, though the time of year was unexpected and the observer was inexperienced with the species (J. Rorabaugh, pers. comm.).

Outside of California, *B. alvarius* ranges across southern Arizona to the southwestern corner of New Mexico and south into Sonora and the northern edge of Sinaloa, Mexico. The known elevational range extends from near sea level to 1615 m (Cole 1962).

Trends in Abundance

Though the paucity of records from California makes assessing former abundance difficult, *Bufo alvarius* was apparently common at Yuma, Arizona, on the California border, along the Lower Colorado River, and in parts of the Imperial Valley (Slevin 1928, Klauber 1934). As no populations are currently known in these areas, declines leading to probable population extirpations or extremely low population sizes have clearly occurred. The species is also known to be declining in New Mexico (Degenhardt et al. 1996). Throughout the rest of the range the species appears to be stable and abundant at many localities (Fouquette et al. 2005, Lazaroff et al. 2006), though some have suggested that declines are occurring throughout the range (B. Brattstrom, R. Ruibal, and C. Schwalbe, pers. comms., reported in Jennings and Hayes 1994a).

Nature and Degree of Threat

The causes of declines, and therefore the threats to this species, are poorly understood. Declines occurred before any studies were carried out in California, though it is likely that landscape modification and pesticide applications that occurred with the growth of agriculture in the Imperial Valley contributed to declines (Ohmart et al. 1988, Jennings and Hayes 1994a). Bufonids are generally very susceptible to amphibian declines (Stuart et al. 2004). In California, toad declines have been linked to habitat loss and pesticide use (Davidson et al. 2002) and pathogenic fungi (Green and Kagarise Sherman 2001).

Status Determination

The declines and possible extirpation of *Bufo alvarius* in California are the primary concerns for this taxon. The species may require permanent aquatic environments making it a moderate ecological specialist, given the arid environments that characterize its range. This also makes the taxon sensitive to the effects of climate change, particularly changes in hydrology and the increasing year-to-year variation in precipitation that have been projected (Cayan et al. 2008b). Finally, because little understanding of the causes of declines in California exists, we are poorly positioned to protect any remaining populations should they be found in future surveys.

Management Recommendations

The development of an effective management strategy for *Bufo alvarius* in California is not possible without further distributional and ecological information. As no populations are currently known, the first management priority should be to undertake comprehensive surveys, as described below, aimed at identifying remaining fragmentary California populations. Habitat protection and enhancement would then become the critical management tools to build these populations to larger and viable sizes. Simultaneous ecological research is also needed on habitat use, home range size, life history, and population connectivity before more complex management programs focused on reestablishing the species are considered.

Monitoring, Research, and Survey Needs

A critical first step toward developing a comprehensive management plan for this species is to undertake comprehensive surveys of remaining potential habitat in southeastern California. These surveys should take place during the summer rains and should involve biologists who are familiar with *Bufo alvarius*' breeding behavior. If any remaining populations are found, a population-monitoring program should rapidly be established to determine both geographical extent and population size. As little is known about this species in California, this monitoring program should take place in conjunction with a study of the species' life history and habitat use, in California and/or adjacent Arizona. These surveys should specifically target the remaining moist areas of the southwest California deserts that are known to support other water-dependant vertebrate species, such as the desert mule deer (*Odocoileus hemionus crooki*). Using existing survey data from other, and better known, species may help to guide toad survey efforts toward the wettest areas or most consistent water supplies, thereby increasing odds of detection.

A second critical priority is to work with wildlife managers in Arizona to survey for and study the nearest remaining populations on the Arizona side of the Lower Colorado River. These populations are likely the most ecologically similar to the former California populations and should therefore provide information valuable to the eventual development of management programs in California. Genetic samples from both California and Arizona should be collected to help inform managers about levels of genetic differentiation, and therefore the appropriateness of possible reintroduction of Arizona animals to California.

As any populations that remain in California are likely isolated, study of these populations is unlikely to yield information on the metapopulation dynamics that we presume are key in sustaining this species elsewhere. Reestablishing these dynamics would form an important part of a comprehensive management program in California, and research focused on better understanding these dynamics will also need to take place outside of California, preferably in adjacent Arizona.

ARROYO TOAD

Bufo californicus Camp 1915

Status Summary

Bufo californicus is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 93% (102/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a), and it has been listed as federally Endangered since 1995.

Identification

Bufo californicus is a small to medium-sized (4.6–8.6 cm SVL), light-gray to tannish-brown toad that often has some greenish or olive and dark-brown mottling on the back and sides (Camp 1915, Stebbins 2003). The underside is buff or dirty white and usually unmarked (Stebbins 2003). A light middorsal stripe is rarely present (Jennings and Hayes 1994a, Stebbins 2003). Weak cranial crests are often present and the paratoids are oval-shaped and widely separated (Stebbins 2003). The advertisement call of this species is a musical trill that lasts 3–10 s. The pitch of the call rises quickly and is held constant for the remainder of the call, which ends abruptly (Stebbins 2003, Elliott et al. 2009). Like most toads, the tadpoles are small and black early in life. However, several weeks post-hatching they develop a cryptic tan coloration that closely

Arroyo Toad: Risk Factors

Ranking Criteria (Maximum Score)	Score
i. Range size (10)	10
ii. Distribution trend (25)	25
iii. Population concentration/ migration (10)	10
iv. Endemism (10)	7
v. Ecological tolerance (10)	10
vi. Population trend (25)	20
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	10
Total Score	102
Total Possible	110
Total Score/Total Possible	0.93

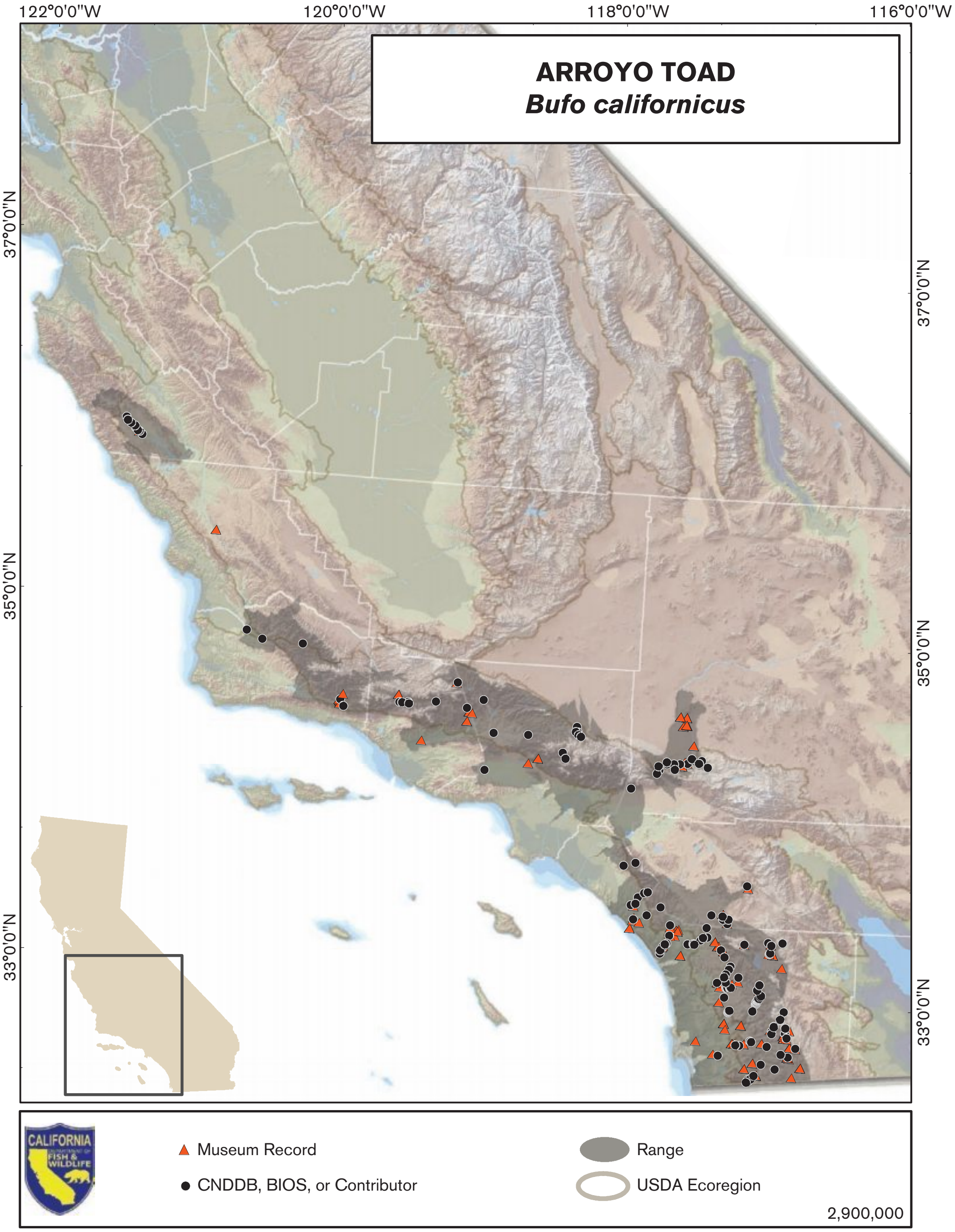


PHOTO ON PREVIOUS PAGE: Arroyo toad, Baja California, Mexico. Courtesy of Rob Schell Photography.

matches the substrate (Sweet 1992, Hancock 2009).

Metamorphosed individuals of this species may be confused with the western toad (*B. boreas*), which is the only sympatrically occurring toad. *Bufo boreas* has a prominent white or cream dorsal stripe and lacks cranial crests (Stebbins 2003). Young tadpoles that still retain the black coloration are difficult to distinguish from *B. boreas*, but older tadpoles are readily distinguishable.

Taxonomic Relationships

Until recently, *Bufo californicus* was considered a subspecies of the Arizona toad (*B. microscaphus*) (Price and Sullivan 1988), although recommendations to recognize it as a full species date back to Myers (1930). Frost and Hillis (1990) recognized this species as distinctive based on the general observation that few allopatrically distributed polytypic species represent single genetically cohesive units, as is implied by retaining *B. californicus* as a subspecies under *B. microscaphus*. Later analyses of allozyme data confirm that *B. californicus* is a distinct lineage, providing support for its recognition as a full species (Gergus 1998). Additional analyses of advertisement calls indicated a substantial amount of variation within the species complex, although the results were equivocal with respect to species status (Gergus et al. 1997). Lovich (2009a) analyzed data from two mitochondrial genes and found additional evidence that *B. californicus* is a distinct species. This work also identified clades within *B. californicus* that roughly correspond to parts of the range north and south of the Los Angeles Basin, respectively.

Frost et al. (2006a) recommended placing this species and many other North American bufonids in the genus *Anaxyrus,* although this proposal and the analyses that support it are controversial (Crother 2009, Frost et al. 2009a, Pauly et al. 2009). We choose not to follow this recommendation at the present time, pending further analyses, and to maintain taxonomic stability.

Life History

Bufo californicus is primarily nocturnal and feeds predominantly on nocturnally active ant species (Cunningham 1962, Sweet 1992, Sweet 1993, Mahrdt et al. 2002). Adults typically emerge from retreats approximately 30–40 min after sunset, remaining active down to temperatures of around 13°C on dry nights and 10°C on rainy nights, with nocturnal activity increased during wet periods (Cunningham 1962, Sweet 1992, Sweet 1993).

Males begin calling at varying times of the year depending on local conditions and elevation, although calling activity appears to initiate when water temperatures reach or exceed 11–13°C (Myers 1930, Sweet 1992). Choruses generally begin in late February in coastal populations and late March or April at higher elevation inland sites, and they may continue into July (Sweet 1992, Sweet 1993, Stebbins 2003, Hancock 2009). Eggs are laid near the male's calling site on a substrate of mud, sand, or gravel, away from vegetation and other submerged debris (Sweet 1992). Hatching occurs after 4–6 days at typical water temperatures (12–16°C), although the larvae remain associated with the egg mass for an additional 5–6 days. Metamorphosis can occur in as few as 65 days, although typically 72–80 days are required (Sweet 1992, Hancock 2009). Larger males and females are more sedentary and tend to breed in the same pools throughout the reproductive season and from year to year (Sweet 1993, Hancock 2009, Mitrovich et al. 2011). The seasonal activity period for adults extends roughly from the beginning of the breeding season to late June or July, after which most toads become inactive (Cunningham 1962, Sweet 1993, Hancock 2009). Juveniles may remain active into October or later following rains (Sweet 1993).

Bufo californicus usually attain reproductive condition in their second (males) or third (females) year. The species is relatively short-lived, with few toads living beyond 5 years of age. In the absence of nonnatural disturbances, survivorship of adult toads is high during the active season, but decreases markedly during the inactive season. Sweet (1993) documented that toads experience ~55% per year mortality mostly during the winter, though other estimates suggest even higher mortality (D. Holland and N. Sisk, unpublished data, reported in Sweet and Sullivan 2005). Eggs and young larvae are apparently unpalatable to most predators, although garter snakes and nonnative fishes prey upon older tadpoles (Sweet 1992). Juvenile toads that have not yet adopted the nocturnal activity pattern characteristic of adults also experience high predation pressure (Hancock 2009). Adult toads experience intense predation from introduced bullfrogs in areas where that species occurs (Miller et al. 2012, R. Fisher pers., comm.). In the absence of bullfrogs, adult toads experience much lower predation intensity (Sweet 1993, Hancock 2009).

Habitat Requirements

Along with its close relative *Bufo microscaphus*, *B. californicus* may have the most specialized habitat requirements of any North American anuran (Stebbins 2003). This species requires shallow, slow-moving stream and riparian habitat. In some areas they may occupy first-order streams, although most populations inhabit second- to sixth-order streams that have extensive braided channels and sediment deposits of sand, gravel, or pebbles that are occasionally reworked by flooding (Sweet and Sullivan 2005). These toads will use either permanent or seasonal streams, although seasonal streams must flow for a minimum 4–5 months for successful reproduction and recruitment (Sweet and Sullivan 2005). At inland sites, radiotelemetry studies indicate that this species rarely moves beyond the immediate upland margin of streams, although in coastal sites arroyo toads appear to occasionally use and disperse across hotter and drier upland sites (Sweet 1992, Sweet 1993, Griffin and Case 2001, Hancock 2009, Mitrovich et al. 2011). Mitrovich et al. (2011) found that radio-tracked toads actively selected channel and terrace stream habitats, and largely avoided surrounding scrub, grassland, and forest. On average, males were found in closer proximity to flowing sections of stream than females, possibly to maximize reproductive opportunity (Mitrovich et al. 2011). *Bufo californicus* is known to occasionally use and breed in human-made habitats, such as artificial stream terraces and ponds (Price and Sullivan 1988, Mahrdt et al. 2002). It is unknown whether the species can persist in these habitats.

Distribution (Past and Present)

Bufo californicus historically occurred in coastal drainages from the San Antonio River, Monterey County, California, southward through the Transverse and Peninsular Ranges to the vicinity of Arroyo San Simón in Baja California Norte, Mexico (Price and Sullivan 1988, Gergus et al. 1997, Grismer 2002, Lovich 2009a). Almost all populations occur along the coast or on the coastal slopes of the southern California mountains. Six localities were previously recognized from the desert slopes of Los Angeles, Riverside, San Bernardino, and San Diego Counties, California (Patten and Myers 1992, Jennings and Hayes 1994a). Desert slope populations are known to occur at Little Rock Creek, Los Angeles County, and the Mojave River, San Bernadino County. Populations at Whitewater River, Riverside County, Borrego Springs (listed as San Felipe Creek in Jennings and Hayes 1994a), Vallecito Creek, and Pinto Canyon, San Diego County, are probably in error and are the result of misidentifications (Ervin et al. 2013). The known elevational range extends from near sea level to approximately 1000 m (Stebbins 2003; S. Sweet, pers. comm.).

The present distribution of *B. californicus* is considerably smaller than it once was. Jennings and Hayes (1994a) estimated that this species had disappeared from 76% of its former range in California, although more recent estimates place this loss at 65% (Sweet and Sullivan 2005).

Trends in Abundance

In addition to the extirpations discussed above, extensive declines in abundance have been documented in most *Bufo californicus* populations that do survive. Extensive collections from the 1930s, largely stemming from the work of L.M. Klauber, suggest that this species was formerly present at much higher densities (S. Sweet, pers. obs., reported in Sweet and Sullivan 2005).

Nature and Degree of Threat

A recent 5-year review of the status of *Bufo californicus* thoroughly discusses the ongoing threats to this taxon (USFWS 2009). We follow the findings of that document and recommend that readers consult it for additional detail.

The greatest threat facing this taxon is loss and degradation of habitat that stems from modifications to hydrology from reservoir construction, roads, flood control, development, recreational activity, and mining (USFWS 2009). In addition, declines are occurring even in areas that are not subject to development and direct habitat degradation from human activities (Hancock 2009). These additional declines stem largely from introduced predators (primarily bullfrogs and green sunfish) and introduced plants, which degrade habitat and/or decrease survivorship of toads (Sweet 1992, Hancock 2009, USFWS 2009, Miller et. al. 2012). Off-highway vehicle use has also caused both habitat degradation and direct mortality in this species (Ervin et al. 2006)

Status Determination

Major declines in both distribution and abundance, coupled with several ongoing threats, combine to warrant a Priority 1 Species of Special Concern status for *Bufo californicus*.

Management Recommendations

Management efforts for *Bufo californicus* should mirror those outlined by the USFWS recovery plan and 5-year review for this taxon (USFWS 1999, USFWS 2009). The recent 5-year review suggests that management efforts to date have been effective, and the outlook for this species has improved somewhat since it was initially listed (USFWS 2009). The most important management strategy is to preserve existing stream habitat that supports this species and to restore additional habitat that can support self-sustaining populations. Restoration efforts should include dam removal to allow streams to meander and rebuild sand and gravel bars, and removal of exotic plants and vertebrate predators.

Monitoring, Research, and Survey Needs

Monitoring, research, and survey needs are covered in depth in the USFWS recovery plan for this taxon and the recent 5-year review. We refer the reader to these documents for additional detail (USFWS 1999, USFWS 2009). Monitoring efforts should focus on recovering populations, particularly those in newly restored habitat. It is particularly important to continue monitoring through drought and El Niño cycles given that this is a short-lived species and several years of consistent drought could be extremely damaging to recovering populations.

In addition, research aimed at characterizing variation in this species' life history in different parts of its range should be undertaken, as these differences might have an impact on future management efforts. For example, the two desert slope populations may differ substantially in several aspects of life history relative to the coastal slope populations. Additional research into the prevalence and potential impacts of *Bd* fungus on this species is also particularly important. Finally, molecular analyses of population size and connectivity might be particularly valuable in this taxon.

YOSEMITE TOAD

Bufo canorus Camp 1916a

Status Summary

Bufo canorus is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 84% (92/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a). This species is also listed as Threatened under the US Endangered Species Act.

Identification

Bufo canorus is a moderately sized (1.0–7 cm SVL) sexually dichromatic toad (Camp 1916a, Grinnell and Storer 1924, Kagarise Sherman 1980, Stebbins 2003). Females and juveniles have tan or brown dorsal coloration with extensive dark blotches over the dorsal surface and legs that are edged with white or cream. Males are pale green-yellow or olive green above without, or with only minimal, dark blotching or flecking (Camp 1916a, Stebbins 2003). A pale, very thin middorsal line is often present in juveniles and young females, but is usually lost in males and older females (Camp 1916a). The paratoid glands are large, flat, and circular, and are separated by a space less than or equal to their diameter (Camp 1916a). The advertisement call of this species is a musical trill lasting 3–9 s (Elliott et al. 2009). The specific epithet "*canorus*" (Latin for

Yosemite Toad: 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)	20
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	92
Total Possible	110
Total Score/Total Possible	0.84

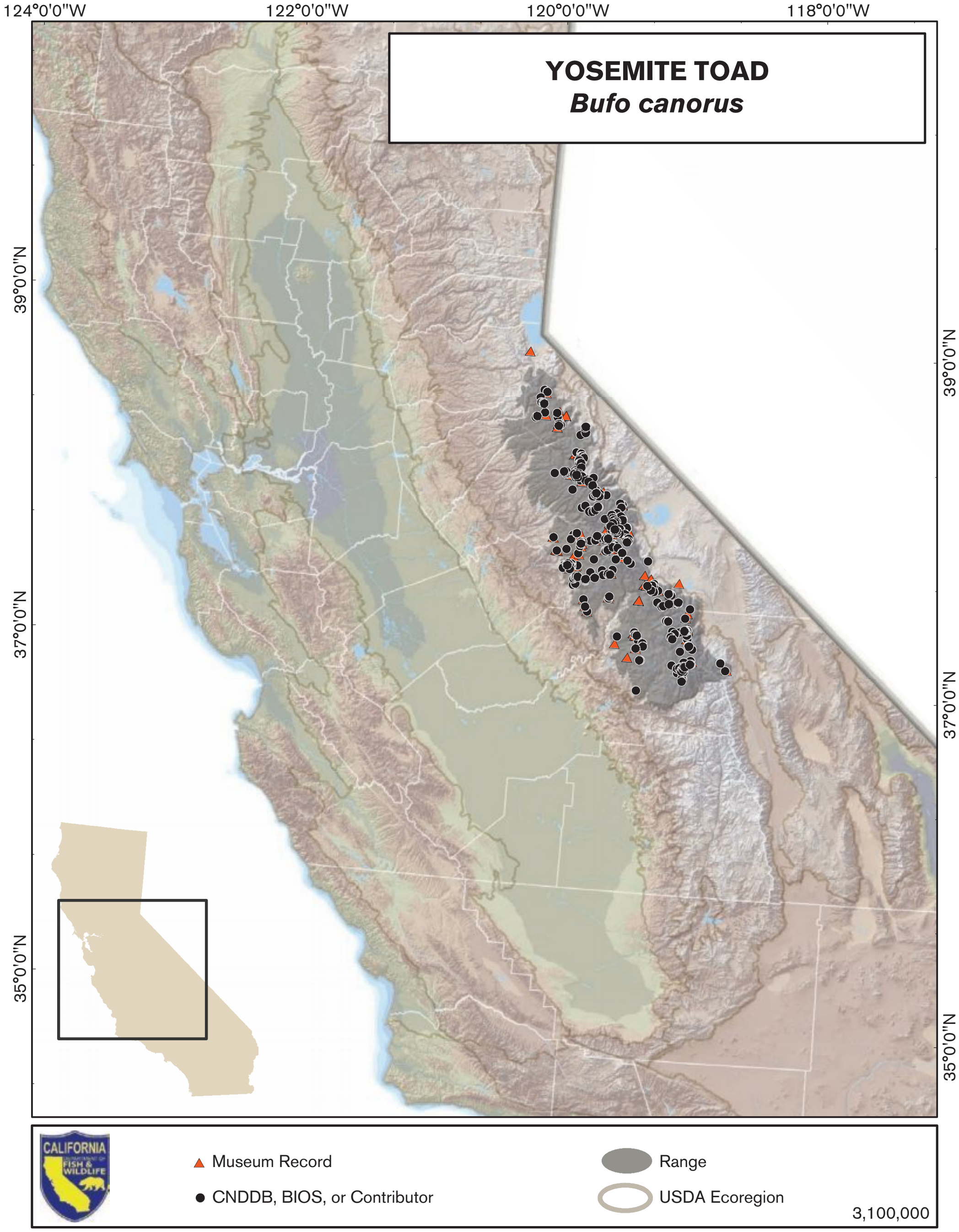


PHOTO ON PREVIOUS PAGE: Yosemite toad, Mono County, California. Courtesy of Rob Grasso.

"tuneful") refers to the melodic quality of the call (Camp 1916a, Karlstrom 1962).

This species may be confused with high-elevation populations of the western toad (*B. boreas*). *Bufo boreas* often has a conspicuous light middorsal stripe and smaller, more widely spaced paratoid glands that are separated by a space approximately twice their diameter (Stebbins 2003). *Bufo boreas* also has more extensive webbing on the hind feet than *B. canorus* (Camp 1917). Populations of *B. boreas* that occur in the Sierra Nevada do not produce advertisement calls; thus, toad breeding choruses are diagnostic for *B. canorus* (Mullally 1956). Juveniles of the two species (<20–30 mm) are very similar to one another, and identifications of this size class should be made with caution (Karlstrom 1962).

Taxonomic Relationships

Bufo canorus was initially described on the basis of coloration and morphology (Camp 1916a). Its status as a distinct species has gone largely unquestioned since this time. Subsequent genetic analyses based on mitochondrial data suggest that this taxon is a close relative of *B. boreas* (Shaffer et al. 2000, Pauly et al. 2004). In addition, mitochondrial DNA data suggest that *B. canorus* may be paraphyletic with respect to the black toad, *B. exsul*, and some lineages of *B. boreas* (Graybeal 1993, Shaffer et al. 2000, Goebel et al. 2009). These relationships have not been corroborated with nuclear sequence data, and thus it is unclear whether cryptic diversity exists within the taxon or if this is a case of mitochondrial introgression. Unpublished genetic data suggest that mitochondrial introgression associated with past or ongoing hybridization may explain these results (G. Pauly, unpublished data). Some morphological variation has been observed in size and degree of melanism in eggs and larvae of this taxon (Karlstrom and Livezey 1955, Karlstrom 1962). Whether this is plasticity in response to elevation and/or the local environment or genetic differentiation has not been investigated further. Mitochondrial data also indicate that some intraspecific variation and isolation by distance may exist within the taxon (Shaffer et al. 2000, Pauly et al. 2004, Goebel et al. 2009), although sample sizes in these studies were small. Wang (2012) collected data from 10 microsatellites for toads from 24 populations in Yosemite National Park and found significant variation in the amount of genetic distance between populations. This analysis concluded that environmental factors such as slope and precipitation were associated with genetic structure.

Frost et al. (2006a) recommended placing this species and many other North American bufonids in the genus *Anaxyrus*, although this proposal and the analyses that support it are controversial (Crother 2009, Frost et al. 2009a, Pauly et al. 2009).

Life History

Bufo canorus is primarily a diurnal toad that occasionally exhibits crepuscular or nocturnal activity on warm days (Mullally 1956, Martin 2008). Males emerge from hibernation as soon as snowmelt pools form along the margins of preferred high-elevation meadow habitat and quickly form breeding choruses (Karlstrom 1962). As in many high-elevation amphibians, the timing of emergence is correlated with elevation, and generally occurs in May and June (Karlstrom 1962, Kagarise Sherman 1980). Males are territorial and often maintain interindividual spacing of 7–14 m (Karlstrom 1962, Kagarise Sherman 1980). Fighting occurs between males that encroach on one another's territory (Kagarise Sherman and Morton 1984). Breeding activity and egg-laying commence soon after males begin calling, with females depositing eggs along shallow edges of pools and streams in meadows (Karlstrom 1962, Kagarise Sherman 1980). Hatching occurs in as few as 3–4 days at relatively high water temperatures (20–23°C) or up to 10–14 days at lower temperatures (16–17°C) (Karlstrom 1962, Kagarise Sherman 1980). Metamorphosis occurs approximately 40–60 days after oviposition, again depending on temperature and elevation (Karlstrom 1962, Kagarise Sherman 1980, Kagarise Sherman and Morton 1984). The seasonal activity period extends into late September and early October, after which toads hibernate in rodent burrows, crevices under rocks, and root tangles (Kagarise Sherman 1980). Adult toads do not begin to breed until they are 3–6 years old, after which females may only breed every few years (Kagarise Sherman 1980, Kagarise Sherman and Morton 1984). Adults grow slowly, averaging only 2.5 mm per year at Tioga Pass, Tuolumne County, California (Kagarise Sherman and Morton 1984). The post-metamorphic diet consists of a variety of small arthropods including ants, bees, flies, wasps, beetles, millipedes, and spiders (Grinnell and Storer 1924, Mullally 1953, Kagarise Sherman and Morton 1984). The slow growth rate and lack of breeding every year is likely attributable to low metabolic rates associated with low caloric intake and relatively cold temperatures (Kagarise Sherman and Morton 1984). In the wild, adults appear to be able to tolerate a relatively wide range of temperatures (from 2°–30°C) (Karlstrom 1962), although they prefer temperatures higher in this range (Cunningham 1963). The estimated critical thermal maximum is 37–40°C for adults and 36–38°C for larvae (Karlstrom 1962).

Bufo canorus is known to occur sympatrically with *B. boreas* in two areas (see the "Distribution" section) and may occasionally hybridize. At the Frog Lakes locality, individuals that are morphologically intermediate in paratoid gland width and the extent of webbing on the hind feet occur and may represent natural hybrids (Morton and Sokolski 1978). No putative hybrids have been described from the Blue Lakes locality (Karlstrom 1962), although some authors suggest that hybridization may also occur there (Stebbins 2003). Artificial crosses in the laboratory readily produce hybrids (Karlstrom 1962).

Habitat Requirements

Bufo canorus prefers relatively open high-elevation meadows vegetated with grasses, sedges, rushes, and/or willow stands (Karlstrom 1962). This species can be found in the margins of water bodies that form from snowmelt runoff, as well as in moist meadows. During the early part of the active season, individuals are often localized along meadow margins within approximately 30 m of the forest edge. This behavior may allow them to easily retreat to forest cover at night to avoid freezing temperatures (Karlstrom 1962). As the active season progresses and nights become warmer, the toads tend to move toward the center of meadows and become less restricted to the margins (Karlstrom 1962). This species prefers shallow (probably <7.5 cm) snowmelt pools on the margins of meadows or very slow moving runoff streams in which to breed, although they have also been found in deeper (>3 m) permanent pools (G. Fellers, pers. comm.). These need to be deep enough to avoid premature desiccation—a significant cause of mortality for larvae—but shallow enough to achieve the temperatures needed for rapid development (Karlstrom 1962). This species may prefer to oviposit in dark-bottomed pools, particularly at high elevations, as these may provide warmer water temperatures and more rapid larval development (Karlstrom 1962). The presence of pocket gopher, mouse, and vole burrows may provide additional beneficial cover and protection from predation (Grinnell and Storer 1924, Karlstrom 1962).

Distribution (Past and Present)

Bufo canorus is restricted to a relatively small area approximately 240 km (north–south) by 60 km (east–west) in higher elevation areas of the Sierra Nevada (Karlstrom 1962, Kagarise Sherman and Morton 1993). It ranges from the vicinity of Blue Lakes, Alpine County, California, south past Kaiser Pass to the Evolution Lakes area, Fresno County, California (Grinnell and Storer 1924, Livezey 1955, Karlstrom 1962, Jennings and Hayes 1994a, Stebbins 2003, Davidson and Fellers 2005). The known elevational range extends from 1950 to 3599 m, with most localities between 2590 and 3048 m (Karlstrom 1962).

Between 1915 and 1992, this species exhibited declines throughout some areas of its range. Drost and Fellers (1996) resurveyed localities from Grinnell and Storer (1924) and found that this species had disappeared from 6 of 13 sites in the Yosemite area. Jennings and Hayes (1994a) also estimated that the species has disappeared from low-elevation areas on the western edge of the range, as well as at the northern edge of the range.

Trends in Abundance

In areas where *Bufo canorus* persists, marked declines in abundance have also been documented. In the Drost and Fellers (1996) resurveys, *B. canorus* was present in lower densities than in 1915 at three sites where it was still present. Between 1976 and 1982, the number of male toads entering breeding pools at Tioga Pass meadow declined from a maximum of 342 individuals to a low of 28, a ninefold decrease from the 1974–1978 mean (Kagarise Sherman and Morton 1993). However, the number of females entering breeding pools showed no obvious changes during this time period (Kagarise Sherman and Morton 1993). The average number of toads encountered in daily surveys also declined in the vicinity of Tioga Pass meadow between the early 1970s and 1990. In addition, these surveys documented declines in female toads, although they were not as severe as those documented in males (Kagarise Sherman and Morton 1993). Similar declines in abundance have also been documented at six additional localities in this region (Kagarise Sherman and Morton 1993).

Nature and Degree of Threat

The causes of decline in *Bufo canorus* require additional study. The declines have occurred in seemingly undisturbed areas and do not appear to be localized, suggesting that they are being driven by general changes to the environment, rather than localized causes such as habitat destruction. Several possible causes have been advanced, and more than one factor may be playing a role. These causes include environmental contamination, disease, drought and/or climate change, habitat modification due to grazing or other activities, human disturbance of breeding choruses, increased predation pressure from birds and fish, and pesticides. Based on current data it is not possible to understand in detail which, if any, of these factors are most important in *B. canorus* declines.

Snowmelt pools have extremely low acid neutralizing capacity, leading to the hypothesis that acidification of aquatic breeding habitat due to atmospheric deposition may be contributing to declines. Bradford et al. (1992) examined the effect of increasing acidification and the associated increase in dissolved aluminum on embryos and hatchlings of *B. canorus*. Embryos and hatchlings exposed to decreasing pH (and increasing aluminum solute) showed no increase in mortality at levels found in nature. However, these factors did cause earlier hatching and smaller body size at metamorphosis. Bradford et al. (1994) attempted to correlate the distribution of declining Sierran amphibians with these environmental factors and found no relationship, concluding that acid deposition was an unlikely source of amphibian declines in the Sierra Nevada.

Disease has also been considered as a factor in declines, though there is little evidence to date. Green and Kagarise Sherman (2001) examined the cause of death in 12 adult *B. canorus* that were found during a die-off that immediately preceded the population declines documented at Tioga Pass meadow by Kagarise Sherman and Morton (1993). They found that a variety of diseases and parasites were present in the population, and chytridiomycosis and septicemia, alone or in combination, caused the death of at least four individuals. However, no single infectious disease was present in more than 25% of the samples, which is far below the proportion typically observed in other die-offs caused by these diseases (Worthylake and Hovingh 1989, Berger et al. 1998, Vredenburg et al. 2010).

California experienced a relatively severe drought between 1987 and 1992, a time when *B. canorus* population declines were occurring (Roos 1992, Drost and Fellers 1996). Although it may have played an exacerbating role, drought alone seems unlikely to be responsible for declines. California experiences drought with some regularity, including during the time period of the Grinnell and Storer (1924) survey, which occurred before any major declines in *B. canorus* were observed (Drost and Fellers 1996). Drought does affect year-to-year reproductive success for this species, and prolonged drought may have a cumulative effect on populations (Kagarise Sherman and Morton 1993). Because climate change is expected to impact the amount of snow present in the Sierra Nevada and the speed and timing of snowmelt (Cayan et al. 2008b), drought might play an increasing role in declines of this species in the future.

Habitat modification is a leading cause of decline in many species throughout California and has been suggested as a factor for *B. canorus*. However, *B. canorus* is found largely within the boundaries of Yosemite National Park and other public (mostly National Forest) lands that have experienced varying impacts over the last 100 years. Drost and Fellers (1996) compared photos of habitat from the Grinnell and Storer (1924) surveys with current habitat and saw no apparent differences. Over the course of their 20-year study, Kagarise Sherman and Morton (1993) were also unable to detect significant habitat changes. That said, local impacts from changing habitat remain a potential driver of declines. All-terrain vehicle and snowmobile use in some localized areas may degrade habitat quality (D. Emery, pers. comm.). Some workers have postulated that livestock grazing in alpine meadows of National Forest land causes changes to hydrology, which may affect the suitability of breeding habitat and increase sedimentation in pools. Two recent, relatively short-term studies (5 years) have addressed this hypothesis: one that used experimental fencing treatments to exclude livestock from *B. canorus* breeding meadows and a second that included occupancy surveys across gradients of meadow moisture and livestock use levels (K. Tate and A. Lind, pers. comm.). Both of these studies demonstrated that meadow wetness was more influential in determining the current distribution and abundance of *B. canorus* than the level of livestock use. Sean Barry (pers. comm.) documented that toads seem to persist and even concentrate in areas that had been disturbed by cattle in the Kaiser Meadow population. It is also possible that the presence of cattle feces increases insect food supply for adult toads, although this remains untested. Martin (2008) suggests that the practice of fencing individual breeding pools to prevent grazing might actually lead to stronger habitat disturbance from cattle grazing in the terrestrial foraging habitats, potentially increasing the overall impact from grazing. These local-scale influences of livestock grazing along with more detailed and longer-term investigations of livestock use in the context of *B. canorus* metapopulation dynamics require further study.

Some researchers have suggested that increasing predation pressure could be causing declines. A variety of avian predators are known to feed on adult and larval toads, and increasing densities of common raven (*Corvus corax*) have been postulated as a possible cause of decline (Kagarise Sherman and Morton 1993). Ravens are known to prey upon other toad species in the *B. boreas* complex and likely also take *B. canorus*. Evidence suggests that declines in other amphibian species have occurred in areas where fish have been introduced (Drost and Fellers 1996). However, fish alone are unlikely to explain the declines in *B. canorus*. Most *B. canorus* reproduction takes place in ephemeral water bodies that do not contain fish (Drost and Fellers 1996). Knapp (2005) found no evidence for an effect of introduced trout on *B. canorus* presence and absence. Further, Grasso (2005) and Grasso et al. (2010) examined the palatability of early life stages of *B. canorus* to introduced brook trout and found that all life stages were highly unpalatable, suggesting that introduced trout may have little direct impact on populations.

Some authors have noted that breeding choruses of *B. canorus* are sensitive to human disturbance. Grinnell and Storer (1924) documented that choruses would abruptly stop calling when humans entered a meadow. They specifically noted that *B. canorus* seemed to be more sensitive to this disturbance than the sympatric Pacific treefrog (*Pseudacris regilla*). Karlstrom (1962) as well as several biologists presently working on this species disagree that this species' calling behavior is impacted more strongly by human disturbance than other toad species. Karlstrom (1962) did notice wariness at night and that cars moving through the area even at 0.8 km distance would cause choruses to cease calling and that "the almost continual daytime traffic in [Yosemite National Park] might help to explain the paucity of roadside populations" of *B. canorus*. It has also been suggested that the relatively frequent handling and study experienced by some populations could induce stress and immunosuppression, which may also be playing a role in declines (Green and Kagarise Sherman 2001). To our knowledge, this possibility has not been investigated.

Davidson et al. (2002) found that areas where toads had disappeared were downwind from disproportionately large areas of agricultural land (primarily the low-elevation populations on the western side of the Sierra Nevada), suggesting that wind-borne agrochemicals may be a factor in declines. However, this relationship was not statistically significant.

When this evidence is taken together, it is clear that the causes of decline for *B. canorus* are still poorly understood. It is possible that several factors act in combination, perhaps interacting with variation in life history or metapopulation dynamics. Individual populations may be susceptible to localized extirpation due to small population sizes and the species' slow maturation rate. Increasing frequency of localized extirpations could cause a breakdown of broader-scale metapopulation dynamics, leading to additional declines as recolonization ceases to counteract local population extirpations. Landscape genetic data suggest that migration rates between local populations are already low in several areas and this situation would likely be exacerbated by additional localized population declines and extirpations (Wang 2012). These inferences are still speculative, however, and further research is needed on many aspects of *B. canorus* population biology to better understand ongoing declines.

Status Determination

Declines in both distribution and abundance, coupled with a poor understanding of the factors leading to decline, are the major factors justifying a Priority 1 Species of Special Concern status.

Management Recommendations

An effective management program for this species will depend on identifying and prioritizing the factors leading to observed, ongoing declines. Until this is accomplished, protecting breeding meadows from disturbance of natural hydrologic regimes and water table dynamics and limiting human disturbance to meadows during the breeding season may be helpful in safeguarding populations. In addition, upland wintering habitats adjacent to breeding areas should also be protected from grazing and other disturbances.

Monitoring, Research, and Survey Needs

Ongoing monitoring and study of this species is required with a particular aim of identifying the major factors leading to decline. It is possible that some populations are relatively stable, and comparisons with declining sites could lead to important insights into reasons for declines and potential management solutions. Experimental work, going beyond the primarily correlational studies that have been carried out thus far, could also be helpful in identifying the most important factors. In particular, experimental studies of human disturbance, susceptibility to disease, and the potential role of reduced snowpack on hibernation and breeding biology would all be useful. Populations should also be monitored for disease outbreaks.

Further genetic work also needs to be completed to characterize genetic diversity within the species. Several studies have already been carried out, although they rely primarily on mitochondrial data alone, which is unable to distinguish true population substructure (or multiple lineages) from introgression from nearby *B. boreas* populations. Wang (2012) adds important information from the nuclear genome, but focuses on Yosemite National Park rather than the species' range as a whole. Future studies should utilize multiple unlinked nuclear markers to clarify the diversity present in the species, gene flow among meadows, and effective population sizes.

Finally, the majority of survey efforts to date have focused on populations within the boundaries of Yosemite National Park. A committed survey effort is needed to better understand the location of populations, their trends in distribution and abundance, and their disease status and level of infection (or lack of) in areas outside of the park itself.

NORTHERN RED-LEGGED FROG

Rana aurora Baird and Girard 1852

Status Summary

Rana aurora is a Priority 2 Species of Special Concern, receiving a score of 55% (61/110). Previously this species was included as a Species of Special Concern by Jennings and Hayes (1994a).

Identification

Rana aurora is a medium-sized (70–100 mm SVL) brown, red, gray, or tan frog with dorsolateral folds (Dumas 1966, Nussbaum et al. 1983). The dorsum varies from having indistinct, irregular black spots 2–3 mm in diameter with many tiny flecks to an allover network pattern of black lines (Dunlap 1955, Dumas 1966, Nussbaum et al. 1983). The dorsum can also be largely unmarked, though this is less common. A light lip line from eye to shoulder is usually present, often with a dark mask above (Nussbaum et al. 1983). Ventrally, the chest and abdomen are often marbled gray, with the groin area heavily and darkly mottled (Dunlap 1955). Red coloration on the venter and underneath the hind legs is typical but varies in intensity and extent (Dunlap 1955). Yellow coloration is common in the groin, as well as red and sometimes green coloration (Dunlap 1955). Larvae are dark brown from above, with scattered small clumps

Northern Red-Legged Frog: Risk Factors

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

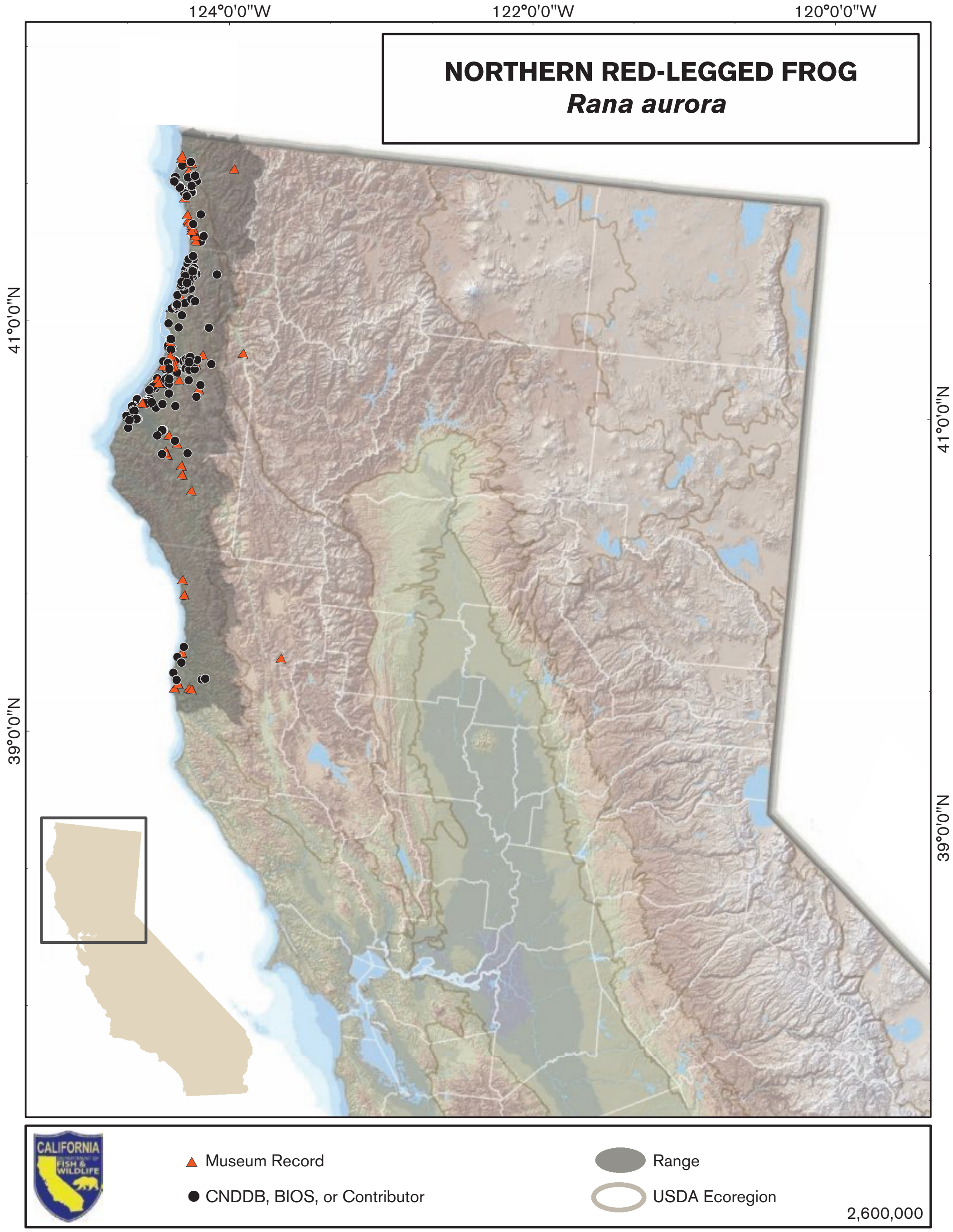


PHOTO ON PREVIOUS PAGE: Northern red-legged frog, Humboldt County, California. Courtesy of William Flaxington.

of metallic flecks and are 50–75 mm in TL (Nussbaum et al. 1983). In southern Mendocino County, this species could be confused with *R. draytonii* (Shaffer et al. 2004). *Rana draytonii* is a larger frog (up to 138 mm SVL; Stebbins 2003), and typically its dark dorsal markings have light centers (Stebbins 2003).

Taxonomic Relationships

Although initially described as a distinct species (Baird and Girard 1852), for much of the twentieth century northern red-legged frogs were regarded as the subspecies *Rana aurora aurora* (Camp 1917). Studies over the last few decades have supported the current taxonomic arrangement, with *R. aurora* and *R. draytonii* both recognized as distinct species. While they look superficially similar, these two species differ in morphology (vocal sacs, body size) and breeding behavior, and are genetically distinct (Hayes and Miyamoto 1984, Green 1985, Green 1986a, Green 1986b, Hayes and Kremples 1986, Shaffer et al. 2004). A narrow contact zone occurs between *R. aurora* and *R. draytonii* in southern Mendocino County (Shaffer et al. 2004).

Life History

Limited information is available on *Rana aurora* life history, with most studies occurring outside of its California range. Adults migrate to wetlands to breed for a few weeks between December and April when temperatures range from 4°C to 18°C (Storm and Pimentel 1954, Storm 1960, Dumas 1966, Licht 1969, Calef 1973). Males call beneath the water (Licht 1969, Brown 1975b). During one breeding season in Humboldt Bay National Wildlife Refuge, two breeding events more than a month apart produced over half of the egg masses (J. Betasso et al., unpublished data). Egg masses are 15–25 cm in diameter and contain approximately 500–600 eggs on average (Calef 1973, Licht 1974, Brown 1975b). Eggs are attached to emergent and floating vegetation, branches, or logs up to 150 cm below the water surface (Brown 1975b, Storm 1960, Calef 1973, Cary 2010). Surveys in Humboldt Bay National Wildlife Refuge found that most egg masses occurred between 30 and 60 cm elevation in the stream channel, at an average height of 37 cm above the bottom and 8 cm below the water surface (J. Betasso et al., unpublished data). In southwestern British Columbia, eggs were found at least a meter away from the pond edge or river bank (Licht 1969). Water temperatures near developing eggs in a Washington pond were 6.2°C on average (Brown 1975b), and embryos tolerate temperatures from 4°C to 21°C (Licht 1971). Dumas (1966) reared embryos at 11°C, 15°C, and 20°C, and observed the greatest embryo mortality at 20°C.

Embryonic development (from laying to hatching) takes 34–49 days (Storm 1960, Licht 1971, Brown 1975b). In Humboldt Bay Wildlife Refuge, most egg masses (103/232) persisted for 4 weeks before completely hatching out (J. Betasso et al., unpublished data). Larvae hatch at 8–12 mm long (Storer 1925, Storm 1960, Brown 1975b) and grow to up to 80 mm TL (Brown 1975b). Metamorphosis occurs after 3–7 months (Brown 1975b, Storer 1925) and metamorphs are 18–29 mm (Brown 1975b, Storm 1960, Calef 1973). Larger metamorphs are more likely to survive and to emigrate farther (Chelgren et al. 2006). In a Washington population, eggs were laid in February and March, the first larvae hatched in April, and metamorphosis was completed in late July (Brown 1975b).

Rana aurora juveniles disperse from breeding sites within days or weeks after transformation (Licht 1974, Licht 1986a). While daily movements of adults may be on average only a few meters per day, movements of several hundred meters to 4.8 kilometers have been documented over longer periods (Haggard 2000, Hayes et al. 2001, Chan-McLeod and Wheeldon 2004, Hayes et al. 2007).

Larvae are algal grazers (Dickman 1968). Metamorphs and adults are generalist predators of insects, spiders, and mollusks (Licht 1986b).

Habitat Requirements

Rana aurora occurs in mesic forests and riparian areas, which in its northern California range are primarily steep coniferous forests, coastal terraces, and floodplains (Nussbaum et al. 1983, Stebbins 2003). *Rana aurora* is relatively terrestrial for a ranid frog. Adults can occur hundreds of meters from water, and are often found in dense vegetated or downed log cover (Dunlap 1955, Dumas 1966). Adult frogs radio-tracked from March to July in Humboldt County were detected on land 90% of the time and usually within 5 m of water, though animals were found up to 80 m away from water (Haggard 2000). In habitat choice experiments, juvenile frogs spent most of their time out of the water (Pearl et al. 2004).

Both permanent and temporary breeding habitats are used, such as ponds, freshwater lagoons, lakes, and slow-moving streams (Licht 1969, Cary 2010, Sun 2012). Artificial habitats such as drainage ditches are also used (T. Fuller, J. Garwood, and M. van Hattem, pers. comm.). Coastal streams may be important dispersal corridors to inland populations. For example, *R. aurora* have been found outside of the breeding season in coastal streams in Humboldt and Del Norte Counties, and egg masses have been found in backwaters and alcoves of the Smith River where surrounding areas have been diked, drained, and converted (J. Garwood, pers. comm.). Both aquatic and terrestrial vegetation are important determinants of breeding habitat quality. In Humboldt County, egg mass presence was positively correlated with low canopy cover (ponds with less than ~40% canopy cover are more likely to have egg masses present; Cary 2010). Egg mass density was higher in smaller ponds (2000 m^2) and in ponds where the percentage of floating and emergent vegetation cover was at least ~40% (Cary 2010). Surveys in Oregon also found support for the importance of emergent vegetation, as wetlands used for breeding had 27% open water on average compared to 50% open water in unused wetlands (Pearl et al. 2005a). Occupancy models fit to 5 years of survey data in Oregon predicted that local extinction probability decreased as the percentage of trees along the shoreline increased and surface area of emergent vegetation increased (Adams et al. 2011).

Distribution (Past and Present)

Rana aurora occurs from Mendocino County, California, north along the west side of the Cascade Crest up through Vancouver Island and the adjacent mainland coast of British Columbia (Stebbins 2003). Populations also occur on Graham Island, British Columbia (Ovaska et al. 2002), and on Chichagof Island, Alaska (Hodge 2004). The elevational range extends from near sea level to 1160 m in Lane County, Oregon (Dunlap 1955), with populations in California occurring up to approximately 300 m (Jennings and Hayes 1994a). Two localities included on our map possibly extend the eastern edge and elevation range in California, and are in need of further investigation. A specimen collected by Camp in 1913 from eastern Mendocino County is in the UC Berkeley collection (MVZ 5068), photographs of which were reviewed by several experts. It is possible that the specimen is a misidentified *R. draytonii*, or it may be that *R. aurora* was historically more widespread. Despite the presence of potentially suitable habitat, contemporary CDFW biologists working in this region have not observed any *R. aurora* east of Highway 101 or in Mendocino National Forest (T. Fuller, pers. comm.). At another site, two individuals were found recently in eastern Humboldt County at around 800 m elevation (M. van Hattem, pers. comm.).

In California, surveys have found *R. aurora* to be mostly absent from the river bottom lands of the Eel, Mad, and Smith Rivers. These areas have undergone extensive habitat conversion to beef, dairy, and bulb farming, though populations may persist on inaccessible private lands (M. van Hattem, unpublished data). Surveys in Oregon's Willamette valley found *R. aurora* at 50% of sites, with highest occupancy probability observed in seasonal sites without fish (Rowe and Garcia 2013).

Trends in Abundance

Population declines have been suspected for *Rana aurora*, particularly in Oregon's Willamette Valley (e.g., Nussbaum et al. 1983; Hayes and

Jennings 1986). However, systematic surveys are lacking. Data on *R. aurora* abundance in California are limited, particularly with regard to documenting trends over time. Mean density of egg masses in breeding ponds in Humboldt County during one breeding season was 0.2/m^2, with densities up to 0.7/m^2 observed (Cary 2010). In Del Norte County, 382 egg masses were found in a 40 × 40 m area of a pond near the confluence of East Fork Mill Creek and West Branch Mill Creek (J. Garwood, unpublished data). Surveys in California have found more egg masses in areas where natural vegetation buffers the breeding habitat compared to developed areas (M. van Hattem, unpublished data).

Nature and Degree of Threat

The major threat to *Rana aurora* is development and forest conversion leading to habitat loss and degradation. Other threats include introduced predators, disease, and climate change, though more data are needed on each of these stressors.

Due to issues such as low capture rates, it is unclear whether *R. aurora* abundance varies consistently with stand age in harvested forests (reviewed in Pearl 2005). For example, terrestrial (Welsh et al. 2007) and aquatic (Ashton et al. 2006) amphibian surveys in northwestern California forests have documented only a handful of *R. aurora*. In Washington, breeding sites with high primary forest cover within 2 km had higher egg mass counts, as did breeding sites greater than 0.25 km away from roads (Holcomb 2012). On Vancouver Island, radio-tracked frogs tended to move toward old-growth stands and away from clear-cuts <12 years old, suggesting that recolonization of impacted sites may require several years (Chan-McLeod 2003). In an Oregon study, the highest capture rates of *R. aurora* were in mature, mixed large sawtimber forest (Martin and McComb 2003).

Agricultural and residential development has likely contributed to habitat loss and degradation for *R. aurora*, and is projected to continue to increase in the future. For example, much of the Smith River coastal plain in Del Norte County has been converted to lily bulb production (J. Garwood, pers. comm.). In addition to habitat loss, such agricultural conversion can further degrade habitat through use of chemicals such as pesticides, herbicides, and fungicides. Similarly, the emerging issue of largely unregulated marijuana cultivation can degrade watersheds through grading and roadbuilding (which both destroy habitat and create runoff into aquatic habitats), application of pesticides and herbicides, and through dewatering of springs, streams, and wetlands used for irrigation (e.g., Thompson et al. 2014). Residential and commercial development is likely to increase in northern California, potentially leading to losses of breeding habitat or loss of access to remaining habitat. For example, the Humboldt County General Plan is currently being updated, with some proposals considering a doubling or tripling of rural development. However, *R. aurora* does use artificial habitat for breeding, and amount of urban cover was not a strong predictor of frog occurrence in surveys in Oregon (Rowe and Garcia 2013), suggesting some tolerance for certain kinds of habitat modification.

Introduced predatory fish and bullfrogs are widespread throughout *R. aurora* habitat in California, including sites near the coast (T. Fuller, J. Garwood, and M. van Hattem, pers. comm.). Negative impacts have been documented in mesocosm experiments, but field observations have yielded both negative and neutral effects of fish and bullfrogs on *R. aurora* distribution and abundance. Field-enclosure experiments in Oregon have shown reduced survivorship, shifts in microhabitat use, slower development, and smaller size at metamorphosis of *R. aurora* in the presence of fish and bullfrogs (Kiesecker and Blaustein 1998). Surveys in Oregon and Washington have found evidence for negative associations between *R. aurora* presence or abundance and the presence of nonnative fish but weak or no evidence for an effect of bullfrogs (Adams 1999, Pearl et al. 2005a, Rowe and Garcia 2013). Other studies in Oregon and Washington have not detected any effects of fish or bullfrogs on *R. aurora* presence (Richter and Azous 1995, Adams et al. 1998, Adams et al. 2011). Little data are available from California. Freshwater Lagoon and Big Lagoon in Humboldt County both have a long history of fish stocking, and surveys of suitable habitat during the 2010 and 2011 breeding seasons never found more than 1 egg mass in either lagoon (M. van Hattem, unpublished data). While introduced fish and bullfrogs can prey upon *R. aurora*, the population-level impacts of such predation are unknown. Gut content analysis of 5075 bullfrogs collected over 5 years on Vancouver Island found *R. aurora* in only 0.2% of stomachs (Jancowski and Orchard 2013.

Expected climate changes within the California range of *R. aurora* over the next 100 years include increased temperatures, sea-level rise, changes in hydrology, changes in fire regime, and vegetation shifts (reviewed in PRBO 2011). The frequency of extremely hot days is projected to increase, with roughly nine additional days over 32.2°C (Bell et al. 2004), though the effects of increased temperature are difficult to predict. A mesocosm experiment on larval *R. aurora* found that the combined effects of warming and drying can offset each other: warmer conditions result in more algal resources, allowing larvae to develop faster and escape costs of drying (O'Regan et al. 2014). Sea-level rises as high as 72 cm above 1990 levels are predicted under some models for California (reviewed in PRBO 2011), which may cause saltwater intrusion into estuarine habitat used for breeding. Upwelling is expected to intensify, which may increase fog development and contribute to cooler, moister conditions (Snyder et al. 2003, Lebassi et al. 2009), possibly facilitating terrestrial habitat use by this species along the coast. Potential changes in precipitation are less clear, some models predict either modest increases or decreases in rainfall, while others predict sharp reductions of up to 28%. (reviewed in PRBO 2011). Reductions in water availability due to reduced snowpack and possibly reduced precipitation will affect the timing and magnitude of stream flows, which may negatively affect habitat (Snyder et al. 2004, Stewart et al. 2005, Cayan et al. 2008b). 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 *R. aurora* responds to wildfire is unknown. 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). Loss of moist forest habitat would likely be detrimental to *R. aurora*; however, most of the predicted vegetation changes occur farther inland from its range.

Disease has been repeatedly implicated in amphibian declines, but to date there is little evidence that disease has played a major role in determining *R. aurora* abundance. While *Bd* has been documented from a high proportion of sites examined in Humboldt County (11/13; Nieto 2004, Sun 2012), the prevalence of infected individuals is relatively low (15%; Nieto 2004, Sun 2012). Water mold infection of egg masses has been observed in the field (Cary 2010, M. van Hattem, unpublished data) but population consequences of infection are unknown. Terrestrial versus aquatic life stages may respond differently to fungal infection. Juvenile metamorphs infected with *Saproglenia* in the lab did not have significantly higher mortality than uninfected individuals (Romansic et al. 2007), while two weeks of exposure was lethal to *R. aurora* larvae (Romansic et al. 2009a).

Status Determination

Rana aurora has a small range in California in a region that is undergoing continuing development, agricultural use, and timber harvest, making it a Priority 2 Species of Special Concern.

Management Recommendations

Management of *Rana aurora* should focus on addressing habitat degradation and loss due to development, timber harvest, and agriculture (including marijuana cultivation), introduction and spread of nonnative predatory fish and bullfrogs, and on minimizing unintended negative impacts due to salmonid restoration. Observations of higher abundance in breeding habitat with intact terrestrial vegetation nearby (though not excessively shading ponds; Cary 2010, Adams et al. 2011, Holcomb 2012) coupled with the terrestrial habitat use and long distances traveled by adults (Hayes et al. 2007) support the idea of maintaining vegetation buffers around breeding habitat in forested areas and setbacks between wetlands and development. Current regulations for development setbacks under the California Coastal Act of 1976 give distances from breeding wetlands of up to 30 m depending on land use. However, these setbacks are reducible upon request and we recommend that consistent, biologically based setbacks be developed. *Rana aurora* may experience less impact from timber harvesting methods that leave residual tree patches, particularly if multiple trees are included in patches between 0.8 and 1.5 ha in size and are near streams (Chan-McLeod and Moy 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. Populations of introduced fish and bullfrogs should be prevented from invading *R. aurora* breeding habitat. While bullfrogs may already be widespread, intentional fish stocking should be restricted to avoid *R. aurora* habitat. Restoration projects for native salmonids should also take into consideration potential impacts to *R. aurora* that may be caused by converting freshwater wetlands to estuarine habitats and salt marshes.

Monitoring, Research, and Survey Needs

Monitoring of *Rana aurora* egg mass counts should continue in order to provide baseline data on distribution and abundance and to detect declines. *Rana aurora* management would benefit from additional study of movement and habitat use, life history, effects of marijuana cultivation, and impacts of introduced species in the field. Particularly as habitat becomes increasingly fragmented, data on connectivity among habitat patches, effects of road density, and use of terrestrial habitat away from breeding ponds can help inform appropriate setback distances and buffer configurations. Genetic studies may also be helpful for understanding patterns of frog movement across the landscape. Basic life history information overall and from the California range in particular is also lacking. Understanding saltwater tolerance of different life stages would be useful for predicting the extent of sea-level-rise effects on coastal populations. Field research on impacts of marijuana cultivation on amphibian populations would contribute to developing environmental regulations for this growing industry. Much of the concern for bullfrog impacts on *R. aurora* is from experimental mesocosm studies. Additional research that addresses the effects of bullfrogs and fish on *R. aurora* in the field is necessary to understand the community context of impacts, as the effects of bullfrogs in combination with fish may be greater than either singly (Kiesecker and Blaustein 1998), and fish may be facilitating bullfrog survival (Adams et al. 2003). Under the assumption that eradication of well-established introduced species is unlikely to be feasible at a large scale, a main goal of this work should be identifying factors that can potentially be manipulated to promote coexistence between *R. aurora* and nonnative predators, such as managing terrestrial and aquatic vegetation cover and hydroperiod.

FOOTHILL YELLOW-LEGGED FROG

Rana boylii Baird 1854

Status Summary

Rana boylii is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 83% (91/110). During the previous evaluation, it was also considered a Species of Special Concern, with varying levels of threat in different parts of the range (Jennings and Hayes 1994a).

Identification

Rana boylii is a small to medium-sized frog (up to 81 mm SVL) (Stebbins 2003). The skin usually appears rough and granular, with many tiny tubercles on the surface, including on the tympanum (Nussbaum et al. 1983). The dorsal coloration is variable and can be gray, brown, reddish, or olive, sometimes with extensive brick-red coloration around the weak dorsolateral folds (Nussbaum et al. 1983). Individuals can also change their overall coloration from relatively light to dark (Wheeler et al. 2005). An inverted triangle-shaped patch of buff coloration is usually present on the snout, but its distinctiveness varies (Stebbins 2003). The ventral coloration is typically yellow on the hind legs and posterior abdomen, with the rest of the venter mostly white with dark mottling on the throat and chest (Nussbaum et al. 1983).

Foothill Yellow-Legged Frog: 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)	7
vi. Population trend (25)	25
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	7
Total Score	91
Total Possible	110
Total Score/Total Possible	0.83

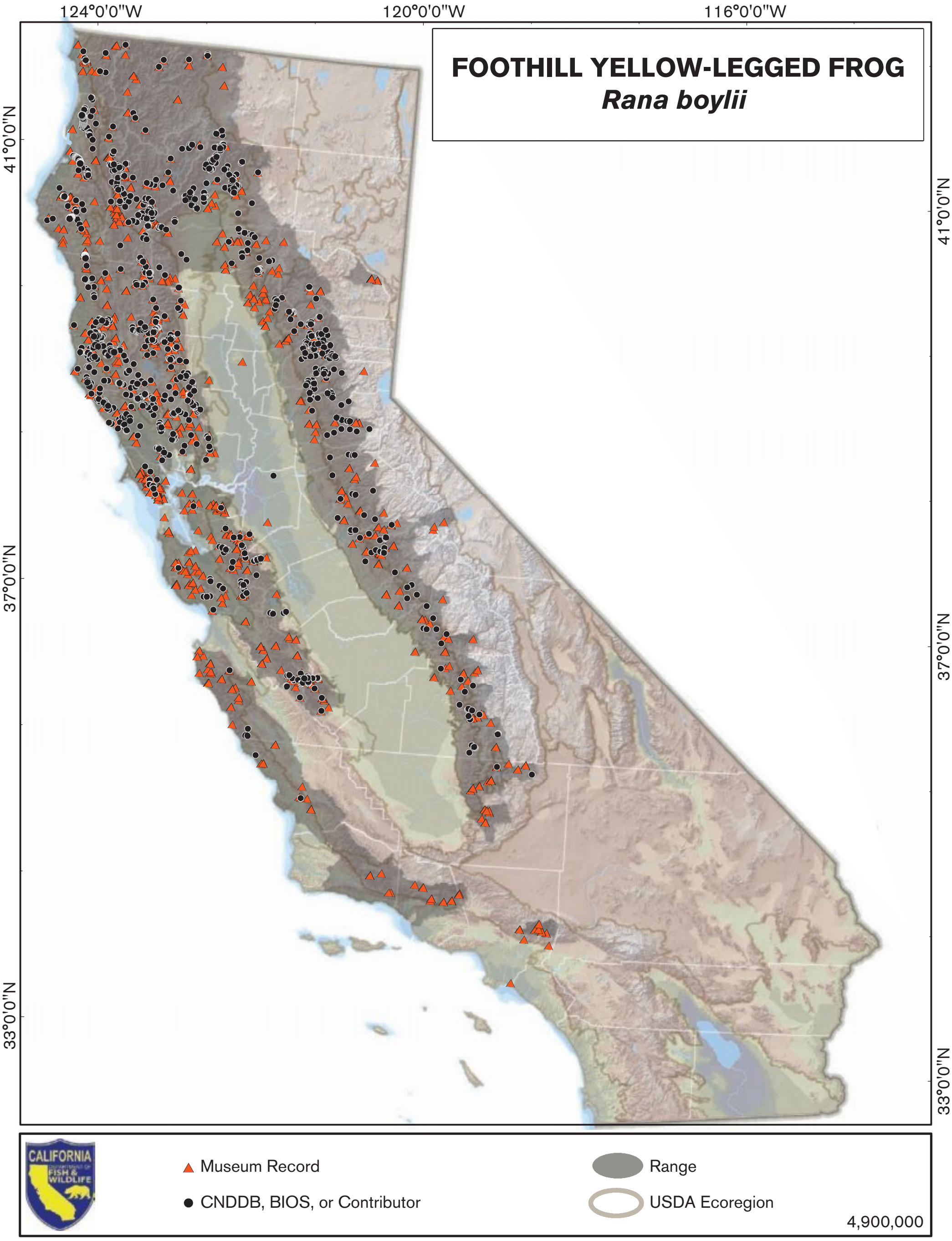


PHOTO ON PREVIOUS PAGE: Foothill yellow-legged frog, Del Norte County, California. Courtesy of Rob Schell Photography.

Jennings and Hayes (2005) documented orange or red coloration on the ventral surfaces of the hind limbs in post-metamorphic animals from Glenn, Tehama, and Stanislaus Counties. Tadpoles reach a maximum size of 55 mm and are usually olive dorsally with dark spots or mottling that matches the stream substrate and a silvery venter (Nussbaum et al. 1983). Males call primarily underwater but will also call above (MacTague and Northen 1993).

Other species that *R. boylii* could potentially be confused with in California include the California and northern red-legged frogs (*R. draytonii* and *R. aurora*), the mountain and Sierra Nevada yellow-legged frogs (*R. muscosa* and *R. sierrae*), and juvenile bullfrogs. *Rana draytonii and R. aurora* have smooth skin, a prominent jaw stripe, distinct dorsolateral folds, and usually have red coloration under the hind limbs (although *R. boylii* can also have red ventral coloration, and young *R. draytonii and R. aurora* often have yellowish thighs) (Stebbins 2003, Jennings and Hayes 2005). *Rana muscosa and R. sierrae* have smoother skin, smooth tympana, and tend to lack the light patch on the snout (Stebbins 2003). Bullfrogs occasionally co-occur with *R. boylii* but tend to be greenish in color, with smoother skin, and large, smooth tympana (Stebbins 2003).

Taxonomic Relationships

Rana boylii has been recognized as a distinct species for a long period of time, although its phylogenetic placement among other North American ranids has been revised repeatedly (Baird 1854, Macey et al. 2001, Hillis and Wilcox 2005). Zweifel (1955) documented variation in color and morphology among California *R. boylii* populations. Recent phylogeographic studies have found that genetic variation among *R. boylii* populations is structured along hydrologic boundaries (Dever 2007, Peek 2010, Lind et al. 2011). In a range-wide phylogeographic study, Lind et al. (2011) identified some peripheral populations that are deeply divergent from populations within the core of the range. In California, populations in southernmost Monterey County west and south of the Salinas River Valley and populations from the southern Sierra Nevada were found to be phylogenetically distinct from the rest of *R. boylii*, suggesting a long history of isolation. While extreme southern populations from Los Angeles County are now extirpated, Lind et al. (2011) hypothesized that animals from those localities may also have been genetically distinct.

Life History

As a stream-dwelling frog, the life history of *Rana boylii* coincides with seasonal patterns in river flows associated with California's Mediterranean climate. The most sensitive life stages (eggs and larvae) develop during relatively stable conditions when streams are at their lower stages (Kupferberg et al. 2009b). Breeding and oviposition occur in spring after flood waters recede, and tadpoles metamorphose in late summer through early autumn before winter rains (reviewed in Lind 2005, Haggarty 2006, Wheeler and Welsh 2008). Southern populations breed earlier than northern populations (Zweifel 1955), and the onset and duration of breeding can be influenced by water temperature, cessation of rainfall, water velocity and depth, and day length (Zweifel 1955, Kupferberg 1996a, Lind et al. 1996). Between 2002 and 2007 at a site in Del Norte County, breeding activity was initiated in early April and lasted for 19–52 days, with earlier breeding occurring in low-flow years (Wheeler and Welsh 2008). Breeding activity ceased briefly during rain events that increased flows (Wheeler and Welsh 2008).

Females lay a single cluster of up to 2000 eggs (Zweifel 1955) attached to pebble or cobble substrates (Fuller and Lind 1992) or to bedrock (M. van Hattem, pers. comm.). Eggs take 2–3 weeks to hatch, depending primarily on water temperature (Kupferberg 1996a). Major sources of natural egg mortality are desiccation through stranding in dry years, and scour from floods in wet years (Kupferberg et al. 2009b). Adults breed at 2 or 3 years of age depending on the geographic location, and this translates into fluctuations in adult populations being determined by environmental conditions during recruitment 2–3 years prior (Kupferberg et al. 2009b). Metamorphosed animals captured in Tehama County were 1.2–7.2 years old based on skeletochronology (Bourque 2008), suggesting that they can be relatively long-lived.

Radiotelemetry studies are beginning to offer more insight into terrestrial movements. Adults aggregate at pools in the spring but become more difficult to find in the summer (Van Wagner 1996, Haggarty 2006, Wheeler and Welsh 2008). In one study in Tehama County, frogs used watercourses for movement and were rarely more than 12 m from the stream channel (Bourque 2008). Females tended to move upstream during spring and downstream during the fall and winter. Travel rates in this population were up to 1386 m/day, faster than previously thought. In other studies, the longest distances traveled have been closer to 500 m at rates of tens to a few hundred meters per day (Van Wagner 1996, Drennan et al. 2006, Wheeler et al. 2006). Females tend to move farther distances than males, with female movements up to 7 km documented in one study (Bourque 2008, Gonsolin 2010). At one locality in Del Norte County, 68% of males remained in one breeding site during the reproductive season, with average home range sizes of 0.58 m^2 (Wheeler and Welsh 2008). At a site where the availability of permanent water is a limiting factor in Santa Clara County, resident tributary frogs moved to the main stem to breed and moved greater distances than resident main stem frogs (Gonsolin 2010). Greater than 90% of movements were associated with movements to or from breeding sites, and all movements outside of the breeding season were made in response to the channel drying back or to rainfall (Gonsolin 2010).

Larvae appear to be herbivorous, while metamorphs and adults consume terrestrial and aquatic insects. Algae with epiphytic diatoms are a preferred food for larvae, and the abundance of floating algae indicates the quality of larval food resources (Kupferberg 1996b, Kupferberg 1997). Metamorphosed animals primarily forage terrestrially (Zeiner et al. 1988, Van Wagner 1996, Haggarty 2006, Hothem et al. 2009). Spiders, beetles, and flies are common prey items (Haggarty 2006, Wiseman and Bettaso 2007, Hothem et al. 2009). Gut content analyses of adults collected from 22 sites in the Cache Creek watershed found that 98% of individuals contained terrestrial prey, 28% contained aquatic prey, and one animal contained mammal hair and bone fragments (Hothem et al. 2009). Two occurrences of adults cannibalizing juvenile conspecifics have been documented (Wiseman and Bettaso 2007).

Habitat Requirements

Rana boylii is primarily stream dwelling and requires shallow, flowing water in streams and rivers with at least some cobble-sized substrate (Hayes and Jennings 1988). Different life stages use different habitat types for development, foraging, and overwintering.

Breeding and oviposition occur at the margins of relatively wide and shallow channel sections, habitats that experience reduced flow variation (Storer 1925, Fitch 1936, Kupferberg 1996b, Lind et al. 1996). Breeding sites are often located near tributary confluences (Kupferberg 1996a, Bourque 2008). Egg masses are attached in low-flow locations behind and sometimes under rocks. The most commonly used substrates for breeding sites are cobble, boulders, and gravel (Fuller and Lind 1992, Kupferberg 1996a). Eggs have been found at water depths up to 87 cm (C. Bondi, S. Yarnell, and A. Lind, pers. comm.), in water velocities of 0–0.21 m/s, and up to 12.5 m from shore (Kupferberg 1996a, reviewed in Lind 2005). The critical thermal maximum for embryos is 26°C, and eggs have been found in water ranging from 9°C to 21.5°C (Zweifel 1955). Density of egg masses was highest in Eel River reaches when July mean temperatures were between 17.5°C and 19°C (Catenazzi and Kupferberg 2013). Egg mass surveys from 1991 to 2002 across 11 small and large streams in the Northern Coast Ranges and the Sierra Nevada found that oviposition sites occurred in a very narrow range of microhabitat conditions that were different from randomly selected habitats, strongly suggesting active habitat selection by frogs (Lind 2005). High-quality breeding areas are often used over multiple years (Lind 2005). Larvae tend to stay in natal habitats until they metamorphose (Van Wagner 1996). Surveys in the Mattole Watershed in northern coastal California across different channel types found that tadpole presence was best predicted by relatively warmer water temperatures (Welsh and Hodgson 2011). Tadpoles were never found in water colder than 13°C, and tadpole abundance increased with water temperature (Welsh and Hodgson 2011). In choice experiments, tadpoles selected temperatures between 16.5°C and 22.2°C (Catenazzi and Kupferberg 2013).

Metamorphosed animals use a variety of aquatic habitats, including riffles, pools, and glides (reaches intermediate between riffles and pools) depending on the life stage and season (Van Wagner 1996, Yarnell 2000, Lind 2005, Yarnell 2005, Haggarty 2006). At Red Creek in Tehama County, post-breeding season adults and subadults preferred pool and riffle habitats, while young of the year metamorphs selected slower-moving glides and runs (Haggarty 2006). In Nevada County, all age classes used riffles after the breeding season (Van Wagner 1996). In the Sierra Nevada foothills, subadults chose fast-flowing sections of stream, while adults used slower-moving pool habitats (Yarnell 2000, Yarnell 2005). In the Mattole Watershed, the best predictor of adult presence in streams was canopy openness (Welsh and Hodgson 2011). Abundance of adults and larvae was positively associated with larger basin areas and finer substrates, conditions more typical of alluvial channels than other channel types (Welsh and Hodgson 2011).

Less is known about terrestrial habitat use. Adults typically occur along waterways with some degree of shading (Fitch 1938, Zweifel 1955, Moyle 1973, Hayes and Jennings 1988, Van Wagner 1996), although they also occur in open habitats (Welsh et al. 2005, Haggarty 2006, Welsh and Hodgson 2011). During the spring, radio-tracked males and females in Tehama County were often found on land near water (38% and 66% of the time, respectively; Bourque 2008). The average distance from water was less than 3 m in all seasons, although adults occasionally used habitat up to 40 m distant from streams (Bourque 2008). Adults move to tributaries or upland habitats to avoid floods following large rain events (Kupferberg 1996b, Van Wagner 1996, Yarnell 2000, Bourque 2008). Tributaries are also used for overwintering in early spring before adults are abundant on the principal channels (Kupferberg 1996b, Yarnell 2000). Juveniles will also move into tributaries, with maximum movements of 860 m from hatching site to upstream tributaries observed in Santa Clara County (Gonsolin 2010). Adults may aggregate above ground in terrestrial microhabitats on tributaries post-breeding (Leidy et al. 2009).

Distribution (Past and Present)

Historically, *Rana boylii* occurred in foothill and mountain streams from the San Gabriel River in Los Angeles County to southern Oregon west of the Sierra-Cascade crest (Nussbaum et al. 1983, Stebbins 2003), from sea level to 1940 m (Hemphill 1952). There is an isolated, unverified record from northern Baja California, Mexico, at ~2000 m (Loomis 1965).

Jennings and Hayes (1994a) considered *R. boylii* endangered in central and southern California south of the Salinas River, threatened in the west slope drainages of the Sierra Nevada and Cascades, and of special concern in the Coast Ranges north of the Salinas River. They estimated that *R. boylii* were extirpated from 45% of their historical localities in California, and 66% of historical localities from the Sierra Nevada. Building on that mapping effort, Lind (2005) looked at 394 historic localities in California and Oregon, and found that 201 localities (51%) were no longer occupied, with extirpations largely in southern California and northern Oregon. Kupferberg et al. (2012)

determined current occupancy of 310 randomly selected sites that were occupied prior to 1975. They found that half of the sites still had *R. boylii* populations, with frogs more likely to be present in sites without large dams.

Extirpations likely began in the second half of the twentieth century. Grinnell and Storer (1924) noted several sites in the Sierra Nevada foothills around Yosemite where *R. boylii* were common. In resurveys of those sites and surveys of additional sites in the early 1990s, Drost and Fellers (1996) did not find any *R. boylii*. Surveys by Moyle (1973) in the 1970s found *R. boylii* at only 30/95 sites in the southern and central Sierra Nevada foothills. Field surveys since 1993 have found at least one frog at only 213/804 sites in 28/40 California counties (Fellers 2005a). Fellers (2005a) estimated that extant populations occur in 40% of streams in the Pacific Northwest, 30% of streams in the Cascade Mountains, 30% of streams in the south Coast Range (south of San Francisco), and 12% of streams in the Sierra Nevada.

Trends in Abundance

Kupferberg et al. (2012) compiled egg mass density data from multiple sources on 27 Sierran and coastal populations in northern California between 1991 and 2010. The range of densities reported was between 1.9 and 105.7 clutches/km of reach sampled. Average density was higher in free-flowing rivers (31.1 clutches/km) than in rivers with dams (5.5 clutches/km), but no differences were detected between abundances in coastal versus montane watersheds (Kupferberg et al. 2012). Fellers (2005a) reported that only 30 of 213 occupied California sites had population sizes greater than 20 adults. In the Coast Ranges, population sizes of greater than 100 adult frogs occurred at six sites, and populations greater than 50 adult frogs occurred at nine sites (Fellers 2005a). Small population sizes are presumably due to population declines, leading to predictions that populations in the southern Sierra Nevada will not be viable for more than another decade (Fellers 2005a). Minimum viable population sizes are unknown, however, and may vary across the range.

Nature and Degree of Threat

The main threats to and likely causes of *Rana boylii* decline are human activities that alter natural hydrologic regimes of streams and rivers, such as dams for hydroelectric power generation, water storage, and water delivery. Other potential stressors include land use changes that degrade or destroy riparian habitat (particularly urban and agricultural development), pesticides, disease, and invasive species.

Alterations to the natural flow regime, for example, through dam releases, can have direct mortality effects and indirect negative effects on *R. boylii* by altering habitat availability and quality. Kupferberg et al. (2009b) reviewed published literature and Federal Energy Regulatory Commission hydroelectric dam relicensing reports to assess the effects of pulsed flow releases on *R. boylii*. The data spanned 1997–2007 and included seven major river basins in California. Pulsed flows from dam releases after oviposition resulted in scouring of egg masses, while flow changes during oviposition led to stranding when water levels subsequently dropped and exposed egg masses. Similarly, tadpoles can be scoured and stranded due to pulsed-flow releases. The effect of releases on post-metamorphic animals is less clear, and the impact of flow changes on habitat availability is highly site specific. Reservoirs and dams may also disrupt patterns of connectivity among *R. boylii* populations. Comparisons of genetic structure within and among *R. boylii* populations in three pairs of regulated versus unregulated Sierran rivers found that regulated rivers exhibited lower genetic diversity and greater genetic drift compared to unregulated river populations (Peek 2010).

Kupferberg et al. (2009c) modeled *R. boylii* population growth under different flow scenarios. A major result was that populations in regulated rivers had 4–13-fold greater extinction risk than populations in unregulated rivers due to smaller population sizes. Kupferberg et al. (2009c) simulated how an unregulated population would be affected by flows more typical of regulated rivers. When subjected to aseasonal flow conditions, modeled populations showed a doubling of extinction risk. Many different kinds of hydrologic changes can contribute to these negative effects, and when different stressors are combined, the impact on frog populations is greater than expected from simply adding up the effects of individual stressors.

Field and laboratory experiments conducted by Kupferberg et al. (2011) showed that tadpoles suffered negative effects including death at or below water velocities experienced during aseasonal pulsed flows. For example, most tadpoles could no longer swim or seek refuge at velocities of ~20 cm/s or greater, and in the absence of refugia tadpoles reached exhaustion in ~7 min in a 5 cm/s current. Rates of flow in regulated reaches can be much higher than this. For example, in the North Fork Feather River, surface velocity measured in larval rearing habitat near channel edges can reach over 30 cm/s after releases for recreational purposes (Garcia and Associates 2005).

Smaller-scale hydrologic modification and loss or degradation of riparian habitat due to urban and agricultural use is also a threat to *R. boylii*. Analyses correlating *R. boylii* distribution with landscape characteristics demonstrated negative effects of urban and agricultural land use change and pesticides on *R. boylii* presence (Davidson et al. 2002, Davidson 2004, Lind 2005). Vineyard conversion can have impacts on small creeks, and the establishment of permanent ponds used for irrigation and frost protection can create habitat for bullfrogs (S. Kupferberg, pers. comm.). Marijuana cultivation practices that divert water from small creeks can lead to premature drying. Growers have been observed to construct plastic-lined impoundments in creeks and add fertilizers directly to creek water, as well as use pesticides and herbicides in and around frog habitat (Gonsolin 2010). These practices are suspected to have contributed to declines in some populations near Gilroy (Gonsolin 2010). Similar impacts are likely in Humboldt, Mendocino, and Trinity Counties (CDFG 2013). The large-scale effects of such illegal operations are unknown, and potentially dangerous to study. While in-stream gravel and suction dredge gold mining may have been more of a concern in the past, current regulations protecting salmonids have likely largely reduced the direct impact of such activities on *R. boylii*. For example, in Humboldt County in-stream gravel mining occurs above (in elevation) and outside the wetted channel, and relatively high egg mass density has been documented in reaches where gravel mining occurs in the Mad River (M. van Hattem, pers. comm.).

The current distribution of *R. boylii* is strongly correlated with climate variables, which suggests that this species may be sensitive to future climate changes, particularly those that affect stream hydrology (reviewed in PRBO 2011). Comparisons of occupied and extirpated historic localities found that sites where *R. boylii* persists have higher mean annual precipitation, less variability in precipitation, and fewer dry years than extirpated sites (Davidson et al. 2002, Lind 2005). Within the range of *R. boylii*, warming temperatures are predicted to result in more precipitation falling as rain instead of snow, and consequently less storage of water as snowpack. Reductions of 30–80% in snowpack accumulation are predicted within the northwestern range of *R. boylii*, and up to 90% reduction in snowpack is predicted for the south coast hydrologic region (Snyder et al. 2004, Cayan et al. 2008b). In the Sierra Nevada, snowpack losses of 50–90% are predicted by the end of the twenty-first century, with greatest losses at low to mid-elevations (Knowles and Cayan 2002, Hayhoe et al. 2004, Knowles and Cayan 2004, Maurer 2007, Cayan et al. 2008b). Loss of snowpack is likely to result in earlier runoff and reduced spring and summer streamflows. Timing of spring snowmelt is predicted to shift earlier in the spring in the Sierra Nevada (Snyder and Sloan 2005), while in northwestern California the opposite has occurred over the last 50 years (Stewart et al. 2005). How frogs will respond to these changes in hydrology is unknown, but negative effects due to anthropogenic changes in hydrology are well documented. Reduction in water availability may also lead to more conflict with human use of water and affect how regulated reaches are managed (reviewed in Franco et al. 2011). It is important to note, however, that predictions of changes in precipitation are much less certain than predictions for temperature (Franco et al. 2011, PRBO 2011). In addition, climate change may also affect disease dynamics. Outbreaks of nonnative parasitic copepods occurred during two recent warm years at a long-term study site, resulting in morphological abnormalities and smaller sizes at metamorphosis (Kupferberg et al. 2009a). The outbreak was likely caused by increased summer water temperature, decreased daily discharge, or a combination of these factors. These conditions may increase under a changing climate, but could also occur as a result of marijuana cultivation.

No declines to date have been associated with *Bd*, but the disease does infect *R. boylii* in the field. Padgett-Flohr and Hopkins (2009) examined museum specimens from 1890 to 2000, and found that *Bd* first appeared in *R. boylii* samples from the 1960s, with 10% of specimens infected. In all, 0–40% of specimens were infected with *Bd* in the following decades. In laboratory trials, *R. boylii* appeared to be protected by skin peptides against *Bd* and therefore may not be very susceptible to chytridiomycosis (Davidson et al. 2007). Chytrid infection did not affect survival, even in the presence of a co-applied pesticide, but did suppress growth of recently metamorphosed individuals by approximately 40% (Davidson et al. 2007).

Observational data and surveys have found that *R. boylii* is rare or absent in habitats with introduced fishes and bullfrogs (Hayes and Jennings 1986, Hayes and Jennings 1988, Kupferberg 1997, Lind et al. 2003, Fuller 2008). Breeding populations of *R. boylii* can be an order of magnitude smaller when bullfrogs are present compared to uninvaded reaches (Kupferberg 1997). In field experiments in outdoor enclosures, bullfrog tadpoles caused a 48% reduction in survivorship of *R. boylii* tadpoles, and a 24% decline in mass at metamorphosis. The mechanism behind the negative impacts of bullfrogs was competition for food (Kupferberg 1997). Metamorphosed bullfrogs prey on *R. boylii*, including post-metamorphic individuals (Crayon 1988, Hothem et al. 2009), but the population-level consequences of this predation are unclear. Another nonnative predator, the signal crayfish (*Pacifastacus leniusculus*), has been introduced into several Sierra Nevada drainages from farther north where the two species co-occur (Wiseman et al. 2005). Signal crayfish have been observed eating and dislodging egg masses and attacking larvae (Wiseman et al. 2005). Within *R. boylii*'s range, signal crayfish have been documented at 30–40 sites, with ~25 invaded sites occurring in the Sierran foothills (G. Fellers, pers. comm.).

Status Determination

Documented declines and extirpations of *Rana boylii* populations combined with continuing threats to remaining populations result in a Priority 1 designation for this species.

Management Recommendations

Several aspects of the biology of *Rana boylii* can help inform management efforts. *Rana boylii* use a variety of stream and streamside habitats during different life stages; therefore, protected habitat needs to provide adequate habitat diversity. The timing and pattern of releases of water from dams during April through June should be managed to minimize egg scouring and stranding. For example, dam releases can be staggered to better mimic the natural spring recession in snowmelt-fed streams. Further recommendations for hydrologic management can be found in Kupferberg et al. (2009b, 2009c). Dam removal should be explored where appropriate and is likely to benefit *R. boylii* and other native taxa. River management for other taxa needs to take *R. boylii* into account. For example, in-stream structures to improve habitat for fish such as steelhead can negatively impact *R. boylii* (Fuller and Lind 1992). Habitat restoration and possibly repatriation of southern Sierra Nevada populations should be considered. Southern populations in general should be priorities for conservation because of the degree of losses and distinctive genetic diversity represented in this part of the range (Lind et al. 2011). Removal or management of nonnative predators such as fish and bullfrogs may help restore *R. boylii* habitat. For example, projects that remove artificial pools (e.g., relict mine tailing ponds) by restoring linkages to main river channels would result in more natural hydrologic conditions and reduce breeding habitat for bullfrogs (Fuller et al. 2010). Finally, Lind et al. (2011) suggested that an approach using genetic analyses of *R. boylii* and co-distributed riverine taxa would help in prioritizing drainages for protection based on levels of diversity.

Monitoring, Research, and Survey Needs

Modeling of population dynamics and hydrology are highly site specific and limited by available demographic data, and acquiring those additional data should be a high priority for *Rana boylii*. More research is needed on survivorship of tadpoles and juveniles, especially during overwintering. The mechanisms underlying hydrological effects are currently best understood for egg masses, and we need to develop a better functional understanding of how hydrology affects different life stages (Kupferberg et al. 2009b). More research is also needed on post-metamorphic stages. Post-metamorphic stages may be less at risk from aseasonal pulses in river flow because they are more mobile, but in regulated rivers the timing of pulsed flow events can be decoupled from climatic cues (such as the first appreciable fall rains) that would normally trigger movement to safer refuges (Kupferberg et al. 2009b). Caution should be taken in using radio telemetry to study post-metamorphic animals, as 62% of frogs in one study suffered skin injuries from transmitters (Bourque 2008). Modeling efforts would also be improved by monitoring a Sierra Nevada population in an unregulated reach for comparison with more regulated sites (Kupferberg et al. 2009b). Egg mass counts are commonly used to monitor *R. boylii* populations. Females only lay one mass/year, so egg mass counts accurately reflect the number of reproductive females. However, operational sex ratios are female biased; therefore, accurate population size estimates cannot be made based on egg counts alone (Wheeler and Welsh 2008). Lind et al. (2011) provided important rangewide phylogeographic data, but their study was limited by very low nuclear genetic diversity and relied primarily on mitochondrial data. Additional work could provide valuable additional data on levels of variation and genetic isolation among local hydrologic basins, as might be predicted for this stream-restricted anuran. Finally, efforts to find remnant *R. boylii* populations in the San Gabriel Mountains and upper Piru Creek in southern California should continue.

CASCADES FROG

Rana cascadae Slater 1939

Status Summary

Rana cascadae is a Priority 2 Species of Special Concern, receiving a Total Score/Total Possible of 65% (72/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Rana cascadae is a medium-sized (2.0–8.0 cm SVL) frog with drab-green, tan, or brown dorsal coloration and well-defined black blotches scattered across the back (Slater 1939, Stebbins 2003). The number of blotches varies from very few to about 50 (Slater 1939), and unmarked individuals occur rarely (Jennings and Hayes 1994a). Blotches appear to be on the surface of the frog's skin and are reminiscent of spattered ink (Stebbins 2003). The species has a prominent light stripe above the jaw and strong dorsolateral folds. The venter is cream or buff, usually with yellowish (sometimes reddish) areas posteriorly and on the undersides of the legs. Laterally, the sides are mottled and fade into the ventral coloration (Slater 1939). The male advertisement call is a series of low chucks given in rapid succession, usually ending with one slightly drawn out chuck (Elliott et al. 2009).

Cascades 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)	15
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	72
Total Possible	110
Total Score/Total Possible	0.65

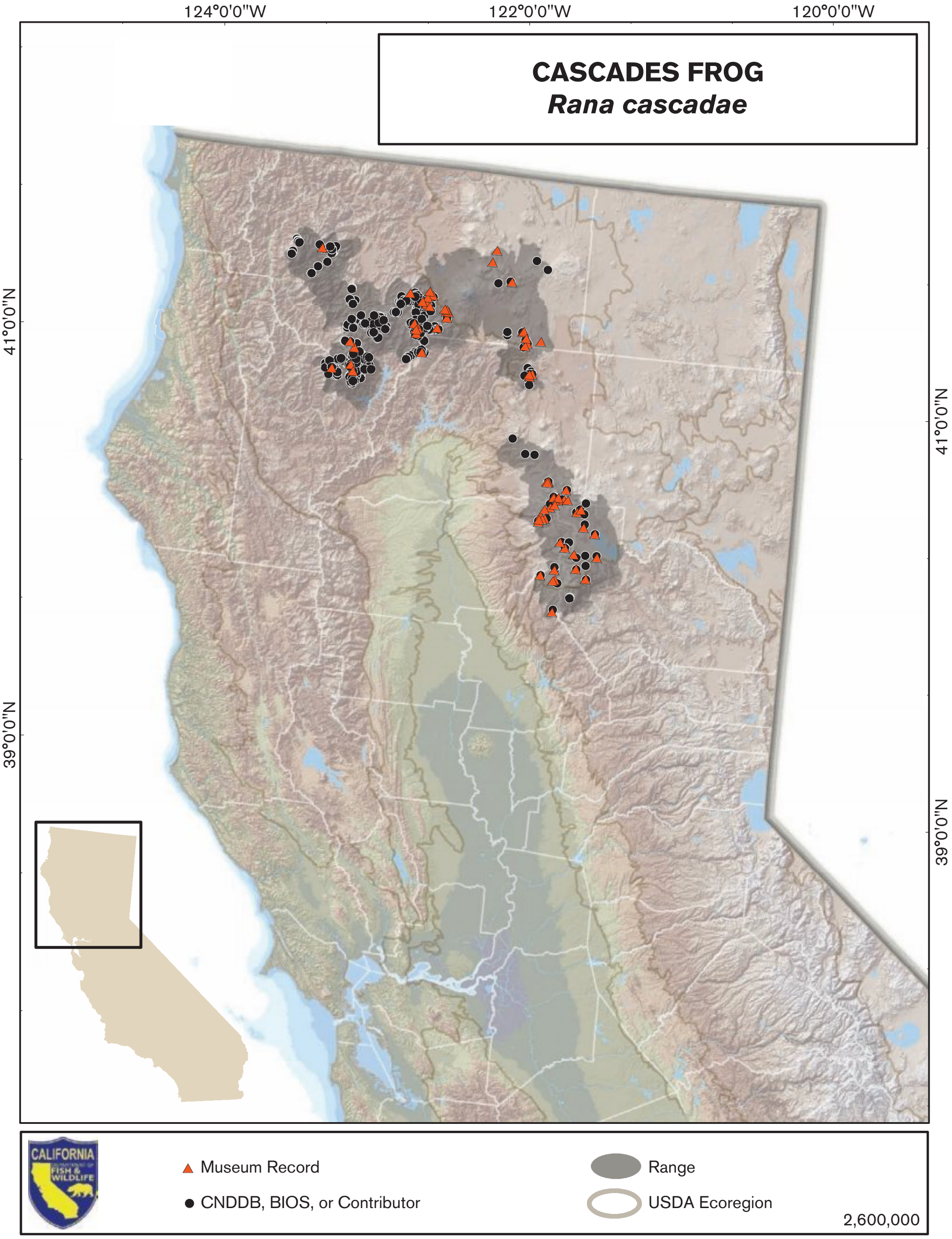


PHOTO ON PREVIOUS PAGE: Cascades frog, Trinity County, California. Courtesy of Adam Clause.

In California, this species could be confused with the California or northern red-legged frogs (*R. draytonii* and *R. aurora*), both of which it resembles in overall body shape. In adults, *R. aurora*/*R. draytonii* have extensive mottling on the venter with red pigmentation on the ventral thighs and groin, rather than the yellow that often characterizes *R. cascadae* (Dunlap 1955). However, the color of the thighs is variable in *R. cascadae* and may not be a reliable character to separate these taxa (S. Barry, pers. comm.).

Taxonomic Relationships

This species is closely related to *Rana aurora* and *R. draytonii* (Shaffer et al. 2004, Hillis and Wilcox 2005). It was proposed as a distinct species based on morphology (Slater 1939), and this interpretation has been repeatedly confirmed with additional morphological and genetic data (Dunlap 1955, Case 1978, Shaffer et al. 2004).

Based on genetic data, the species appears to show considerable differentiation among local populations that is consistent with an overall isolation-by-distance model of gene flow (Monsen and Blouin 2003, Monsen and Blouin 2004). In addition, the California populations appear to be strongly divergent in both mitochondrial and nuclear DNA from the remainder of the species' range in the Cascade and Olympic mountain ranges (Monsen and Blouin 2003). Populations of this species appear to have consistently small effective population sizes (<50; Phillipsen et al. 2011). The available data are primarily from outside of California, although the observed pattern is likely consistent throughout the range. Preliminary results based on mitochondrial DNA, nuclear DNA, and microsatellites suggested little divergence between Lassen and Klamath populations in California (Chang and Shaffer 2010). However, more extensive work with larger range-wide sampling is needed.

Life History

Rana cascadae breeds in the spring, soon after emerging from hibernation and the spring thaw that opens breeding pools (Nussbaum et al. 1983, Stebbins 2003). First-time breeders frequently disperse to new areas of suitable breeding habitat (51% of first-time breeders relative to only 7% of experienced breeders in Echo Lake Basin; Garwood 2009), which may help to connect local subpopulations into larger more stable metapopulations. Breeding occurs at the margins of waterbodies, with oviposition often occurring in large aggregations (Sype 1975, Nussbaum et al. 1983, Garwood 2009). Oviposition behavior appears to be variable throughout the species' range, with some authors reporting diurnal oviposition of largely unattached egg masses (Briggs 1987), and others noting that most egg masses are deposited at night and are attached to vegetation (Nussbaum et al. 1983; K. Pope, pers. comm.). Breeding at individual sites is relatively synchronous and occurs over a few days, although the timing of breeding across the range can vary widely with local weather conditions and elevation (Briggs 1987, Garwood 2009). Embryo development can occur at temperatures ranging from 6°C to 27°C (Sype 1975, Nussbaum et al. 1983). After hatching, larvae sometimes aggregate into dense clusters (generally fewer than 40 individuals) composed primarily of siblings (O'Hara and Blaustein 1981, O'Hara and Blaustein 1985, Blaustein and O'Hara 1987) and choose higher water temperature than those required during embryo development (up to ~28°C; Wollmuth et al. 1987, Bancroft et al. 2008). After metamorphosis, lower water temperatures are again preferred.

Rana cascadae appears to be largely diurnal. The diet of adult frogs is generalized and includes a wide variety of arthropods, as is the case for most other California ranids (Joseph et al. 2011). An analysis of stomach contents for 275 frogs documented the presence of 110 invertebrate taxa (Larson 2012). Frogs across all size classes generally avoided small prey items (<4 mm), and larger frogs more strongly preferred large prey items (Larson 2012).

Habitat Requirements

Rana cascadae utilizes a wide variety of aquatic habitats, including temporary and permanent ponds, lakes, marshes, and streams, as well as adjacent vegetated terrestrial habitat (Nussbaum et al. 1983, Jennings and Hayes 1994a, Stebbins 2003, Pearl and Adams 2005, Garwood 2009). The species will also use wet meadows (often those that have formed from old sphagnum bogs) and can occasionally be found a large distance from water (Nussbaum et al. 1983). They require water year-round at all life stages and cannot tolerate habitats that freeze solid in the winter (K. Pope, pers. comm.). Montane lentic habitat is required for breeding and overwintering, with small, shallow, spring-fed ponds serving as the primary breeding habitat (Garwood 2009). Populations appear to be sustained by a matrix of varying habitat types that individual frogs disperse among throughout the year (Garwood and Welsh 2007, Garwood 2009), suggesting that habitat conservation needs to consider spatial scales larger than single lakes (or other patches of habitat). The presence of predaceous fish may limit their distribution (Welsh et al. 2006, Pope et al. 2008), although this alone cannot explain the broadscale pattern of population declines in this species (Fellers et al. 2008; also see the "Nature and Degree of Threat" section below).

Distribution (Past and Present)

In California, *Rana cascadae* occurs in two population segments. One is in the Lassen area and the extreme northern end of the Sierra Nevada (Lassen, Plumas, Shasta, and Tehama Counties) and is now nearly extirpated. The other occurs in the Trinity Alps and Siskiyou Mountains region. The species' range in California extends from Siskiyou County south to the northern end of Butte County. Outside of California, the range of *R. cascadae* follows the Cascade Range nearly to the United States–Canadian border, with another disjunct population at high elevations on the Olympic Peninsula (Stebbins 2003).

Trends in Abundance

Populations of this frog have declined strongly in the Lassen area, where nearly all known populations have disappeared in the last 30 years (Fellers and Drost 1993, Jennings and Hayes 1994a). More recent surveys in the Lassen region further confirm these declines. *Rana cascadae* was found at only 6 of 856 sites surveyed over 14 years, population sizes were small, and breeding was limited at these 6 sites (Fellers et al. 2008). Populations elsewhere, including the Klamath Mountains region in Siskiyou and Trinity Counties, are also fragmented, generally small, and at risk, although they are more intact overall than in the Lassen area (K. Pope, pers. comm.). Localized declines have also been detected elsewhere in the range (Pearl and Adams 2005, Fellers et al. 2008, Piovia-Scott et al. 2011). Welsh et al. (2006) found *R. cascadae* to be the most common anuran in the Klamath region. By contrast, more recent and ongoing surveys of eight populations in the Trinity Alps within the Klamath region find that only one of the populations is large and robust and that some of the threats present in the Lassen region are likely also operating there (K. Pope, pers. comm.). Pope and Larson (2013) report 11 remaining populations in the Lassen area and find that the number of young frogs was low at all sites that they surveyed.

Nature and Degree of Threat

Threats to this species appear to be complex and derived from multiple stressors. The largest factor contributing to declines in the Lassen region appears to be overall low recruitment due to changing hydrological conditions that lead to detrimentally high water temperatures and desiccation of egg masses and tadpoles, as well as impacts from *Bd* among subadult frogs (Pope et al. 2011). Extensive mark-recapture surveys in the Lassen region between 2008 and 2010 indicate widespread desiccation of egg masses and tadpoles and a lack of metamorphs relative to more stable populations in the Klamath area (Pope et al. 2011, Pope and Larson 2013). In comparisons between two of the remaining Lassen populations, the population with higher *Bd* prevalence and load in adult and subadult frogs had lower survivorship for these two age classes. In both Lassen and the Klamath Ranges, subadult frogs had higher *Bd* prevalence and load than adult frogs, and the prevalence of *Bd* increased throughout the active season for subadult frogs but not for adult frogs. These results are consistent with previous studies of *Bd* in this species that suggest the pathogen has differential impacts depending on age class. Blaustein et al. (2005) examined the effect of *Bd* on larvae and found an increased incidence of mouthpart abnormalities but no effect on mortality or behavior. Garcia et al. (2006), however, found significant mortality in new metamorphs of *Rana cascadae* due to *Bd*.

Interestingly, *Bd* also appears to be widespread in the Klamath region where this species is currently much more stable than in the Lassen region (Piovia-Scott et al. 2011), suggesting more than one factor is playing a role in the declines. Ongoing characterization of *Bd* prevalence in these populations could help determine what factors are involved, although one hypothesis is that *Bd* achieves higher loads on frogs in declining populations than stable populations (J. Piovia-Scott, pers. comm.). Infection by the water mold *Saprolegnia* has also been implicated in *R. cascadae* declines. This pathogen is known to increase mortality in embryos, larvae, and metamorphs (Kiesecker and Blaustein 1999, Romansic et al. 2009a) and may have strong impacts on the outcome of competition between *R. cascadae* and sympatric *Pseudacris regilla* (Kiesecker and Blaustein 1999).

Habitat loss and modification is also a threat to continued persistence of populations in both the Klamath and Lassen regions. The species is highly associated with meadows, which have been impacted by cattle grazing, tree encroachment due to lack of wildfire, and changing hydrology associated with changes in the snowpack (K. Pope, pers. comm.; Pope et al. 2014).

Other possible contributors to *R. cascadae* declines that have been proposed include introduced fishes, environmental contaminants, pathogens, and UV-B radiation. The presence of introduced trout appears to be inversely related to the distribution of *R. cascadae* (Welsh et al. 2006) and almost certainly impacts some populations. Aside from direct predation, introduced trout may affect *R. cascadae* indirectly by supporting higher populations of the aquatic garter snake (*T. atratus*), a predator on both trout and *R. cascadae* (Garwood and Welsh 2007, Pope et al. 2008), and by preemptive competition for aquatic prey (Joseph et al. 2011). However, trout have been present in the Lassen region for nearly a century and are also widely distributed in other areas where *R. cascadae* persists, making it unlikely that they alone can explain the declines over the last 30 years (Fellers et al. 2008). Nevertheless, populations appear to respond favorably to trout removal, showing marked increases in population size and recruitment following fish removal (Pope 2008).

Pesticide use is inversely correlated with the presence of *R. cascadae* (Davidson 2004). In particular, downwind transport of pesticides from intensively farmed areas in the Central Valley appears to be correlated with declines in several species of ranid frogs, including *R. cascadae* (Davidson et al. 2002, Davidson 2004). This hypothesis is attractive in that it explains the differential declines between the Trinity Alps region and the Lassen region because the Lassen region is directly downwind of areas that experience heavy agricultural use, whereas the Trinity Alps are not (Davidson et al. 2002). However, recent field measurements of contaminant residues in sediment and in *R. cascadae* and *P. regilla* tissue do not indicate higher levels in the Lassen compared to the Trinity Alps region, at least for the handful of different chemicals that have been analyzed to date, calling this hypothesis into question (Davidson et al. 2012). In addition, Sparling et al. (2001) measured the presence of cholinesterase levels in the non-declining *P. regilla* as a measure of the extent of pesticides that are locally deposited in an area and found strong effects in the Sierra Nevada but not in the Lassen area; these results seem to indicate that pesticides may not be a major factor in the Lassen *R. cascadae* declines. Environmental contaminants at sublethal levels have also been shown to induce behavioral and morphological changes in *R. cascadae* (Marco and Blaustein 1999), suggesting that low-level agricultural residues may have important biological consequences. In summary, it appears that pesticides may be playing some role in *R. cascadae* declines in the Lassen region, but they are certainly not the entire story.

Finally, UV-B radiation may play a role, possibly in combination with other factors, in causing declines. Some studies have documented larval mortality and retinal damage due to UV-B, although the effect depends strongly on the intensity of UV-B, the duration of exposure, and possibly other factors including the presence of competitors, predators, or supplementary food (Fite et al. 1998, Hatch and Blaustein 2000, Belden et al. 2003, Garcia et al. 2006, Romansic et al. 2009b). The importance of these results has not yet been demonstrated in natural settings, however. Palen et al. (2002) found that dissolved organic matter in natural environments provided protection from UV-B at 89% of the sites examined for *R. cascadae*. Thus, it remains possible that UV-B is having an effect, although its importance in nature remains unclear.

Ultimately, it is likely that no one factor is solely responsible for the precipitous declines in Lassen region *R. cascadae* populations. Further, the causes of the initial range-wide declines may be distinct from the local factors that threaten the continued persistence of the few remaining populations. The most recent work suggests that the major factors playing a role in the range-wide declines are the presence of introduced fishes and *Bd*, while continued local persistence of the remaining populations is also threatened by low recruitment stemming from desiccation and detrimentally high water temperatures. Pope et al. (2014) present a recent and comprehensive review of both regional and local-scale threats to *R. cascadae* throughout the range. The evidence that synergistic effects occur between several alternative mechanisms of decline is now widespread for a variety of amphibian species (Fellers et al. 2008). As declines have occurred, whatever the cause, it is likely that a breakdown of metapopulation dynamics will contribute to further declines as existing populations become more and more fragmented, decreasing the opportunity for population rescue via recolonization.

Status Determination

The catastrophic declines in the Lassen area are the primary reason for the SSC designation. *Rana cascadae* is nearly extirpated in the Lassen region, is undergoing local population declines elsewhere in its range, and appears to be susceptible to a wide range of threats. However, this frog is a moderate ecological specialist that appears to be relatively stable through much of its range, including a significant fraction of its range in California. The factors that caused declines in the Lassen area appear to not have operated in the Klamath area to date, leading us to project moderate future impacts on extant populations and a Priority 2 status. If strong declines begin to occur in the Klamath area, then a higher priority status will rapidly become justifiable.

Management Recommendations

Fellers et al. (2008), Pope et al. (2011, 2014), and Pope and Larson (2013) provide thorough reviews of threats to, and management recommendations for, *Rana cascadae*, and our recommendations largely follow those of these authors.

Habitat that supports this species in the Lassen area should be protected from modification that negatively impacts hydrology while further research is carried out. Pope et al. (2011) began some habitat restoration measures, and these efforts should be continued (coupled with ongoing monitoring to determine their effects). Fish removal in key populations has also been documented to increase recruitment and should be considered as a management strategy, particularly in the Klamath where a larger number of existing populations might be stabilized before declines can occur. Pope et al. (2011) also proposed experimental treatment for *Bd* in newly metamorphosed frogs. Effective treatments for *Bd* may be essential for the long-term survival of many amphibian species, so these efforts should be further explored and potentially implemented if they are successful. At the same time, a captive colony of Lassen-area *R. cascadae* should be established, as the prospects for long-term survival in the wild appears to be low. If additional research can determine the causes of the declines and effective mitigation measures can be enacted, this captive population could eventually form the basis of a reintroduction program.

Monitoring, Research, and Survey Needs

Monitoring efforts should focus on the few remaining Lassen populations, with additional monitoring of stable populations elsewhere in the range as reference populations. Areas that have undergone habitat restoration or experimental treatments for *Bd* infection will require ongoing monitoring to quantify the long-term effects of these efforts and to inform further work aimed at controlling the impact of these threats. If additional declines occur, this monitoring will facilitate early detection and, hopefully, provide the background data needed to understand the causes of declines.

As a reintroduction effort may eventually become necessary, it is important to further characterize the extent of intraspecific variation within this taxon now, before additional declines occur. Preliminary genetic work has been initiated, and it should form the basis of additional work that examines fine-scale population differentiation and structure. The obvious initial focus of such genetic work should be to assess the validity of the Lassen and Klamath regions as separate evolutionary units requiring their own management strategies. This work will also help to identify any potential population segments within either region that may qualify for independent management. Finally, additional studies that quantify the interactive effects among different causes of declines would be useful in providing a more complete picture of conservation threats in this taxon.

CALIFORNIA RED-LEGGED FROG

Rana draytonii Baird and Girard 1852

Status Summary

Rana draytonii is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 76% (84/110). During the previous evaluation, it was also considered a Species of Special Concern (Jennings and Hayes 1994a), and it has been listed as federally Threatened since 1996.

Identification

Rana draytonii is a relatively large (2.5–13.8 cm SVL) brown, gray, olive, or reddish-brown frog (Jennings and Hayes 1994a, USFWS 2002, Stebbins 2003). Prominent dorsolateral folds are usually present. Many small black flecks and larger irregular blotches are present on the back, and these occasionally form a network (Baird and Girard 1852). The larger black spots on the back often have a whitish or light center. The ventral surface is whitish or cream with extensive gray or black mottling, often overlain with red or reddish-orange coloration, particularly in the groin (Baird and Girard 1852, Stebbins 2003). In general, the red coloration in this species is individually and ontogenetically variable, with the undersides of the feet almost always red in adult animals, although the extent of red elsewhere on the legs and belly varies

California Red-Legged Frog: 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)	3
vi. Population trend (25)	25
vii. Vulnerability to climate change (10)	7
viii. Projected impacts (10)	7
Total Score	84
Total Possible	110
Total Score/Total Possible	0.76

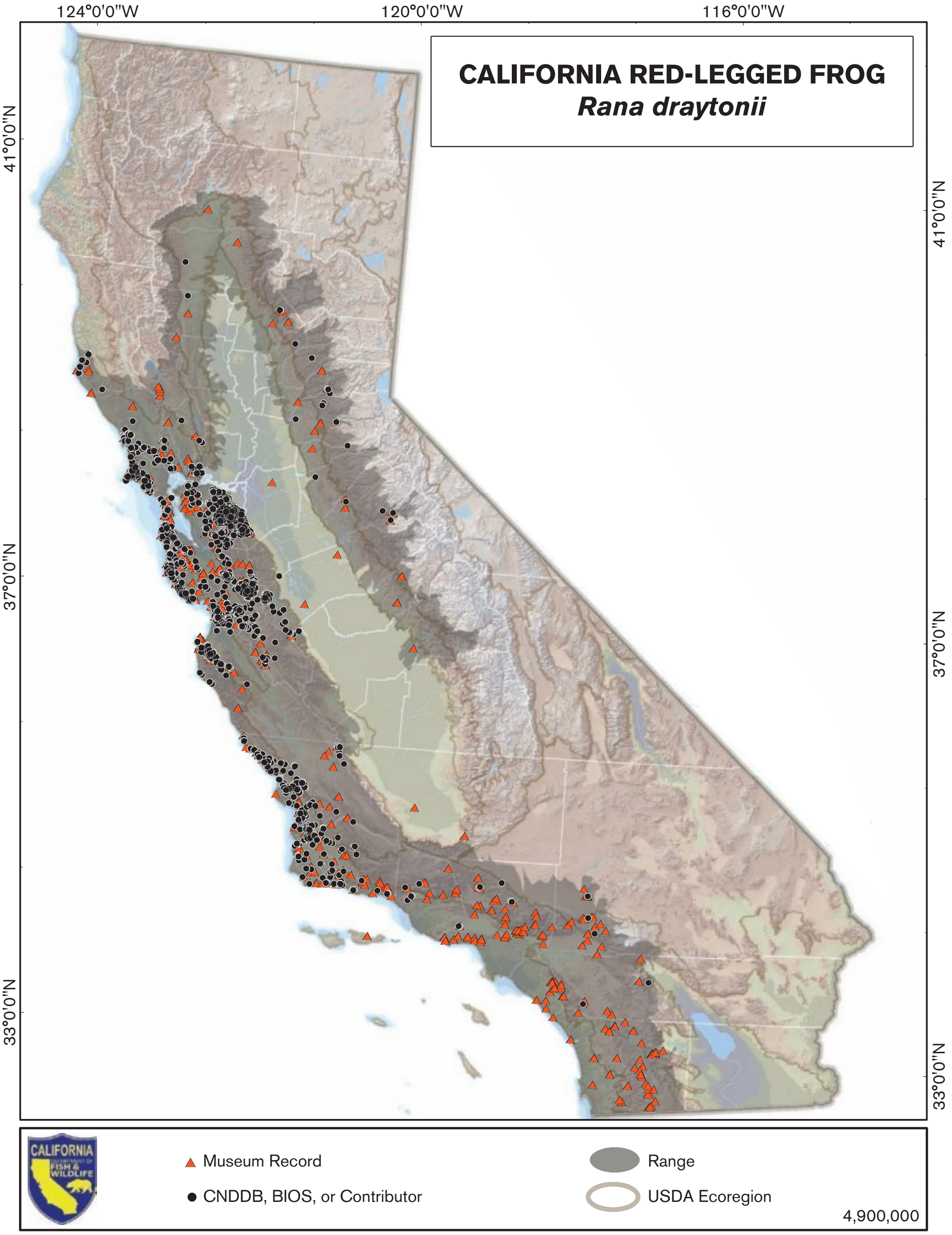


PHOTO ON PREVIOUS PAGE: California red-legged frog, Alameda County, California. Courtesy of Adam Clause.

from extensive to absent (S. Barry, pers. comm.). The advertisement call is a series of low guttural chucks sometimes followed by a low groan (Elliott et al. 2009).

This species could be confused with the northern red-legged frog (*R. aurora*) where their ranges meet in southern Mendocino County (Shaffer et al. 2004). *Rana aurora* is about 3.5–4.0 cm (SVL) smaller than *R. draytonii*, generally lacks light areas in the centers of dorsal blotches, has proportionally smaller eyes, and lacks vocal sacs (Baird and Girard 1852, Hayes and Krempels 1986, Stebbins 2003).

Taxonomic Relationships

Rana draytonii was initially described as a distinct species, although the original description notes that it is similar in appearance to *R. aurora* (Baird and Girard 1852). Subsequent to the original description, Camp (1917) reclassified the two red-legged frogs as subspecies of a polytypic *R. aurora*. This arrangement persisted, occasionally also including the Cascades frog (*R. cascadae*) as a third subspecies, until the mid-1980s. At this time, a series of studies emerged suggesting that a substantial amount of differentiation between the two forms was present in allozymes, morphology, calling behavior, and oviposition behavior, leading several authors to suggest that they may be distinct lineages with a broad zone of contact (Hayes and Miyamoto 1984, Green 1986a, Green 1986b, Hayes and Kremples 1986). Subsequent analyses of mitochondrial DNA variation supported this view but characterized a narrow zone of contact in southern Mendocino County (Shaffer et al. 2004). Based on both DNA and morphological differentiation, Shaffer et al. (2004) suggested that the two be recognized as distinct species, and since then *R. draytonii* has increasingly been recognized as a species distinct from both *R. aurora* and *R. cascadae*.

Life History

Few data are available on seasonal activity patterns, but coastal populations are probably active throughout much of the year due to the moderating effect that the Pacific Ocean has on temperature. The timing of reproduction varies from year to year and according to site but occurs from late November to late April (Storer 1925, Fellers 2005b). Breeding occurs in the water, and eggs are attached to emergent vegetation (in clusters of 300 to >4000; Storer 1925, Hayes and Miyamoto 1984). Hatching occurs in 6–14 days depending on water temperature, after which larvae metamorphose in 3.5–7 months (Storer 1925, Wright and Wright 1949). Larvae are known to overwinter at several sites, metamorphosing the following spring (Fellers et al. 2001). *Rana draytonii* is a generalist predator that feeds predominantly on invertebrates but has also been documented to take vertebrate prey including Pacific treefrogs (*Pseudacris regilla*), western toads (*Bufo boreas*), and California mice (*Peromyscus californicus*) (Hayes and Tennant 1985, Arnold and Halliday 1986, USFWS 2002, Davidson 2010). The prey types taken appear to be determined by the size of the frogs, with individual frogs taking most prey types that they can successfully swallow (Jennings and Hayes 1994a) and large frogs taking proportionally more vertebrate prey than small ones. Baldwin and Stanford (1987) reported a large adult preying upon California tiger salamander (*Ambystoma californiense*) larvae. *Rana draytonii* feed both in the water and by foraging in dense riparian vegetation. *Rana draytonii* is active both diurnally and nocturnally, although adults are generally more active at night (Hayes and Tennant 1985; G. Fellers, pers. comm.).

Wading birds, raccoons, and garter snakes (*Thamnophis sirtalis* and *T. hammondii*) are important native predators on this species (Cunningham 1959b, Jennings and Hayes 1994a). Nonnative fishes are also important predators on larvae and recent metamorphs (Schmieder and Nauman 1994, USFWS 1999).

Habitat Requirements

Rana draytonii chiefly inhabits ponds, although it also uses marshes, streams, lagoons, and other waterways throughout most of its range.

In southern California (from Ventura County southward) it seems to favor slow-flowing streams rather than ponds or pools. Breeding takes place primarily in ponds (at least in central and northern California) and less frequently in quiet pools in streams (Stebbins 2003, Fellers 2005b). This species will also utilize ephemeral water bodies for breeding, although nearby permanent water is probably required to maintain populations over the long term (Jennings 1988a). After breeding, adults often disperse along nearby shaded streams. Similar to *R. boylii*, whose vulnerable early life stages (embryos and tadpoles) are susceptible to ill-timed flow fluctuations controlled by upstream dams and diversions, *R. draytonii* populations breeding in stream habitats suffer from decreased recruitment after anthropogenic perturbation of natural flow regimes (S. Kupferberg, pers. comm.).

Optimal aquatic habitat has traditionally been thought to include dense riparian vegetation overhanging deep (>0.7 m) slow-moving pools (Hayes and Jennings 1988). More recent work has documented an additional, more complex relationship between aquatic vegetation and introduced bullfrogs. D'Amore et al. (2009) documented that *R. draytonii* spend more time in vegetative cover when bullfrogs are present and more time in the open when bullfrogs are removed from ponds, suggesting that the optimal amount of vegetation is somewhat context-dependent for *R. draytonii*. In addition, surveys of 85 ponds occupied by *R. draytonii* in the East Bay Regional Park District showed that there were no significant differences in adult frog density among ponds with 0%, ≤15%, or >15% emergent vegetation, but tadpoles and metamorphs were more abundant in the most open ponds (Bobzien and DiDonato 2007). Outside of the breeding season when conditions are wet, and especially during rainfall, adult frogs will disperse from the breeding habitat and will move to upland sites, where they are often found under logs, rocks, and other debris (USFWS 2002, Bulger et al. 2003, Fellers and Kleeman 2007). At some sites, populations appear to consist of both migratory (11–22% of the adult population) frogs that move 200–2800 m and resident frogs that remain at the breeding site (Bulger et al. 2003). Fellers and Kleeman (2007) found that adult female frogs were more frequently migratory than males, although migration behavior did not differ between the sexes among those individuals that did migrate.

Distribution (Past and Present)

Historically, *Rana draytonii* ranged throughout the Sierra Nevada foothills and the Coast Range mountains south of Elk Creek in southern Mendocino County, California, southward to the Arroyo Santo Domingo, Baja California Norte, Mexico (Hayes and Krempels 1986, Jennings and Hayes 1994a, Grismer 2002, Shaffer et al. 2004). In California, this taxon historically ranged through at least 46 counties, but it is now apparently extirpated from 24 of these (USFWS 1996). It is unclear whether reproductive populations of *R. draytonii* were present in most of the Central Valley, and it is possible that the few valley records represent waifs washed downstream from Sierran populations (G. Fellers, pers. comm.; S. Barry, pers. comm.). If they were present in the Central Valley, they were extirpated before 1960. Populations in the Sierra Nevada may have been connected to the largest remaining populations of the species in the Coast Ranges through the lower Cascade and Tehachapi Ranges (S. Barry, pers. comm.), but today they are isolated (USFWS 2002). A recent comprehensive survey of museum specimens and historical records identified 21 historical localities for this species in the Sierra Nevada. Follow-up surveys at 20 of these 21 sites found that the species persists in large numbers in at least 1 site, there are populations at 6 additional sites, and at least a single individual documented at 3 more sites (Barry and Fellers 2013).

Strong overall declines have clearly occurred across most of the large range of this species, particularly in the southern portion of the range. In the Bay Area and Coast Ranges, populations are more robust, although severe localized declines have been documented (reviewed in USFWS 2002). In southern California, *R. draytonii* has declined drastically through the Transverse and Peninsular Ranges, and very few populations now persist in Los Angeles, Riverside, and Ventura Counties (USFWS 2002, and references therein). One population is known from Santa Cruz Island, although this apparently is an introduction (Sweet and Leviton 1983, Jennings 1988b). The known elevational range of *R. draytonii* occurs from near sea level to 1500 m, although most populations occur below 1050 m (USFWS 2002, Barry and Fellers 2013). Some higher-elevation populations may be introductions (unpublished data reported in Jennings and Hayes 1994a).

Trends in Abundance

Drastic and ongoing declines have been documented throughout parts of this species' range. Many of these declines have resulted in extirpation of populations, and in many areas where this taxon persists, declines in abundance have occurred. Food market collection in the late 1800s apparently drove much of the initial declines (Jennings and Hayes 1985). By 1879, the species had already become rare around San Francisco due to the market trade (Lockington 1879). Population trends of the species in the Sierra Nevada foothills are somewhat unclear, since several new, large (>100 breeding adults) populations have recently been discovered (e.g., in Placer County). However, in southern California, population densities are uniformly low (<25 adults frogs) and generally declining (USFWS 2002). The sole remaining population known in Riverside County at the Santa Rosa Plateau, which was at least somewhat genetically distinctive (Shaffer et al. 2004), is now extirpated.

Nature and Degree of Threat

The largest threat facing *Rana draytonii* is probably habitat loss and alteration, resulting from urbanization and agriculture. The large-scale conversion of habitat to agricultural uses has also resulted in an increase in pesticide exposure, which may have strong negative impacts on this species (Davidson et al. 2002). This effect is particularly strong for cholinesterase-inhibiting pesticides (Davidson 2004), although the species still persists in some heavily agricultural settings in Monterey and Santa Cruz Counties. Additional and ongoing fragmentation of habitats, conversion of wetlands to other uses, and modifications to the hydrology of wetlands also likely have detrimental impacts.

The effect of introduced species, in particular bullfrogs, has been studied both empirically (Moyle 1973) and from a modeling perspective (Doubledee et al. 2003). There is a strong overall negative impact of bullfrogs on native *R. draytonii*, although coexistence of the two species can occur in nature. Human-modified aquatic and terrestrial habitats in central California (Elkhorn Slough, Monterey County) favor introduced bullfrogs compared to native *R. draytonii* (D'Amore et al. 2010). The bullfrog is also a strong competitor with, and predator on, multiple life stages of *R. draytonii*. In addition, crayfish, mosquitofish (Lawler et al. 1999), and other introduced predaceous fishes likely have negative impacts on this species, although this also needs further study (Hayes and Jennings 1986, Fisher and Shaffer 1996, Fellers 2005b).

Chytrid fungus (*Bd*) is known to have caused serious declines in many amphibian species and has been detected in *R. draytonii* in nature. However, the direct impact *Bd* has on *R. draytonii* appears to be relatively slight. In a laboratory setting, *R. draytonii* is susceptible to chytrid infection, but frogs can clear their infections, do not die from the infection, and suffer no growth consequences when they have access to unlimited food (Padgett-Flohr 2008). In nature, across a landscape of ponds where *Bd* presence and absence fluctuated between wet and dry years, *R. draytonii* were generally uninfected and found to be significantly associated with uninfected ponds (Padgett-Flohr 2010).

Predicted climate change over much of California will affect *R. draytonii*, as well as most other pond- and stream-breeding amphibians. In particular, warmer average temperatures, generally reduced levels of precipitation, and increased variability in the timing of rainfall are all predicted to occur (PRBO 2011). While the precise effects of these shifts will vary regionally and at the watershed level, the permanence and reliability of breeding sites are generally predicted to decrease under climate change predictions.

Status Determination

Rana draytonii automatically qualifies as a Species of Special Concern because it is listed under the federal but not state Endangered Species Act. However, sharp declines in both range and abundance, coupled with a variety of ongoing threats to long-term survival, also combine to warrant a Priority 1 Species of Special Concern status.

Management Recommendations

Management of *Rana draytonii* should mirror the guidelines in the USFWS recovery plan for this taxon (USFWS 2002). As further management needs are defined and existing management strategies are refined (through 5-year reviews or other avenues), state-level management should be adjusted accordingly.

The most important management needs for this taxon currently are the protection of habitat that supports the species, reduced pesticide exposure, and elimination of nonnative predators. Land conversion and additional fragmentation should be avoided wherever possible, and adequate, complex upland habitat should be available in order to allow migration to occur naturally. Fellers and Kleeman (2007) found that the median distance of movement away from breeding ponds was 150 m and that there were some long-distance movements up to 1400 m. Unpublished radiotelemetry observations from the East Bay Regional Park District (S. Kupferberg, pers. comm.) demonstrated that ground squirrel burrow density, sometimes more than 100 m from the aquatic habitat, was also a key component of habitat quality. Taken together, these studies indicate that large tracts of terrestrial habitat are important (to accommodate both short- and long-distance dispersal) and that a healthy population of ground squirrels (and possibly other burrowing rodents) may be essential for long-term population viability.

Finally, pesticide use should be curtailed in areas where this species occurs, including areas upwind where pesticides are likely to be blown into areas that support this species. Unpublished data from the East Bay Regional Park District (S. Kupferberg, pers. comm.) indicate that cattle-grazing does not appear to negatively impact this species.

Monitoring, Research, and Survey Needs

Further research is needed to determine what the precise impacts many of these threats identified above are having on *Rana draytonii*. Surveys of private land in the Sierra Nevada are slowly revealing the presence of extant populations that were previously missed (S. Barry, pers. comm.), suggesting that this may be a fruitful strategy elsewhere in the range as well. Managers should partner with private landowners to gain access and survey for remaining populations of this species in areas where it has previously been thought to be extirpated, and these populations, which may be very small in size, should be monitored regularly.

Finally, the only range-wide genetic analysis of the species thus far conducted was based purely on mitochondrial DNA (Shaffer et al. 2004), and supporting data from a large set of nuclear DNA markers is badly needed. In particular, the potential genetic break between populations north and south of Santa Barbara County, and the genetic affinities of remnant populations from southern California and Baja California, Mexico, will form an important part of future management.

Additional monitoring, research, and survey needs are covered in depth in the USFWS recovery plan for this taxon. We refer the reader to this document for more information (USFWS 2002).

NORTHERN LEOPARD FROG

Rana pipiens Schreber 1782

Status Summary

Rana pipiens is a Priority 1 Species of Special Concern, receiving a Total Score/Total Possible of 73% (80/110). During the previous evaluation, it was also designated as a Species of Special Concern (Jennings and Hayes 1994a).

Identification

Rana pipiens is a medium-sized ranid frog with strong, continuous dorsolateral folds that do not angle inward posteriorly. Its dorsal coloration is green to brown with large well-defined black or dark-brown oval or round spots. Each spot is ringed with a narrow band of white or cream. The ventral coloration is white or cream with no mottling or other dark markings (Stebbins 2003). The call is a low, snore-like trill, often followed by low chuckling and/or grunts (Stebbins 2003, Elliott et al. 2009).

Within its range in California, this species can potentially be confused with the Oregon spotted frog (*R. pretiosa*). However, *R. pretiosa* has much smaller, more irregular spots, which often have diffuse borders and are not ringed in white. It also has conspicuous red or salmon markings on the underside, which *R. pipiens* lacks. Other members of the leopard frog complex in California, the lowland leopard frog

Northern 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)	0
vi. Population trend (25)	20
vii. Vulnerability to climate change (10)	10
viii. Projected impacts (10)	10
Total Score	80
Total Possible	110
Total Score/Total Possible	0.73

PHOTO ON PREVIOUS PAGE: Northern leopard frog, Washington County, Utah. Courtesy of William Flaxington.

(*R. yavapaiensis*) and the Rio Grande leopard frog (*R. berlandieri*), have dorsolateral folds that are discontinuous and angle inward posteriorly. In addition, both are yellow ventrally. The Cascades frog (*R. cascadae*) has more numerous, small, irregular black dots that are not ringed in white.

Taxonomic Relationships

The taxonomic history of the leopard frog species complex, and *Rana pipiens* in particular, is complicated (Hillis 1988) and remains incompletely understood. The name *R. pipiens* previously included all members of the leopard frog complex from Canada south to Panama, including *R. yavapaiensis*, also native in California, and the introduced *R. berlandieri*. However, this concept of a single wide-ranging leopard frog species changed in the last several decades, and over a dozen species are recognized at present. The current taxonomy of the *R. pipiens* complex was initially based on variation in morphology and vocalizations (Pace 1974). Subsequent work including molecular analyses recognized several additional taxa and clarified relationships among the contained species (Platz and Mecham 1979, Hillis et al. 1983, Platz and Frost 1984, reviewed by Hillis 1988).

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 pending further analyses.

Life History

No life history data for California populations have been published. Because *Rana pipiens* in California are a mixture of introduced and presumably native populations (see the "Distribution" section) and live on the extreme western edge of the species' range, we are reluctant to use information from more easterly populations as a proxy for those that occur in California. In Colorado, breeding occurs during the first spring nights that have relatively "mild" temperatures near or above freezing (Corn and Livo 1989), and this presumably is also the case in California. Tadpoles are present through the summer months and are not known to overwinter, suggesting a late summer or fall metamorphosis. Further east, adults and juveniles are known to range far from water and breeding sites (Dole 1971), although it is unknown if this also characterizes California populations. Range-wide, *R. pipiens* is a generalist predator, feeding on a wide variety of arthropods and small vertebrates (Knowlton 1944, Linzey 1967, Harding 1997), and this presumably also characterizes the species in California.

Habitat Requirements

Despite the paucity of records from California, this species is known from a variety of habitats, including small streams, rivers, and lakes (Storer 1925, Stebbins 1951, Jennings and Hayes 1994a). *Rana pipiens* occupies a wide variety of habitat types throughout its range, so we are hesitant to speculate on microhabitat requirements in California. Generally, the species hibernates underwater and requires aquatic habitats that do not freeze solid during winter (Emery et al. 1972, Licht 1991), and this presumably is also the case for California populations. Nearby damp upland habitat is utilized for foraging during the active season (Dole 1967). The species has been found in a variety of open grassy areas and meadows, although heavily grazed areas and cultivated fields do not appear to be suitable (Pope et al. 2000). In the Midwestern United States, the presence of quality upland foraging habitat seems to affect the abundance of this species. When grasslands were restored around suitable pond-breeding habitat, the density of frogs increased markedly (K. Mierzwa, pers. comm., in Pope et al. 2000).

Distribution (Past and Present)

Outside of California, *Rana pipiens* ranges widely across North America, from Nova Scotia and Newfoundland, Canada, west to Washington and Nevada. In California, *R. pipiens* populations that may be native are known from Modoc and Siskiyou Counties, the Lake Tahoe basin, and the upper Owens Valley (Jennings and Fuller 2004), although some workers question whether the latter two regions constitute natural, as opposed to purely introduced, populations (S. Barry, pers. comm.). Numerous introductions have occurred throughout the state, including some within the putative native range. The vicinity of Fallen Leaf Lake in the Lake Tahoe Basin is one such example (Bryant 1917). It is also possible that putatively native populations of this frog are all the result of human introductions, and determining their status is an important research priority. The upper Owens Valley supports tiger salamander populations that were recently shown to be introduced (Johnson et al. 2010), demonstrating that similarly distributed nonnative species have been established in this region. The tiger salamander introductions occurred as a consequence of the fishbait industry (Riley et al. 2003), which also sometimes sells leopard frog tadpoles and adults.

We are not aware of any additional recent records in California beyond those reported by Jennings and Hayes (1994a), though an unverified sight record of a "spotted frog" in Surprise Valley, Modoc County, California, could have been *R. pipiens*. However, the circumstances and description of this frog make it more likely that it was *R. pretiosa*, another California Species of Special Concern (see that species account for additional information).

Trends in Abundance

Trends in abundance for California populations of *Rana pipiens* are difficult to interpret because of the uncertainty regarding which populations are native or introduced. However, assuming that historical California populations are native, severe declines have clearly occurred. We are aware of only scattered sight records for the species over the last two decades. Jennings and Hayes (1994a) reported two relatively recent sight records in the early 1990s from Siskiyou and Inyo Counties. Macey and Papenfuss (1991a) reported that leopard frogs occurred on the east side of the White Mountains below Boundary Peak, though they failed to detect the species in follow-up surveys (T. Papenfuss, pers. comm.). More recent surveys of historical localities in the Owens River also did not detect this species and found that much of the habitat currently appears to be unsuitable (Becker and Henderson 2010). We are not aware of any presumed-native populations of this species occurring in the state since these records. Elsewhere in its range, *R. pipiens* has undergone severe declines and localized extirpations, particularly in the western parts of the United States (reviewed by Rorabaugh 2005).

Nature and Degree of Threat

Habitat modification is probably the most important threat for *Rana pipiens* in California. *Rana pipiens* forages in upland habitat having moderately tall vegetation with a moist substrate. Livestock grazing in these habitats tends to reduce vegetation height, which leads to drying of the substrate, apparently rendering this habitat unsuitable for the frog. It is likely that this process contributed to the declines observed in both the Owens Valley and the Modoc Plateau areas where most California records for *R. pipiens* are concentrated. Changing hydrology elsewhere in the range has led to the extirpation of some local populations (Corn and Fogleman 1984). Given that California populations are at the western range limit of the species, projected climate changes may have a strong effect in the state. Current models project warmer summer and winter temperatures, decreases of 8–21% of annual precipitation, and a 34% decrease in snowpack (PRBO 2011). Taken together, these climate projections indicate that the moist soil and wetland complexes favored by this species will probably decrease in the Great Basin of California, further reducing the already sparse habitat for this species.

Some studies have detected significant negative impacts from pesticides on *R. pipiens*, although the importance of this threat in nature is not well understood. In other parts of their range, *R. pipiens* are known to be sensitive to herbicides and pesticides used in agriculture (Relyea 2008, Relyea and Jones 2009), and mixtures of these chemicals can result in 99% mortality rates (Relyea 2008). However, the evidence on this topic is complex and dependent on the specific chemicals tested. A popular herbicide consisting of a mixture of glyphosate and POEA (commonly marketed under the commercial name Roundup®) is one such example. Some studies have found limited impacts from these chemicals and concluded that direct mortality in wild populations from this herbicide is unlikely (e.g., Wojtaszek et al. 2004), while other studies have found very strong direct lethal effects (e.g., Relyea 2005b). When direct lethal effects were not found, several studies demonstrated that chemical contaminants can have lethal impacts when combined with other stressors (e.g., predator cues; Relyea 2005a) or sublethal detrimental effects such as decreased immune system functionality (Christin et al. 2003, Gilbertson et al. 2003, Rohr et al. 2008). These seemingly unpredictable effects of agrochemicals may depend on specific populations and conditions in a local area (Relyea 2005b). Although these results are both complex and sometimes contradictory, substantial evidence exists that environmental contaminants are likely to have significant impacts on *R. pipiens* and other amphibians in California (e.g., Davidson et al. 2002, Davidson 2004).

Other potential threats to *R. pipiens* include introduced exotic bullfrogs and predatory fishes, and extensive habitat modification associated with agriculture (Hayes and Jennings 1986).

Status Determination

Rana pipiens' small range in California coupled with severe declines drives the high score for this species. None of these threats are currently being reversed, so it is reasonable to expect additional declines in the future, assuming that native populations still exist in California. *Rana pipiens* is sensitive to localized extirpation due to drought (Corn and Folgeman 1984), and the expected increase in temperature and decrease in precipitation due to climate change are likely to have additional negative impacts. The combination of these factors justifies a Priority 1 status.

Management Recommendations

The development of an effective management strategy will largely depend on finding remnant populations in the state, carrying out research on the life history of those specific populations to determine their habitat needs, and then taking a proactive management and habitat restoration approach to recover it in its native range. A key first step with any remnant population is to determine whether it is native or introduced. Researchers can most easily accomplish this using DNA markers, and we recommend that larval tail tips be collected for any population that is discovered. A considerable amount of phylogenetic work, particularly using mitochondrial DNA markers, has been published for this species, and straightforward DNA sequencing of California animals should allow them to be placed into a phylogenetic context with other *Rana pipiens* from across the species' range. This approach was used by Johnson et al. (2010) and demonstrated that potentially native populations of tiger salamanders (*Ambystoma tigrinum*) were in fact nonnative introductions. If native populations of *R. pipiens* are found, the habitat supporting them should be protected in order to reduce potential threats such as nonnative predators, agricultural disturbance, grazing, off-highway vehicle use, pesticide applications, and changes to local hydrology. If nonnative populations are found, managers should evaluate their potential to spread and pose a threat to other native taxa. In certain cases, removal programs could be effective at mitigating threats posed by nonnative *R. pipiens*.

Monitoring, Research, and Survey Needs

Comprehensive surveys of historical localities as well as the Modoc Plateau area, including the

Goose Lake Basin and the Warner Mountains, should be conducted to determine whether any viable populations persist in California and to identify areas of potential habitat for ongoing surveys. The most recent records for this species come from the vicinity of Owens Valley, and all drainages flowing into the valley should be carefully surveyed. It is critically important that tissue samples be collected from any extant populations that are found so that frogs can be genetically characterized with respect to their introduced or native status.

Given our current lack of information about the life history of this species in California, basic ecological research is a key priority for any native populations that remain in the state. Information about habitat preferences and requirements, demography, and timing of key life history events would all improve our ability to conserve remnant populations of *Rana pipiens*.

Finally, if remnant populations are found, multi-locus microsatellite or single nucleotide polymorphism DNA data should be analyzed to estimate the effective population size and potential connectivity with other remaining populations. If populations are determined to be native, small, and genetically isolated, *R. pipiens* could be a prime candidate for human-mediated translocations to establish new populations in currently unoccupied habitat patches.

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

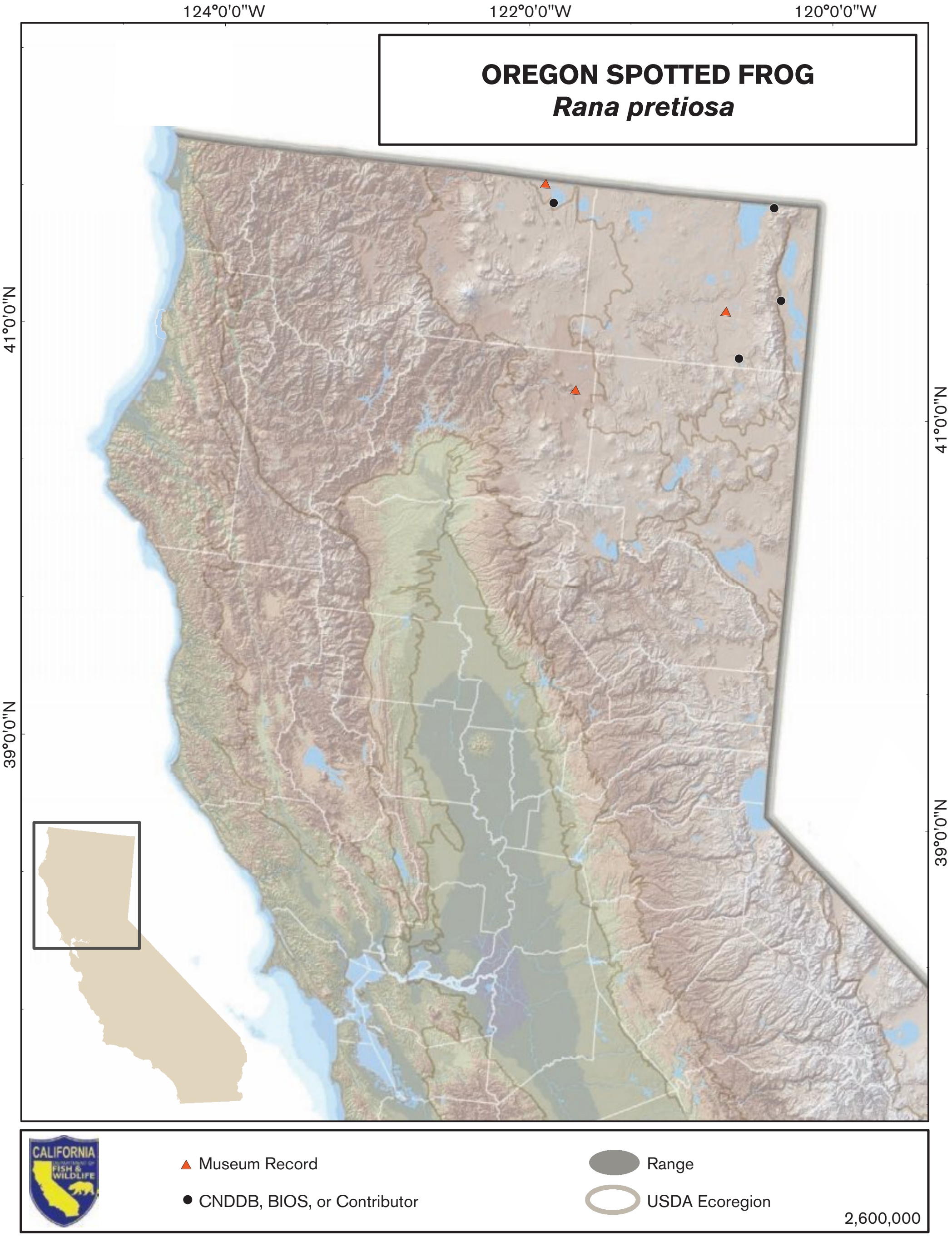


PHOTO ON PREVIOUS PAGE: Oregon spotted frog, Lane County, Oregon. Courtesy of Troy Hibbitts.

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 Plumas 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

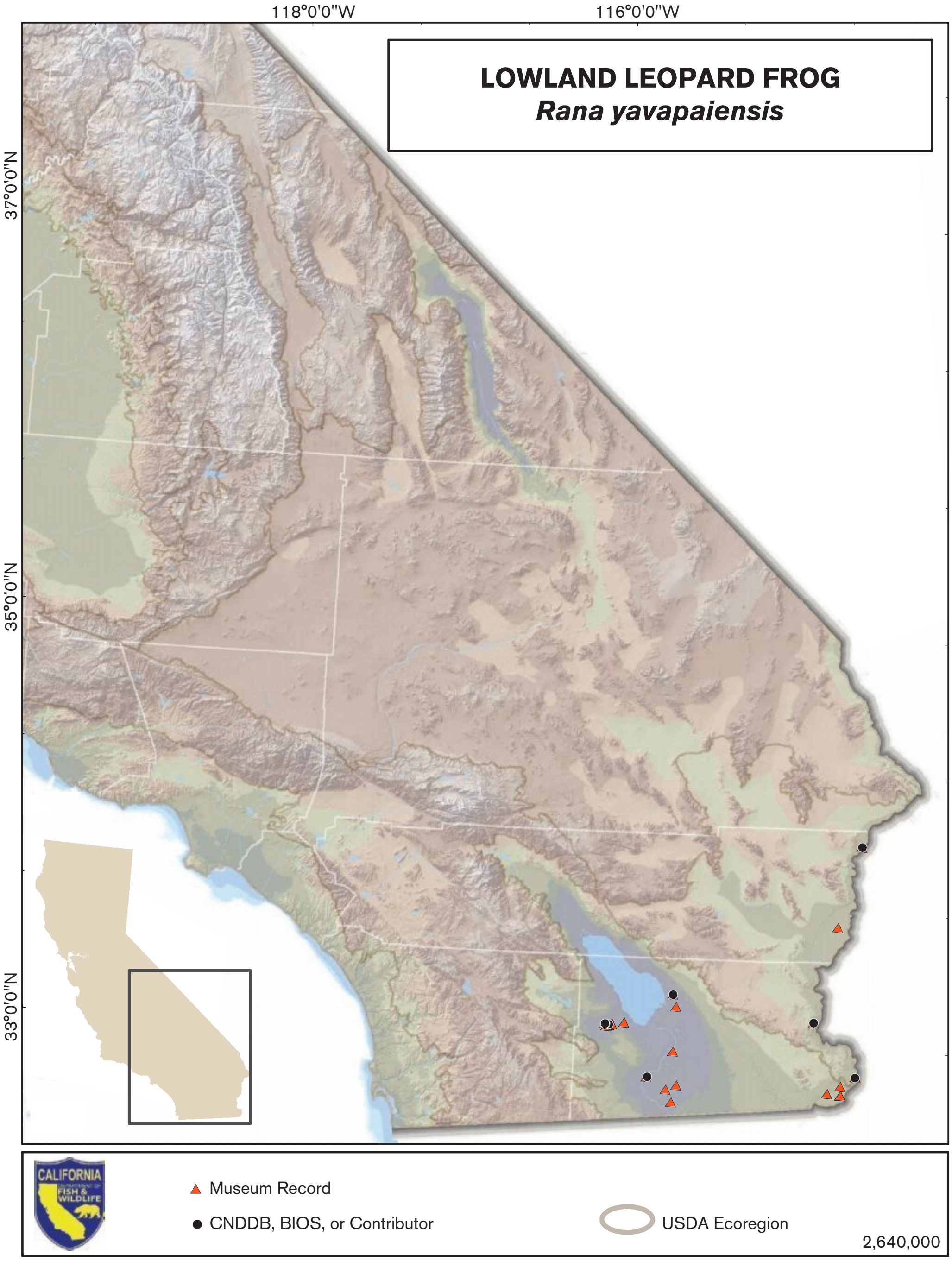


PHOTO ON PREVIOUS PAGE: Lowland leopard frog, Cochise County, Arizona. Courtesy of Brian Freiermuth.

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 be utilized by *Rana yavapaiensis*, and therefore should be surveyed. Because populations are prone to large yearly fluctuations, surveys should be 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

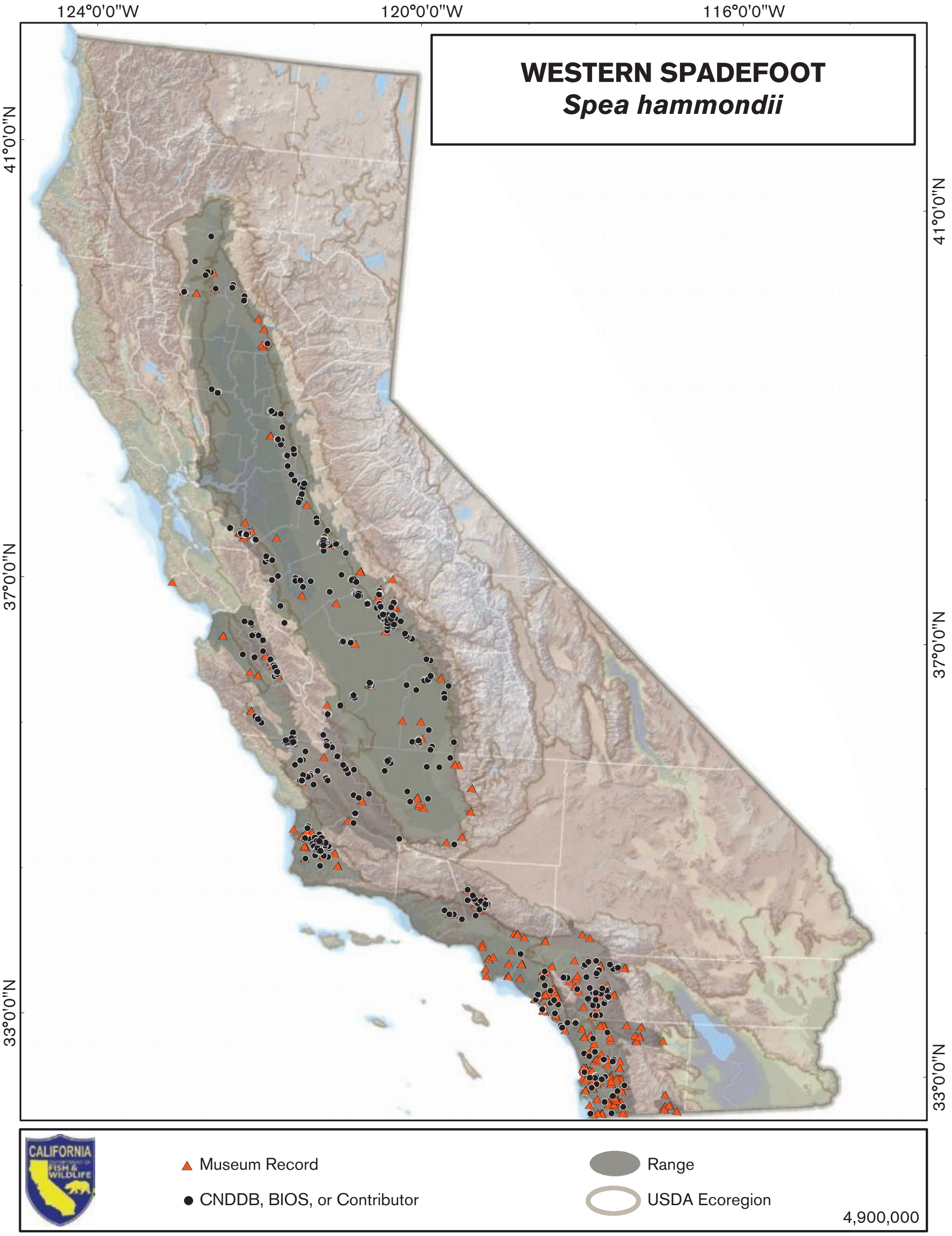


PHOTO ON PREVIOUS PAGE: Western spadefoot, Sacramento County, California. Courtesy of Robert Thomson.

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* (Garcia-Paris 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

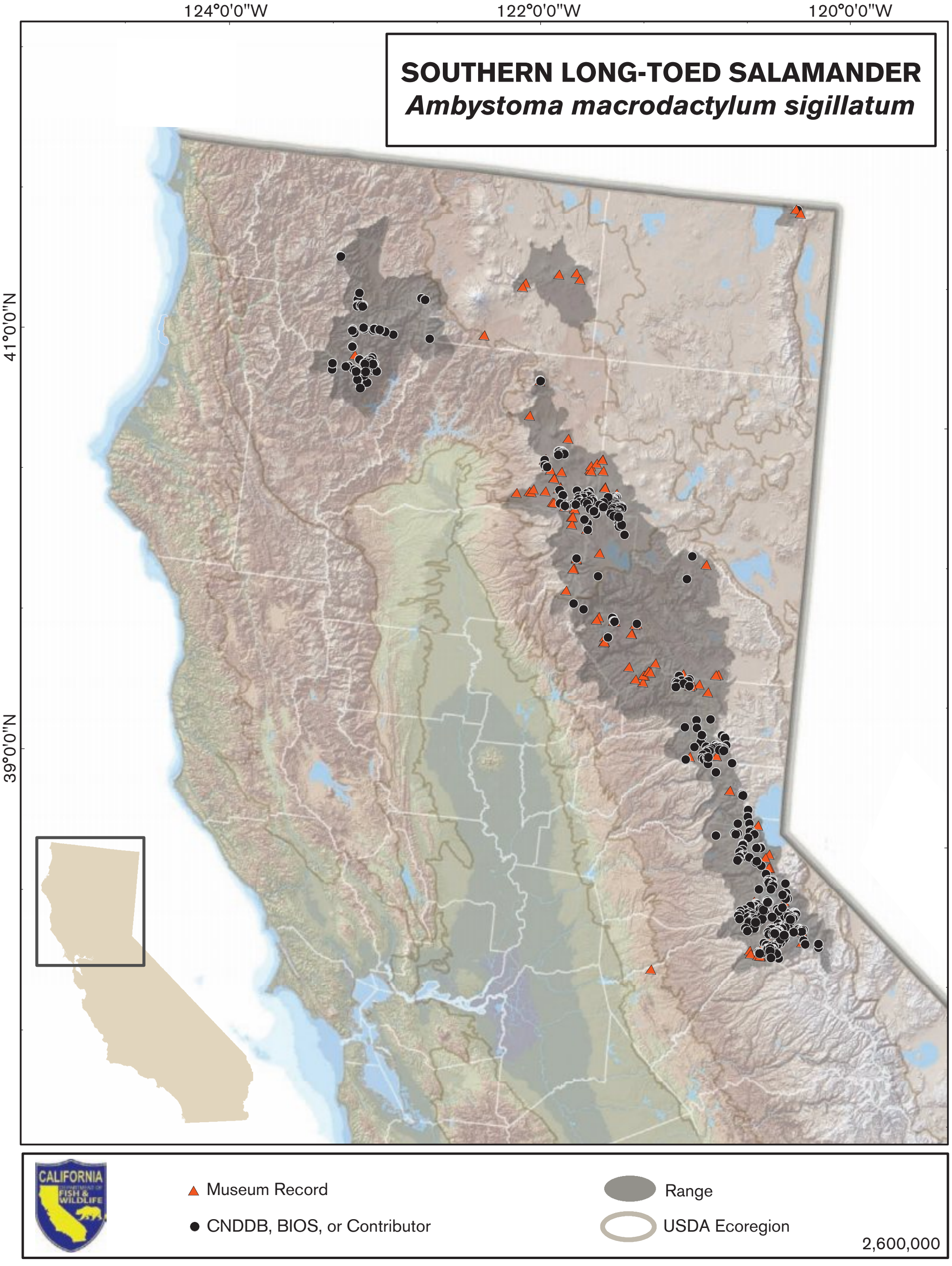


PHOTO ON PREVIOUS PAGE: Southern long-toed salamander, Butte County, California. Courtesy of Robert Hansen.

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

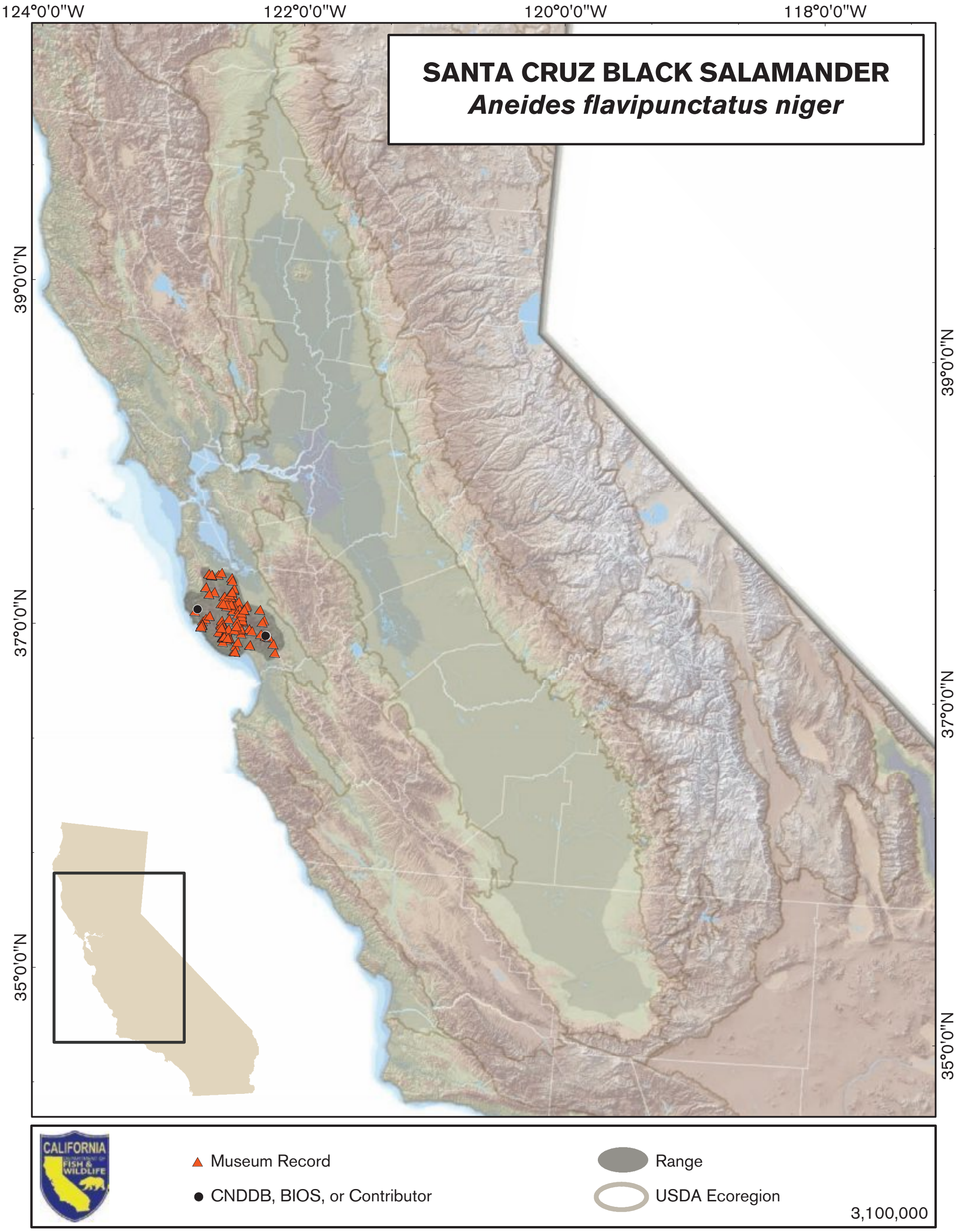


PHOTO ON PREVIOUS PAGE: Santa Cruz black salamander, Santa Cruz County, California. Courtesy of William Flaxington.

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 toe-tips, 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. wright*) 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 existence 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

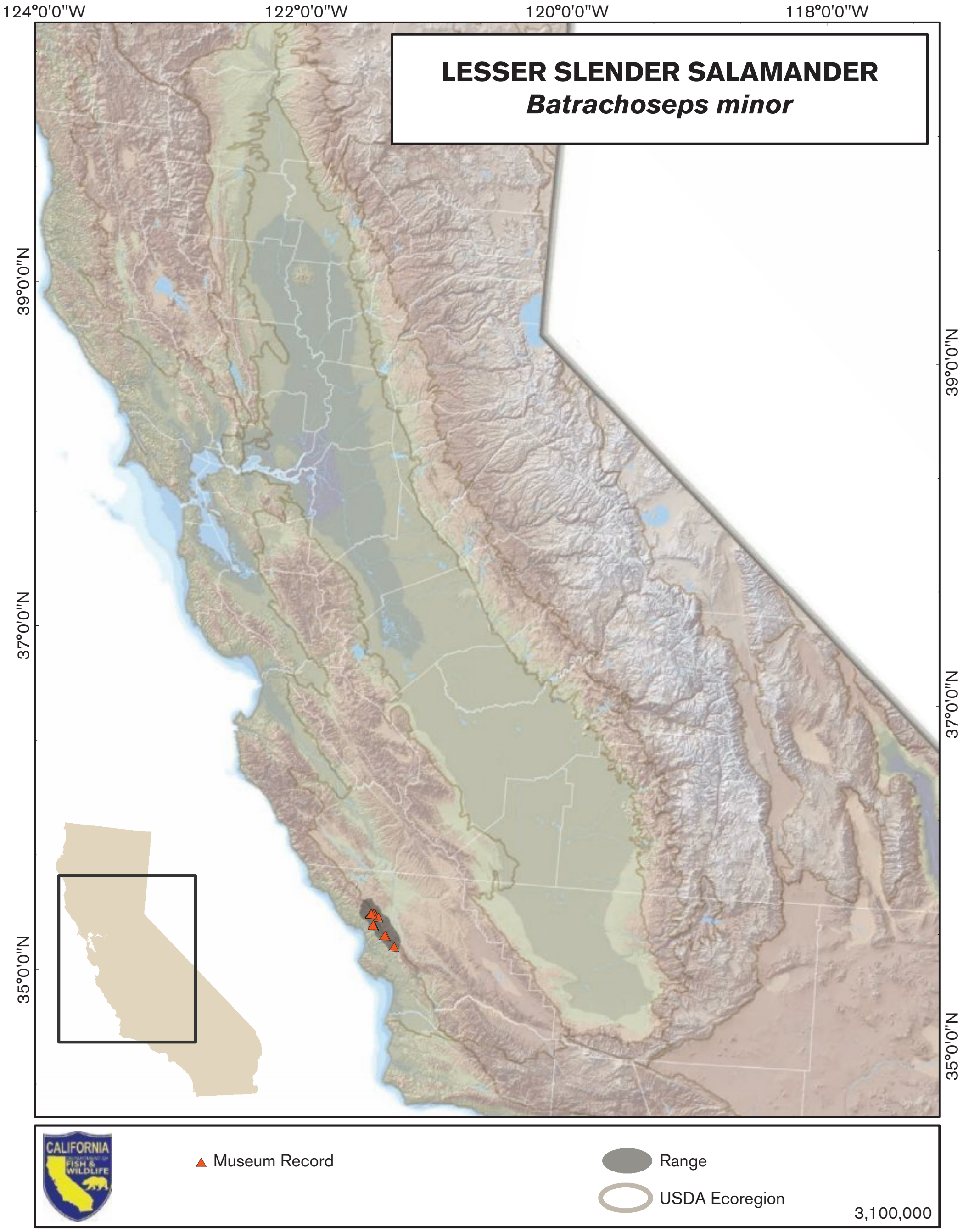


PHOTO ON PREVIOUS PAGE: Lesser slender salamander, San Luis Obispo County, California. Courtesy of William Flaxington.

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

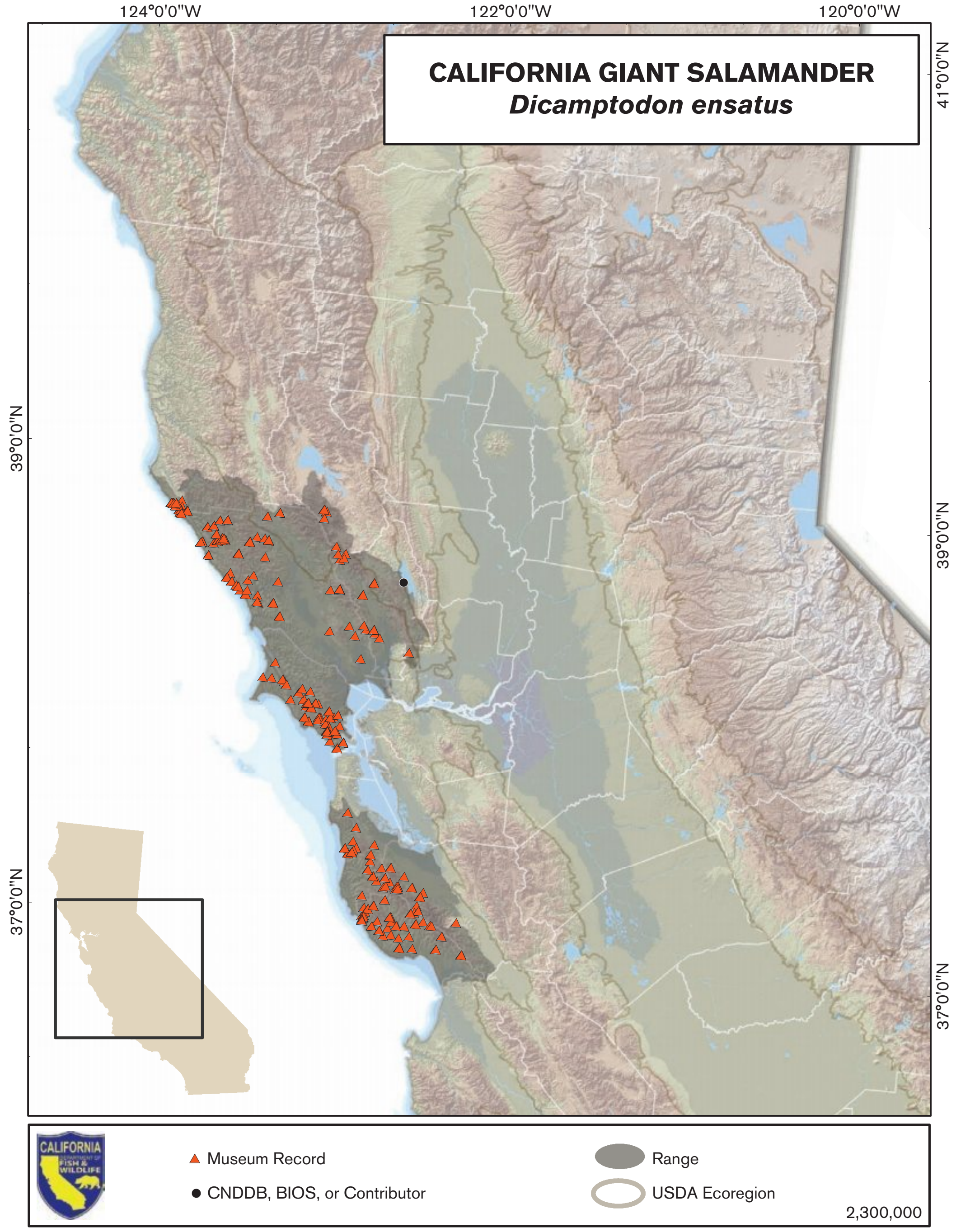


PHOTO ON PREVIOUS PAGE: California giant salamander, Santa Cruz County, California. Courtesy of Nicholas Hess.

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

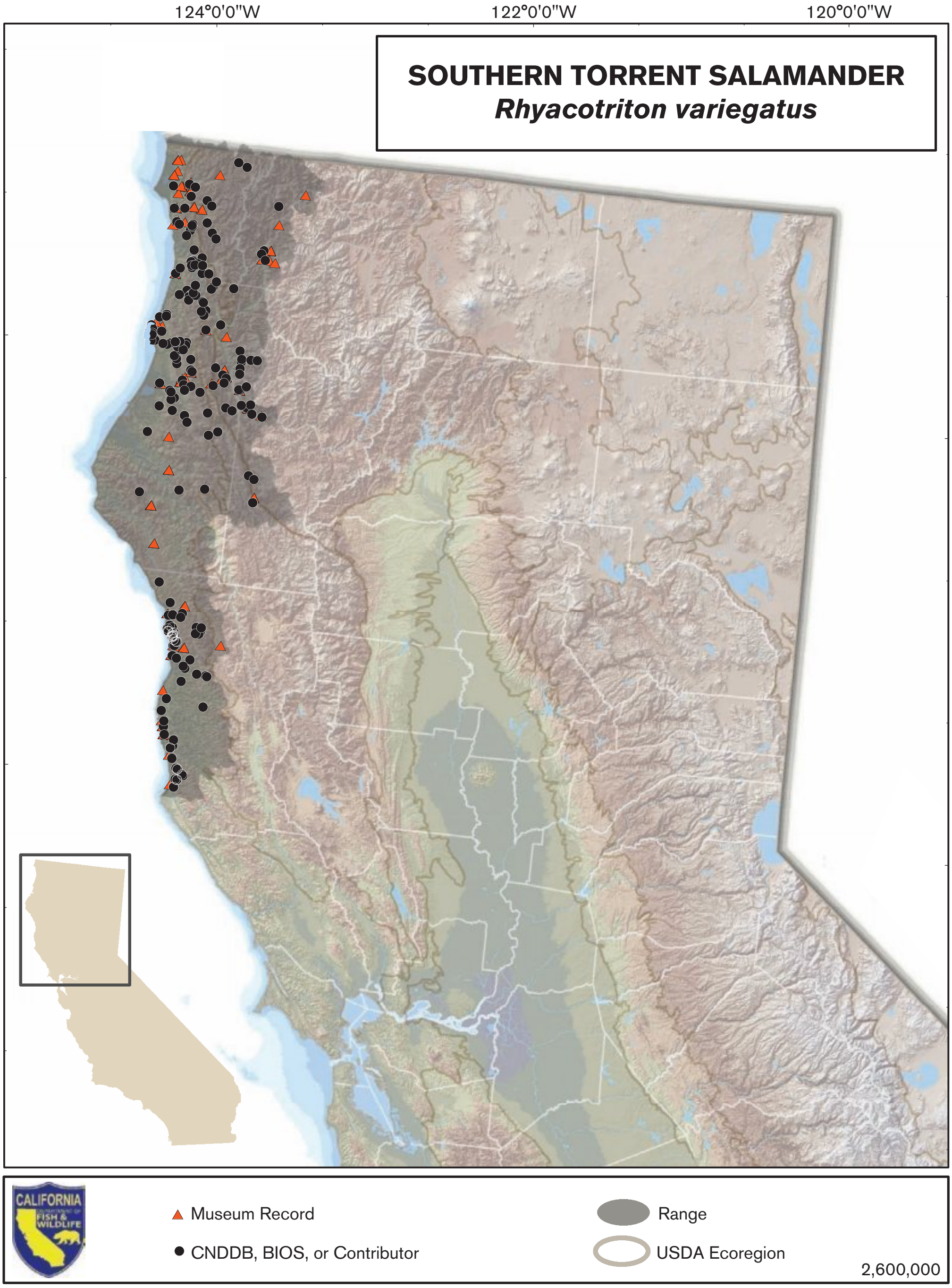


PHOTO ON PREVIOUS PAGE: Southern torrent salamander, Mendocino County, California. Courtesy of Robert Thomson.

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

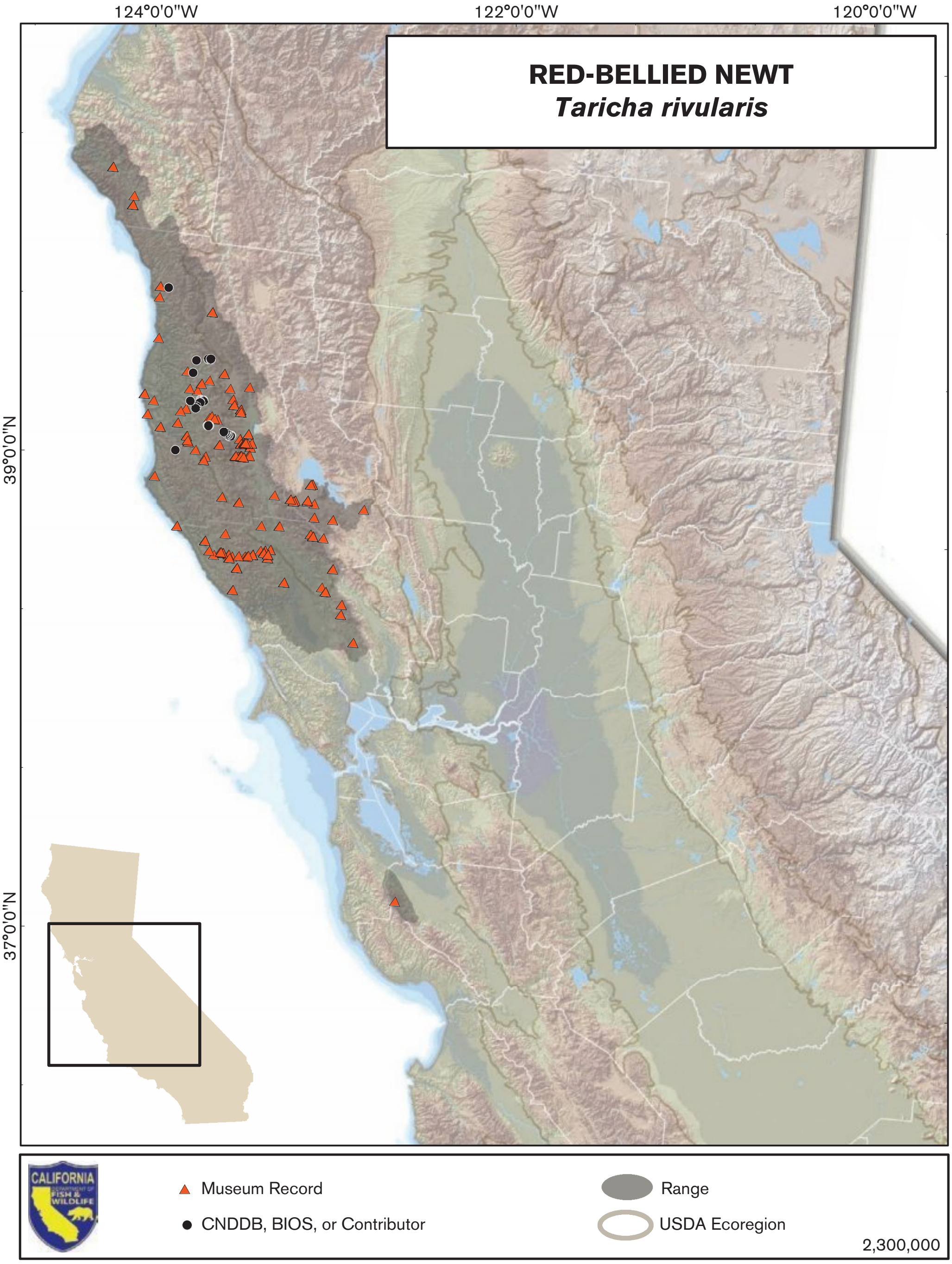


PHOTO ON PREVIOUS PAGE: Red-bellied newt, Mendocino County, California. Courtesy of Adam Clause.

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

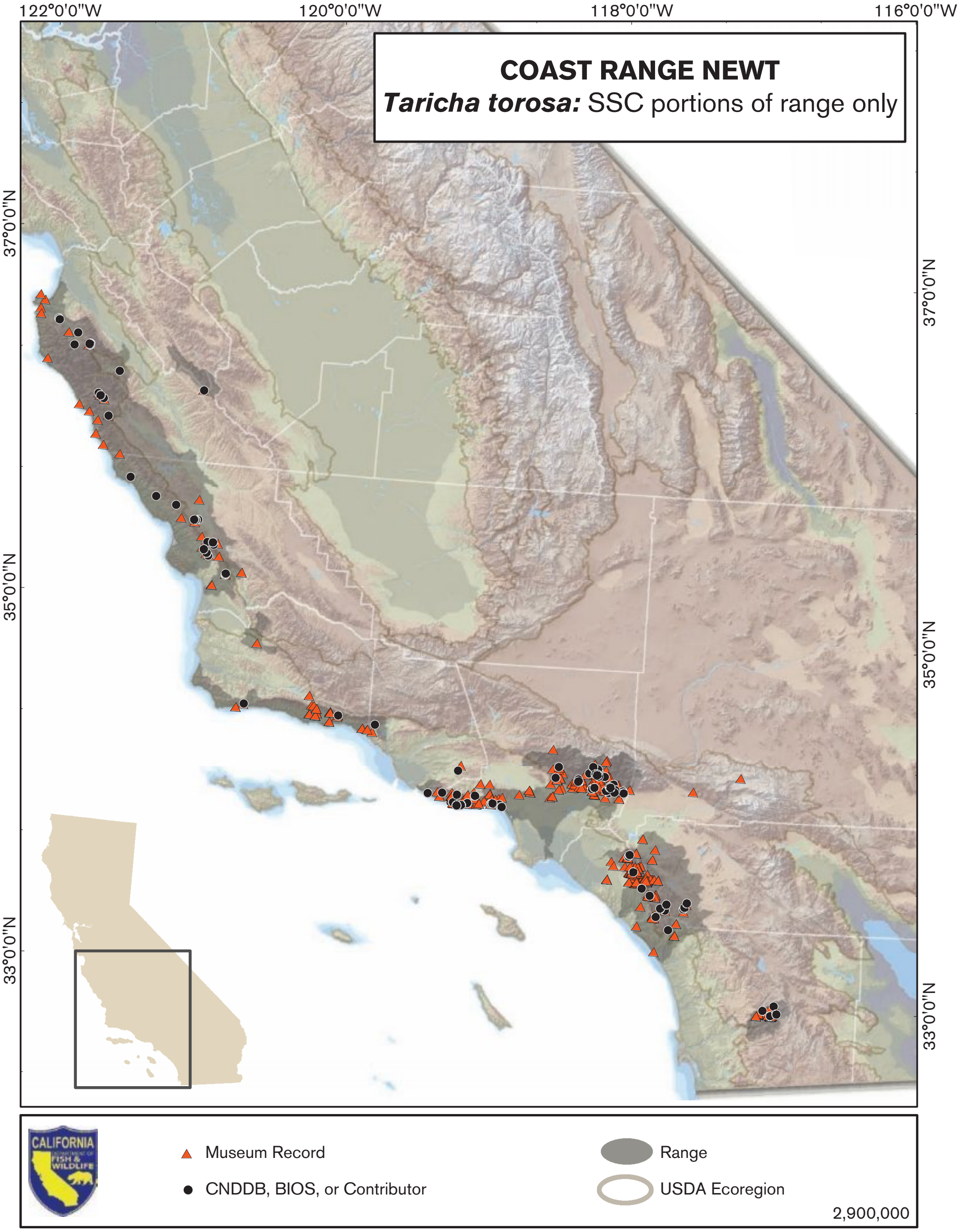


PHOTO ON PREVIOUS PAGE: Coast range newt, southern populations, Los Angeles County, California. Courtesy of Adam Clause.

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

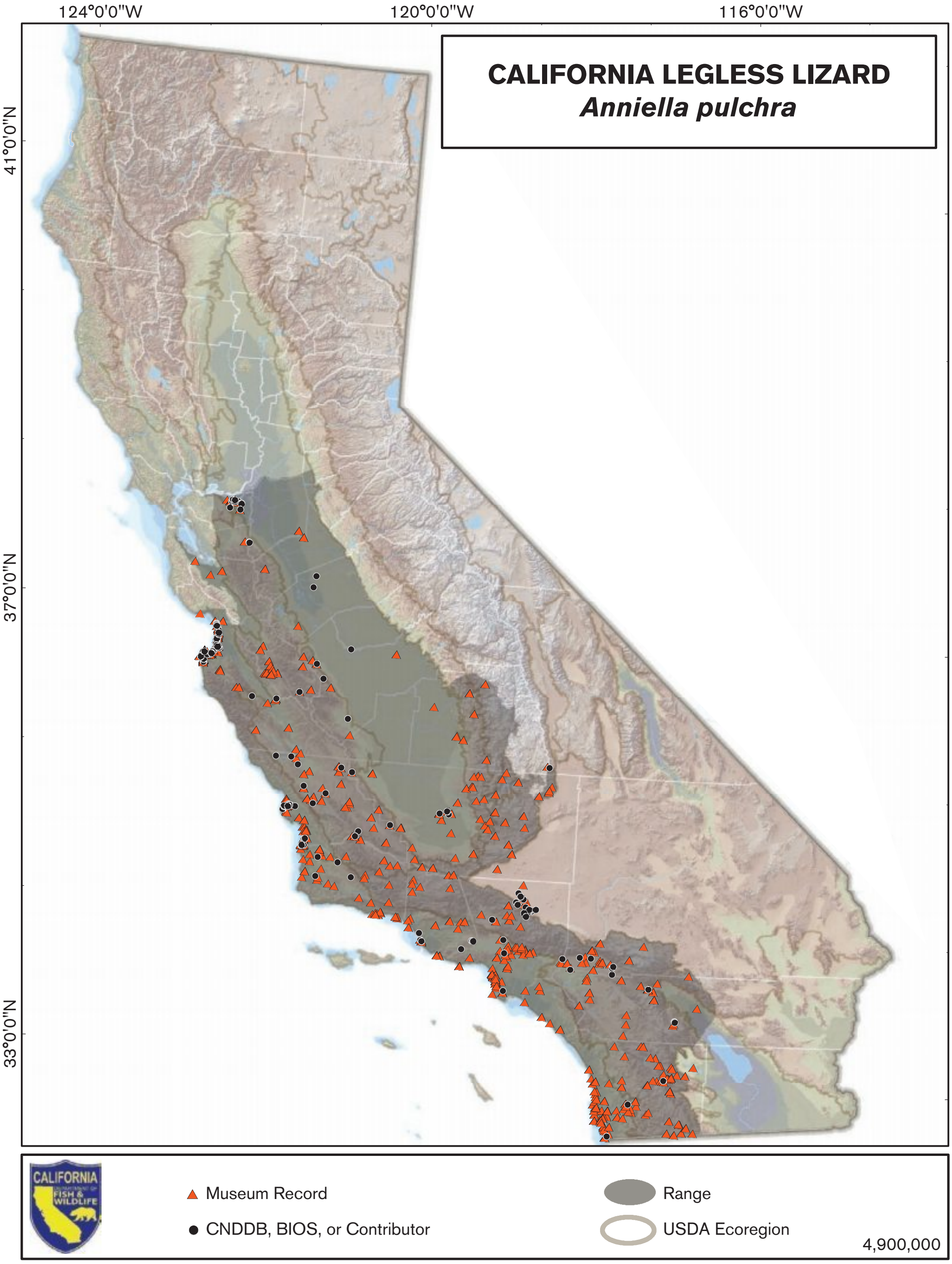


PHOTO ON PREVIOUS PAGE: California legless lizard, Kern County, California. Courtesy of Adam Clause.

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 our 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/m^2 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

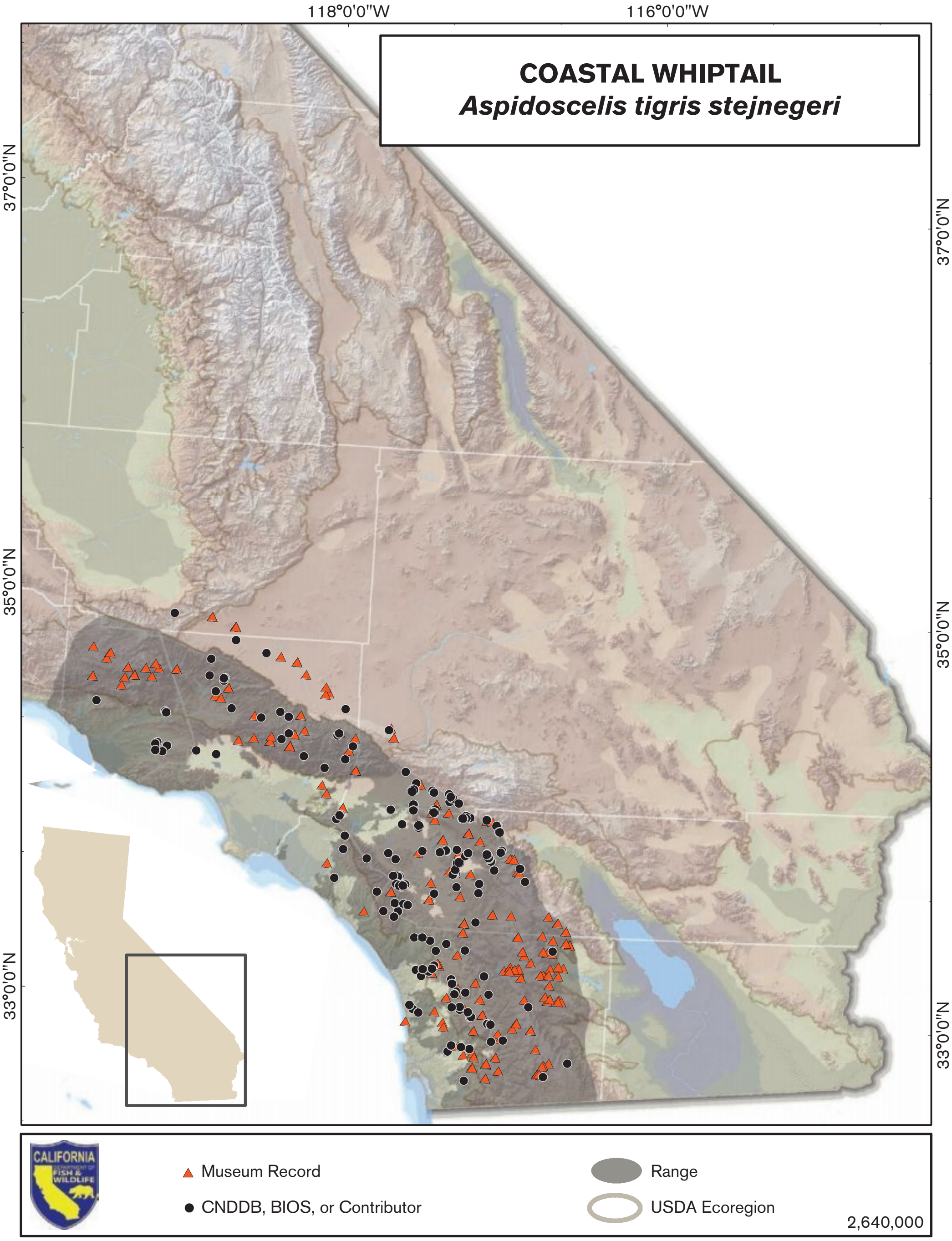


PHOTO ON PREVIOUS PAGE: Coastal whiptail, Los Angeles County, California. Courtesy of Robert Hess.

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

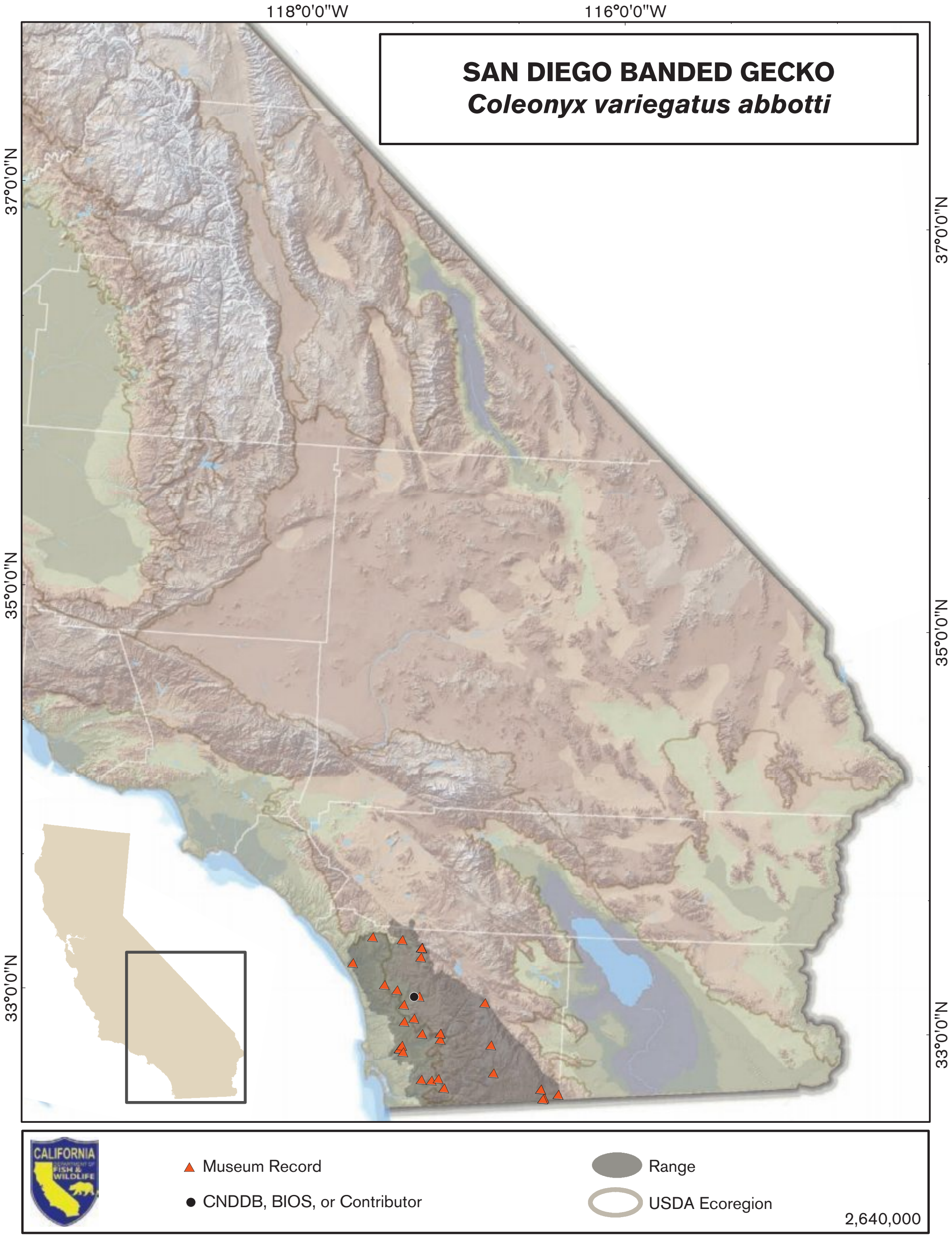


PHOTO ON PREVIOUS PAGE: San Diego banded gecko, San Diego County, California. Courtesy of Jeff Lemm.

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 Peninsular 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

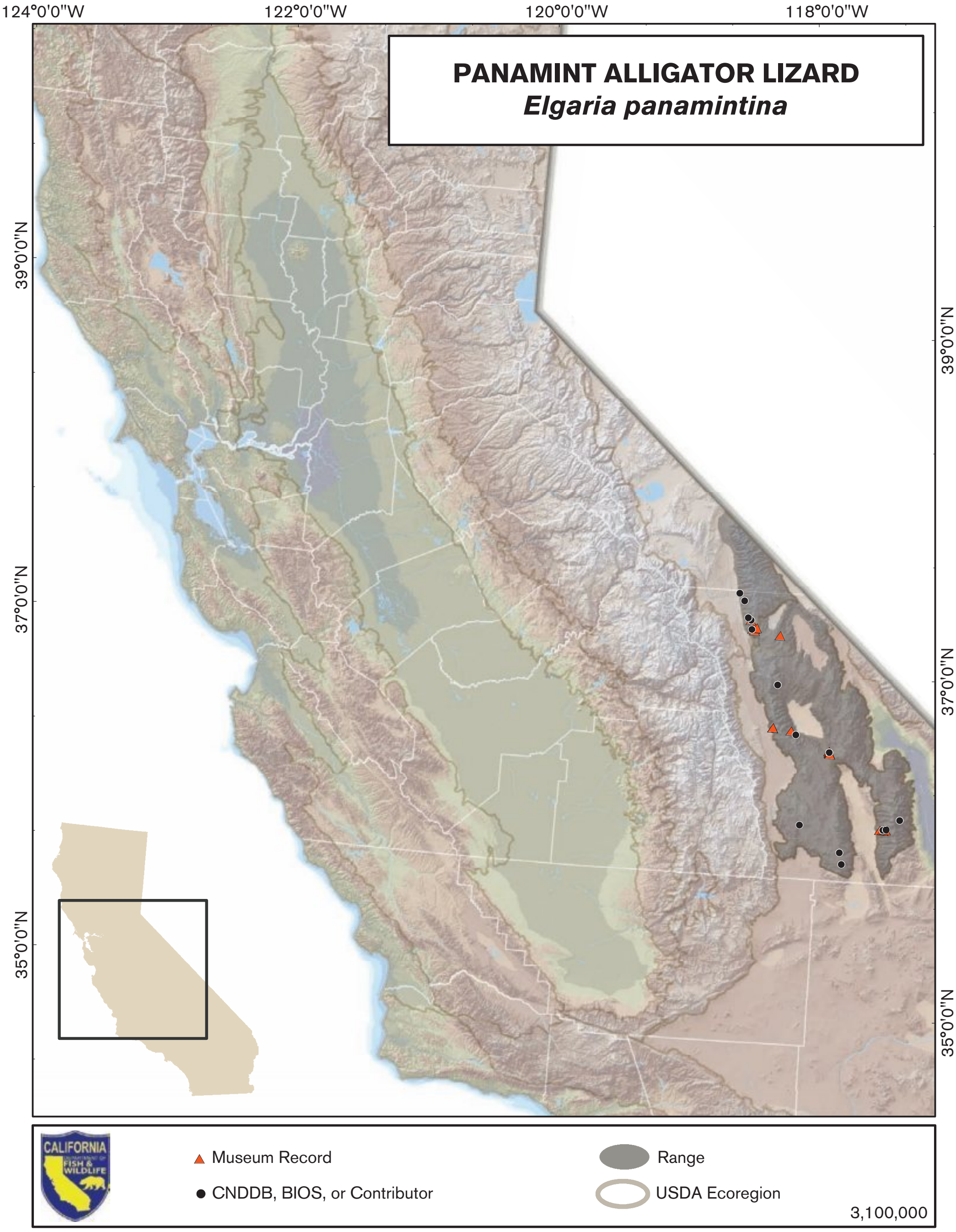


PHOTO ON PREVIOUS PAGE: Panamint alligator lizard, Inyo County, California. Courtesy of Adam Clause.

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

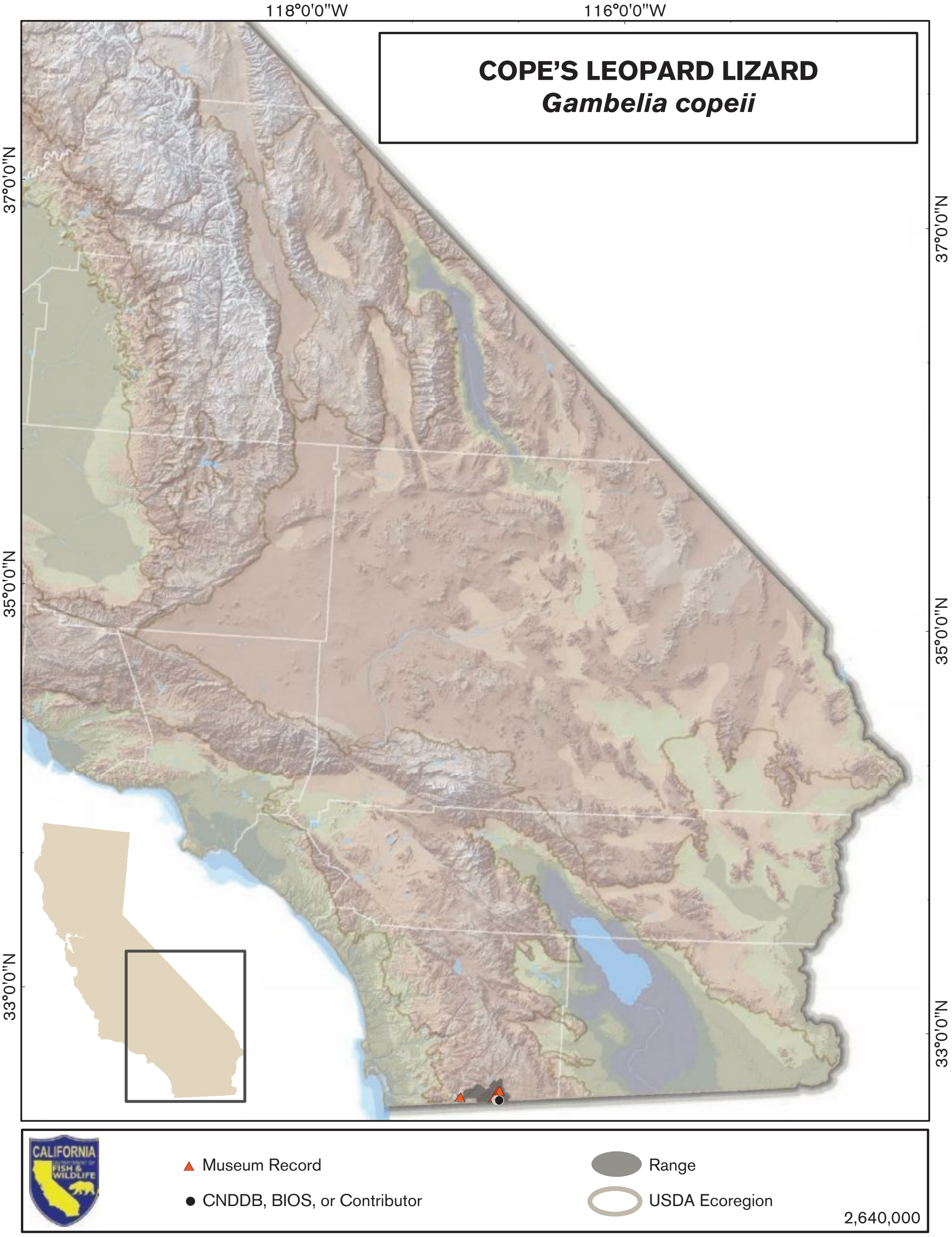


PHOTO ON PREVIOUS PAGE: Cope's leopard lizard, San Diego County, California. Courtesy of Rob Schell Photography.

(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.

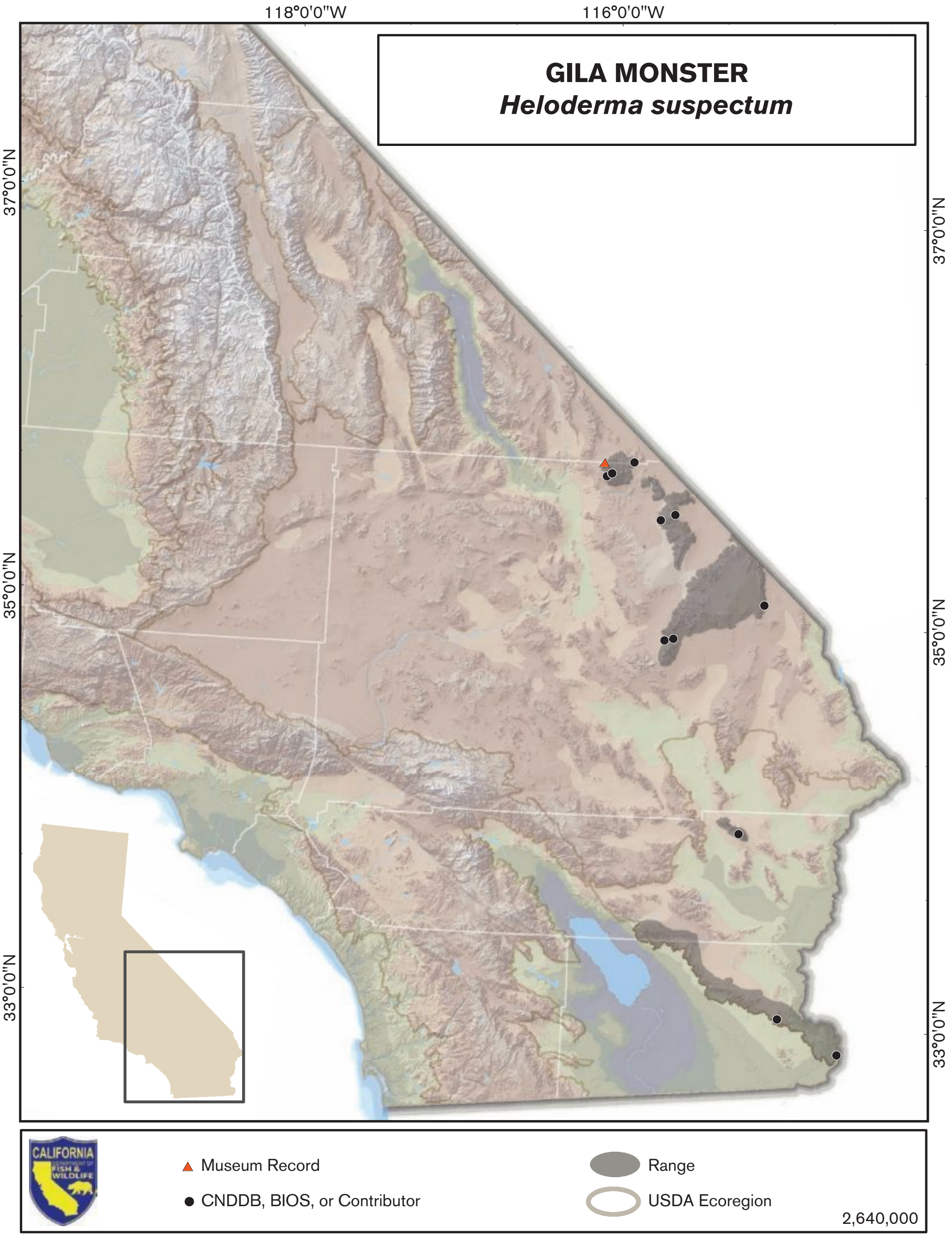
Monitoring, Research, and Survey Needs

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
37°0'0"N
35°0'0"N
33°0'0"N
CALIFORNIA DEPARTMENT OF FISH & WILDLIFE
▲ 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

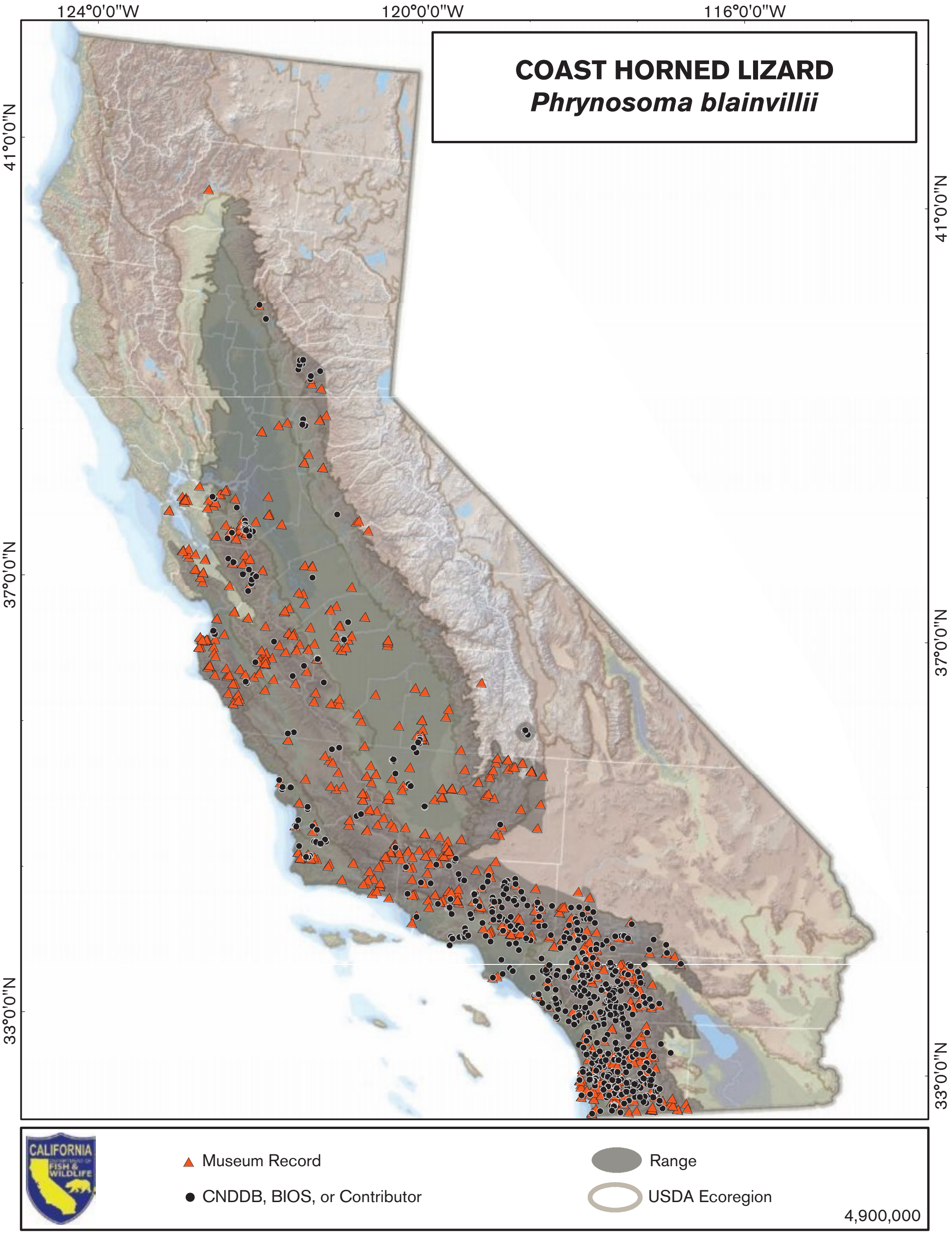


PHOTO ON PREVIOUS PAGE: Coast horned lizard, Kern County, California. Courtesy of Nicholas Hess.

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, HilleRis-Lambers 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

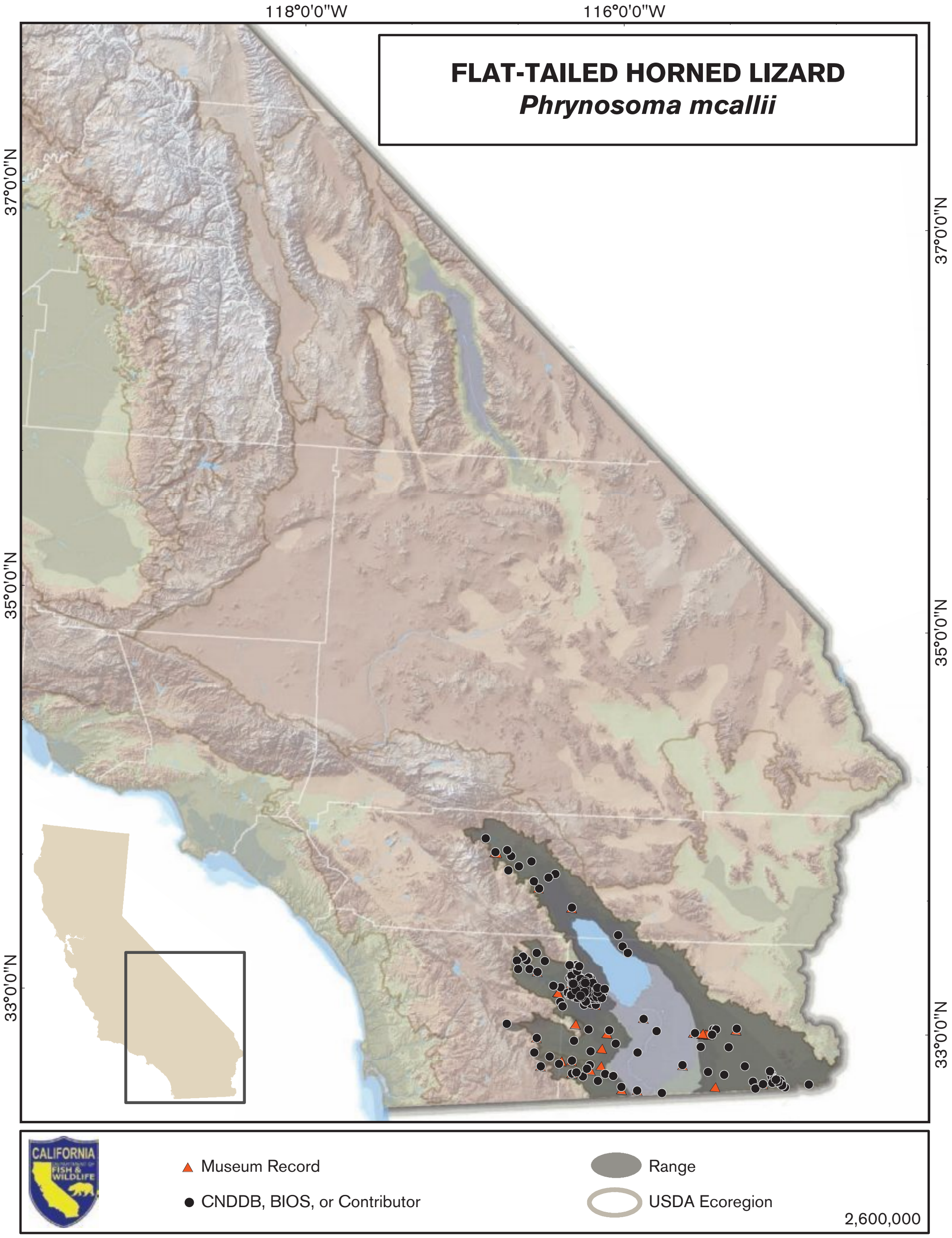


PHOTO ON PREVIOUS PAGE: Flat-tailed horned lizard, Sonora, Mexico. Courtesy of Rob Lovich.

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 rangewide 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

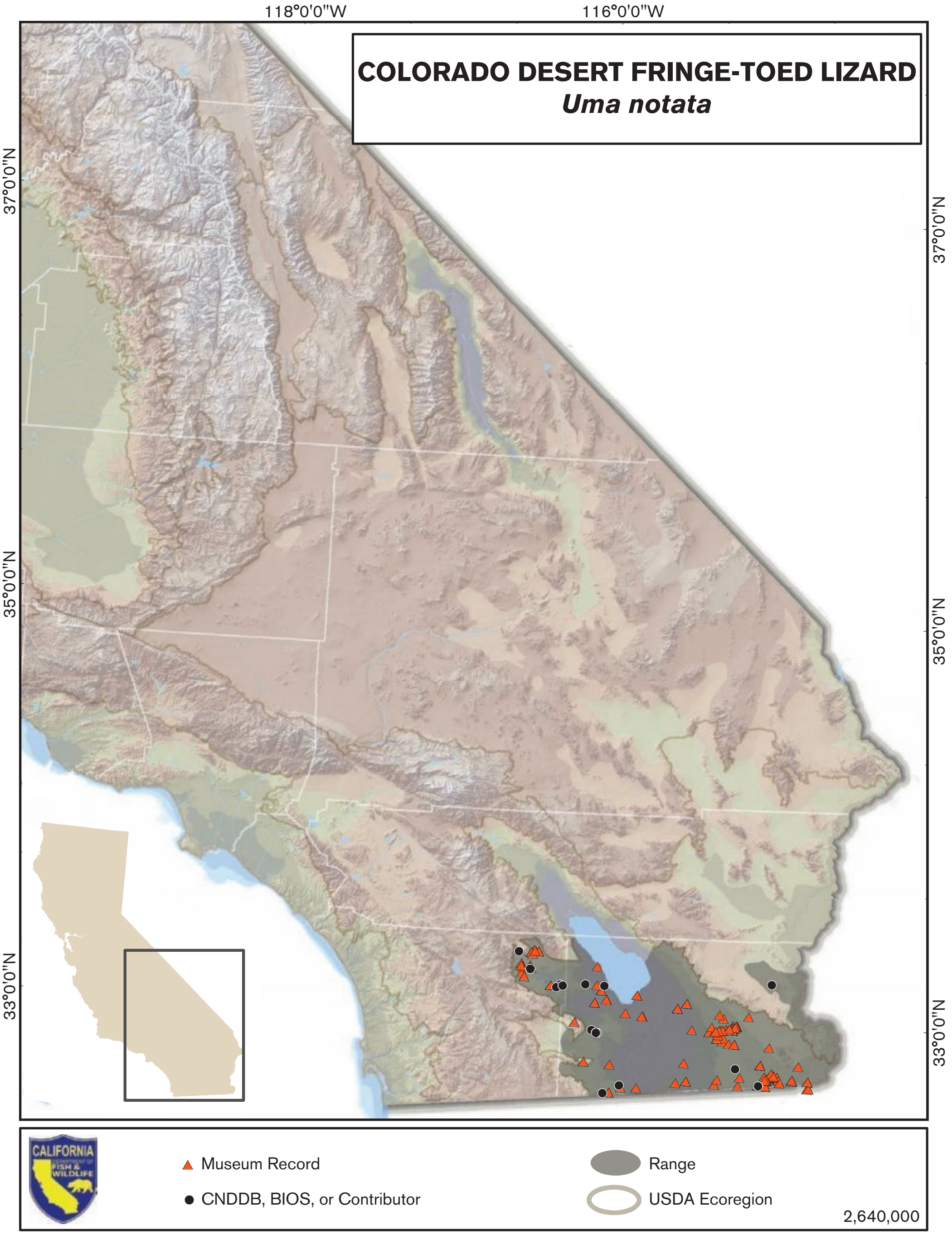


PHOTO ON PREVIOUS PAGE: Colorado Desert fringe-toed lizard, Imperial County, California. Courtesy of Adam Clause.

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

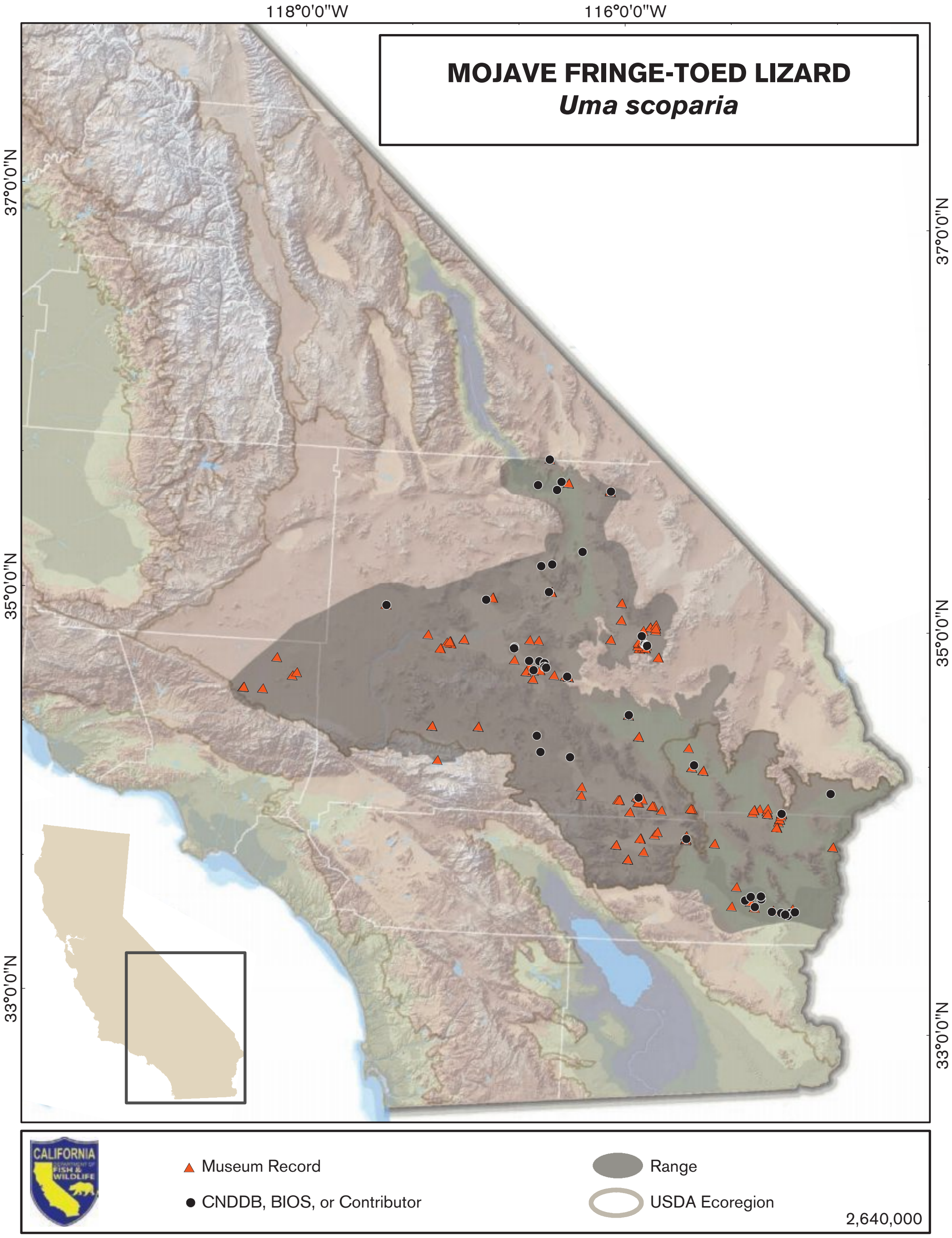


PHOTO ON PREVIOUS PAGE: Mojave fringe-toed lizard, San Bernardino County, California. Courtesy of Luke Mahler.

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 windblown 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 multi-locus 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

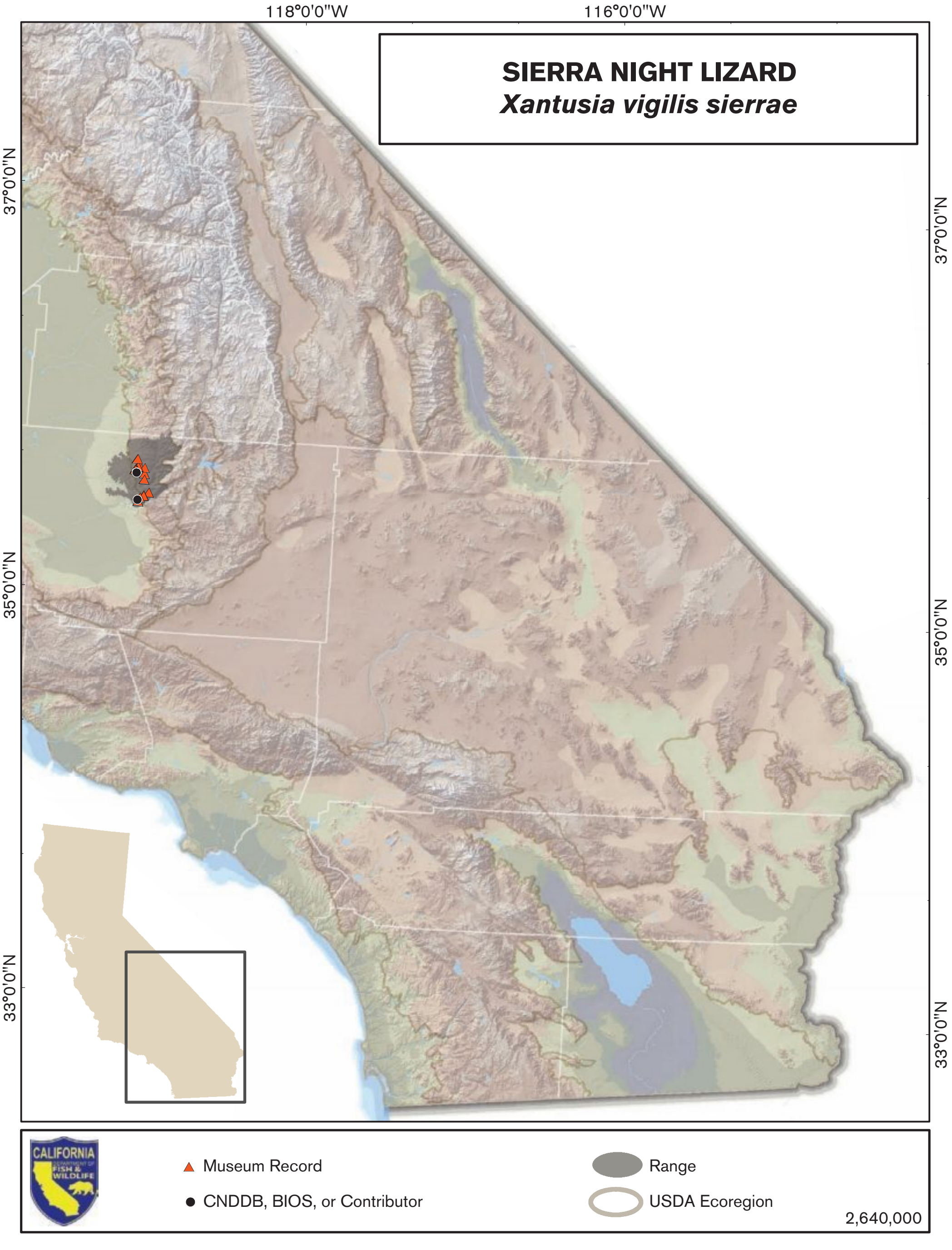


PHOTO ON PREVIOUS PAGE: Sierra night lizard, Kern County, California. Courtesy of Jackson Shedd.

(*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

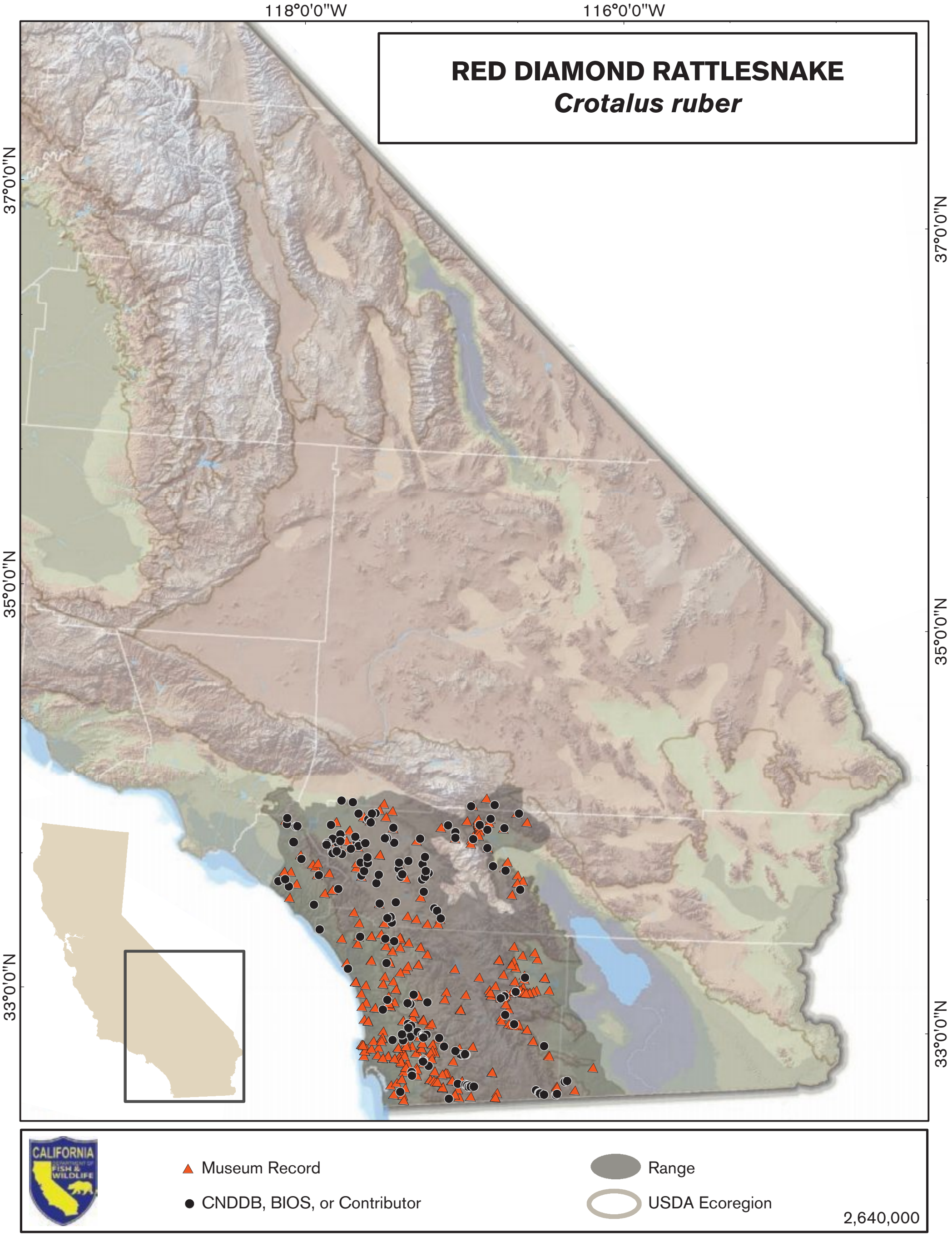


PHOTO ON PREVIOUS PAGE: Red diamond rattlesnake, San Diego County, California. Courtesy of Jeff Lemm.

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

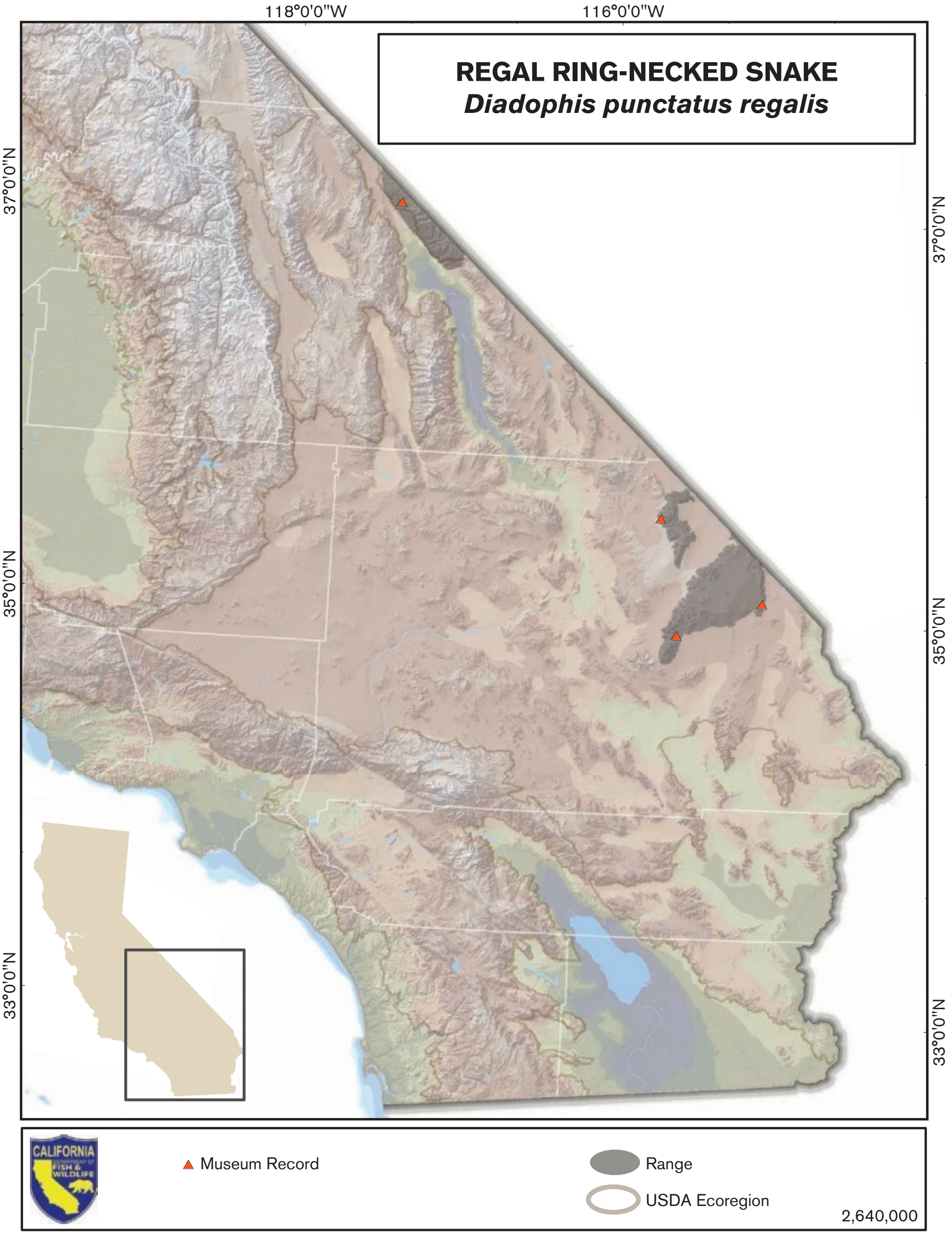


PHOTO ON PREVIOUS PAGE: Regal ring-necked snake, Santa Cruz County, Arizona. Courtesy of Jackson Shedd.

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

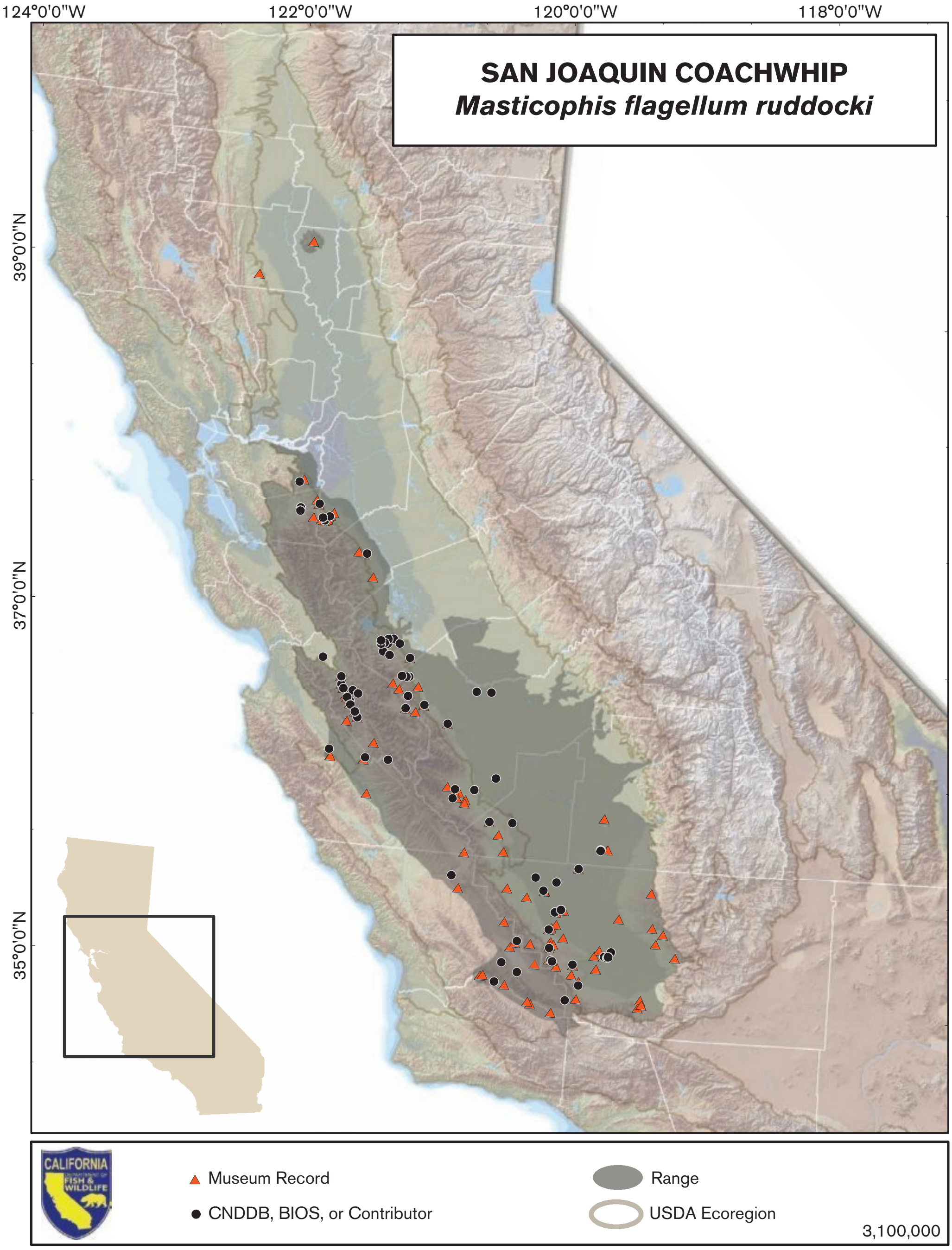


PHOTO ON PREVIOUS PAGE: San Joaquin coachwhip, Kern County, California. Courtesy of Jeff Lemm.

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

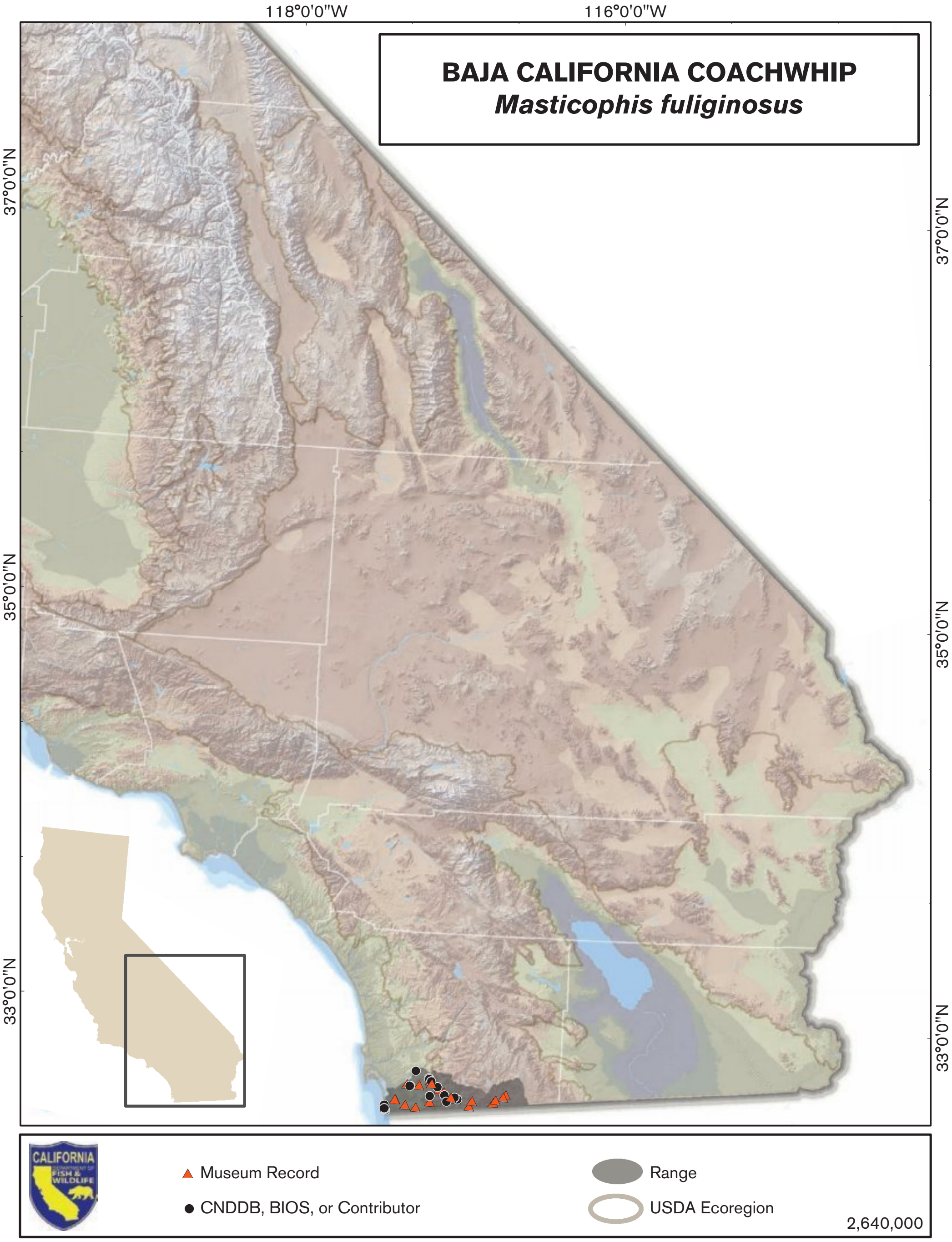


PHOTO ON PREVIOUS PAGE: Baja California coachwhip, Baja California, Mexico. Courtesy of Brad Shaffer.

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

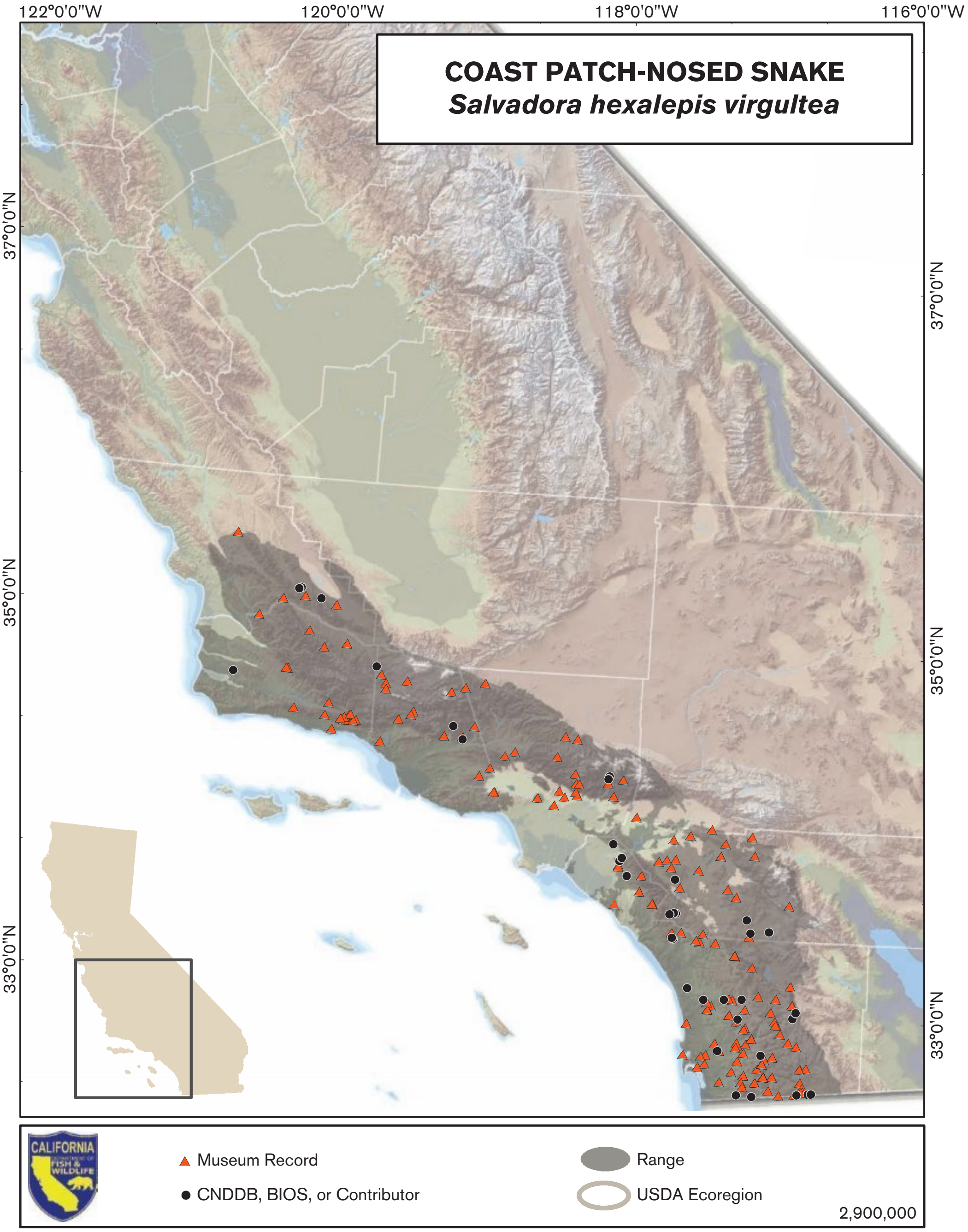


PHOTO ON PREVIOUS PAGE: Coast patch-nosed snake, San Diego County, California. Courtesy of John Andermann.

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

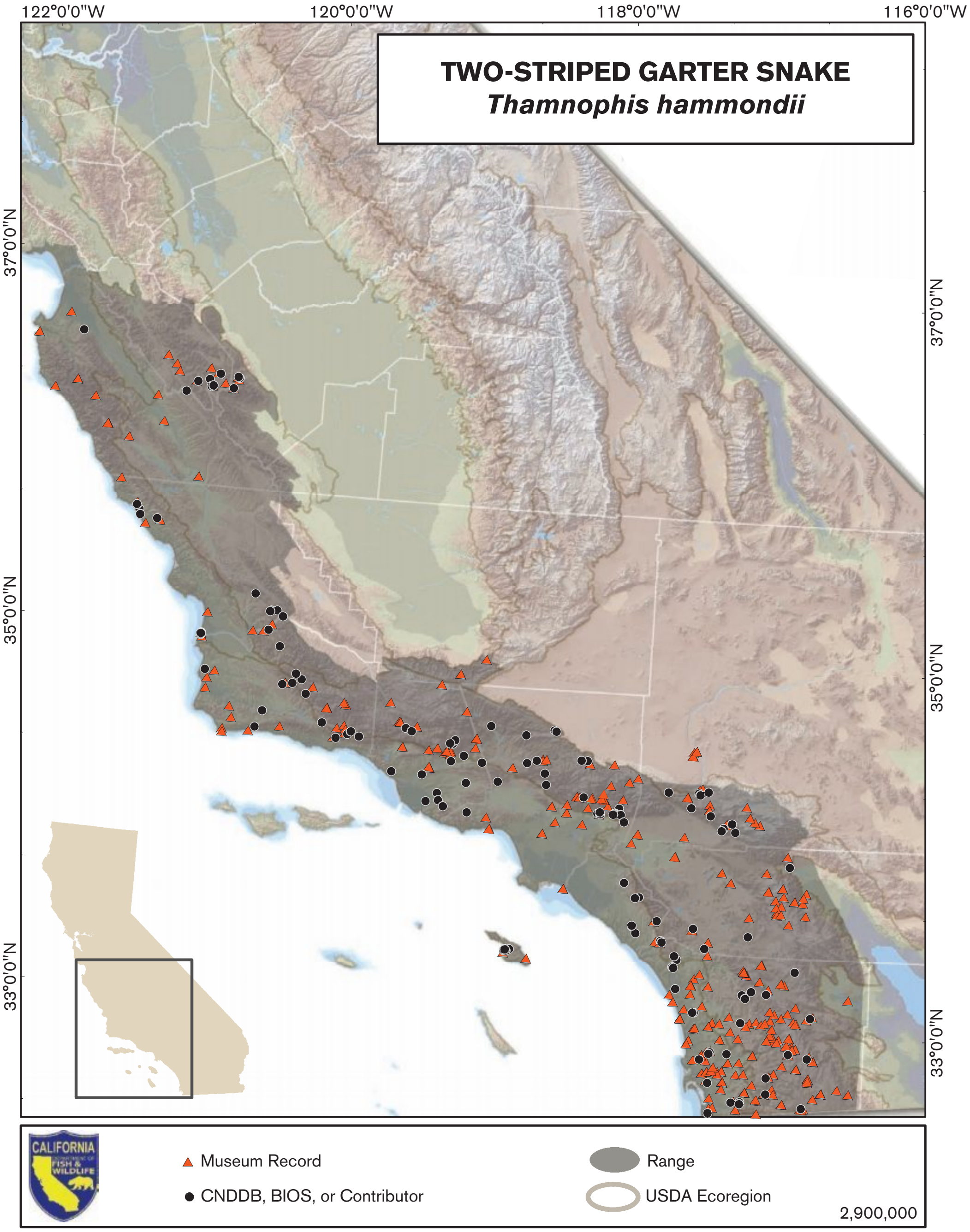


PHOTO ON PREVIOUS PAGE: Two-striped garter snake, Los Angeles County, California. Courtesy of Robert Hess.

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 m^2 (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 m^2 (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

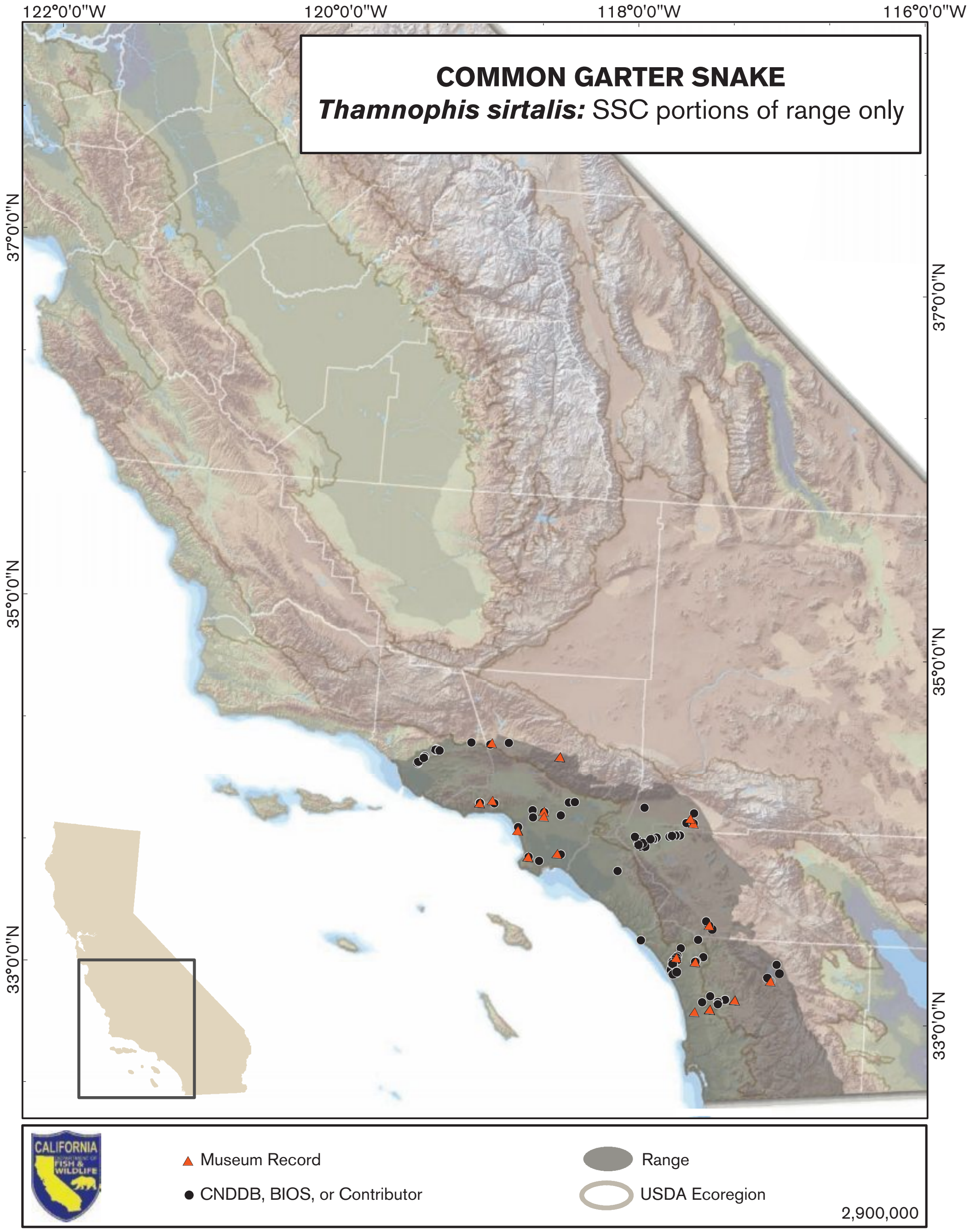


PHOTO ON PREVIOUS PAGE: Common garter snake, Orange County, California. Courtesy of Jeff Lemm.

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.

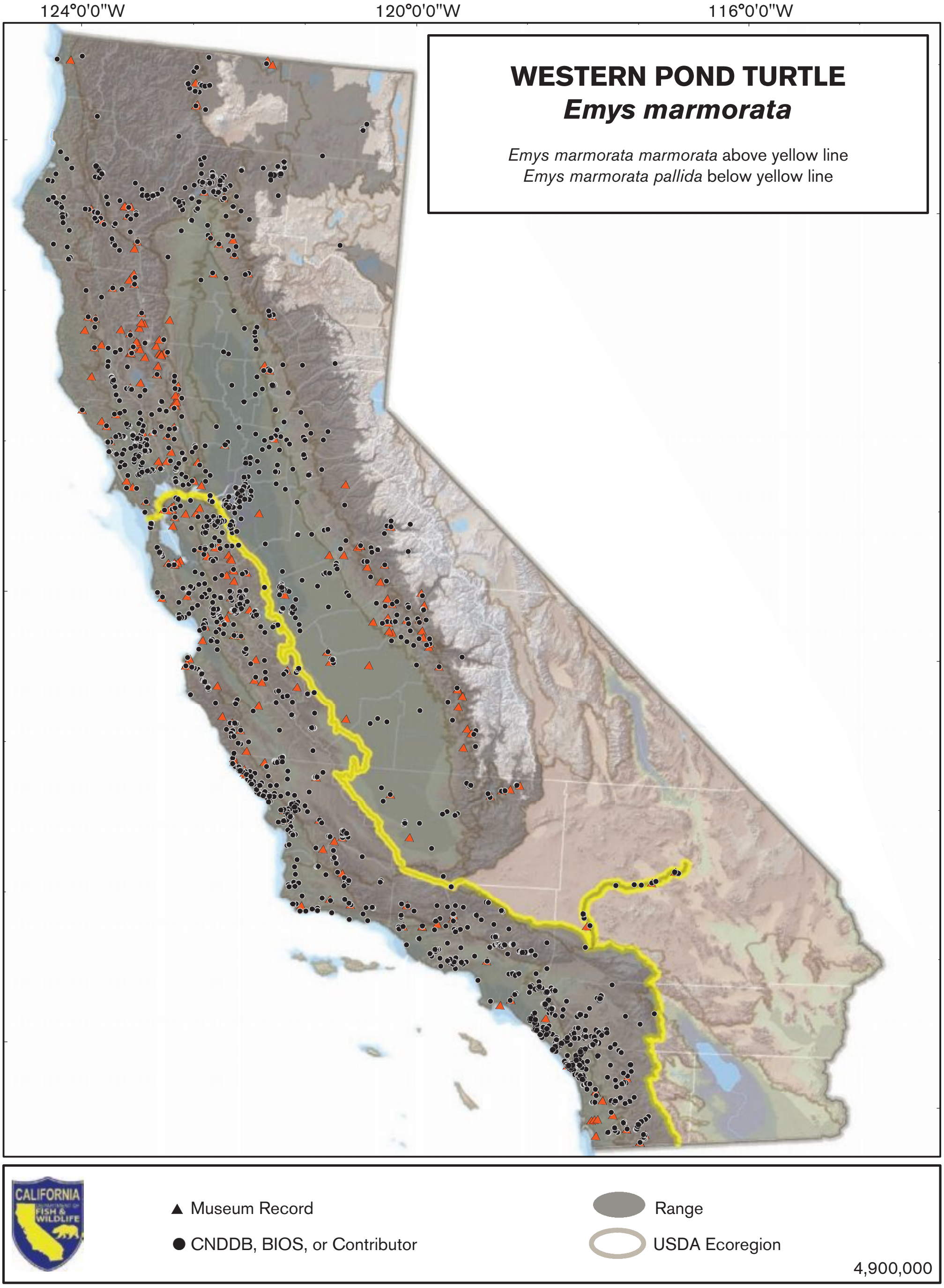
Monitoring, Research, and Survey Needs

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.



WESTERN POND TURTLE
Emys marmorata

Emys marmorata marmorata above yellow line
Emys marmorata pallida below yellow line

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 non-native 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, and 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[13]	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[14]	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[17]	Sierra night lizard	X		SSC	LC		
Xantusia vigilis vigilis[17]	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[18]	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 nuchulata	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[27]	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 Rissler 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. blainvilli*.

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.

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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