



MEMORANDUM

DATE: August 27, 2025

TO: Planning Director

FROM: Jon Mijat, Planner
RMA Planning Division

SUBJECT: Errata Memo – Revised Biologist Letter for PL23-0147

On August 20, 2025, a Planning Director hearing was held for PL23-0147 to consider Maxwell Barn that proposes construction of a 1,296 square-foot footprint (864sqft ground floor + 432sqft loft) wood framed barn without plumbing or electrical. Included in the hearing packet was a Biologist letter, from a County Qualified Biologist, that had analyzed the impacts of the project to Sisar Creek. Public Comments were submitted at the hearing and PL23-0147 was continued to September 4, 2025, to allow staff and Biologist to review comments.

Subsequently, the Biologist provided an updated letter and studies to show the CEQA exemptions used are still appropriate.

This errata memo is meant to document the updated Biologist letter that bolsters its statement of no impacts and no cumulative impacts since there is evidence of the extirpation of Southern California Steelhead and Foothill Yellow-Legged Frog within this area. This combined with the proposed barn being further from the existing pens and behind existing structures, leads to the conclusion that there are no impacts or cumulative impacts to Sisar Creek. This information comes from the sources listed below.

Attachments:

- a. Updated Biologist Letter from Matt Ingamells dated August 26, 2025
- b. Petition to List Foothill Yellow-Legged Frog (*Rana boylei*) As Threatened Under the California Endangered Species Act December 14, 2016
- c. Report to the Fish and Game Commission California Endangered Species Act Status Review of Southern California Steelhead (*Oncorhynchus mykiss*) January 2024

August 26, 2025
Project no. 2402-1661

Caitlin Maxwell
Robert William Company LLC
216 N 8th Street/ PO Box 1025
Santa Paula, CA 93061

**Revised Biologist's Letter for a Proposed Barn
at 15315 Ojai-Santa Paula Road (PL23-0147)**

Background

This letter has been revised in response to comments from Ventura County Planning provided in an email to myself from Aubrie Richardson on August 20, 2025.

The property owner plans to construct a new barn on the property primarily to house young goats. The proposed barn site is located near Sisar Creek which supports wetlands as defined in the glossary of the Ventura County 2040 General Plan.

Policy COS-1.10 of the General Plan requires an evaluation of the potential effects on wetlands and wetland habitats of discretionary development proposed to be located within 300 feet of wetlands pursuant to the County's Initial Study Assessment Guidelines (ISAG)

Policy COS-1.11 of the General Plan requires discretionary development to be sited 100 feet from wetland habitats, except as provided below. The 100-foot setback may be increased or decreased based upon an evaluation and recommendation by a qualified biologist and approval by the decision-making body based on factors that include, but may not be limited to, soil type, slope stability, drainage patterns, the potential for discharges that may impair water quality, presence or absence of endangered, threatened or rare plants or animals, direct and indirect effects to wildlife movement, and compatibility of the proposed development with use of the wetland habitat area by wildlife. Discretionary development that would have a significant impact on a wetland habitat shall be prohibited unless mitigation measures are approved that would reduce the impact to a less than significant level.

Policy COS-1.10 references the ISAG, which lists the following activities or project features that may result in a potentially significant Impacts to wetlands:

- Removal of vegetation.
- Grading.
- Obstruction or diversion of water flow.
- Change in velocity, siltation, volume of flow, or runoff rate.
- Placement of fill.

Attachment A: Updated
Biologist Letter from Matt
Ingamells;
dated August 26, 2025

- Placement of structures.
- Construction of a road crossing.
- Placement of culverts or other underground piping.
- Any disturbance of the substratum.
- Disruptions to wetland or riparian plant communities that would isolate or substantially interrupt contiguous habitats, block seed dispersal routes, or increase vulnerability of wetland species to exotic weed invasion or local extirpation.
- Interference with ongoing maintenance of hydrological conditions in a water or wetland.
- Not provide an adequate buffer for protecting the functions and values of existing waters or wetlands. Factors to be used in determining adjustment of the minimum 100-foot buffer include soil type, slope stability, drainage, patterns, presence or absence of endangered, threatened or rare plants or animals, and compatibility of the proposed development with the wildlife use of the wetland habitat area.

Wetland Definition

The glossary of the Ventura County 2040 General Plan defines wetlands as:

Lands that are transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is periodically covered with shallow water. The frequency of occurrence of water is sufficient to support a prevalence of vegetative or aquatic life that requires saturated or seasonally saturated soil conditions for growth and reproduction. Wetlands include marshes, bogs, sloughs, vernal pools, wet meadows, river and stream overflows, mudflats, ponds, springs, and seeps.

Wetland Delineation

On April 23, 2024 a County-approved biologist (myself) conducted a wetland delineation of the northern portion of Sisar Creek (adjacent to the property) within a minimum of 200 feet of the proposed barn site. Sisar Creek was flowing at the time of the wetland delineation. The delineation was based on identifying areas with a shallow water table (saturated or seasonally saturated soils along the creek bank) and vegetation requiring saturated or seasonally saturated soil conditions. The presence of wetland plants (facultative, facultative wetland and obligate wetland plants) listed in the Corps of Engineers' 2022 National Wetland Plant List (Arid West Region) was used to identify the wetland boundary beyond saturated or seasonally saturated soils along the creek bank. Wetland plants found along Sisar Creek adjacent to the property included western sycamore (facultative plant) and arroyo willow (facultative wetland plant). The wetland delineation boundary adjacent to the property is provided in Figure 1.

Existing Conditions

The southern portion of the property currently supports a goat pen, pony pen, two chicken coops and a small barn (about 250 square feet) (see Figure 1). Fencing is used to separate the livestock pens and is also located along the north bank of Sisar Creek. This fencing is approximately 3.5 feet tall and composed of wire mesh about 1.5 by 3 inches. Barbed wire is not used on this fencing.

Proposed Barn

The proposed 864 square foot (24 by 36 feet) barn would be located approximately 20 feet east of the existing barn and separated from Sisar Creek by two fence lines. The site of the proposed barn is nearly level and only minimal earthwork would be required for the barn foundation. The proposed barn would be located as close as 90 feet from wetlands of Sisar Creek (see Figure 1).

Setback Assessment

Direct Impacts. The proposed barn would not directly affect wetlands, including removal of vegetation, grading or other earthwork, placement of structures, or disturbance of the substrate.

Hydrologic Impacts. The proposed barn would not indirectly affect adjacent wetlands as it would not affect surface flow velocity, siltation, flow volume, interrupt contiguous habitats, block seed dispersal routes, increase vulnerability of wetland species to exotic weed invasion or local extirpation, alter maintenance of hydrological conditions. The small increase in impervious surfaces (barn roof, 864 square feet) would be surrounded by natural soil surfaces and have only a negligible effect on stormwater runoff volumes and rates.

Water Quality Impacts. Runoff from the proposed barn roof would flow to the south and percolate within the existing goat pen and not affect water quality in Sisar Creek.

Rare, Endangered or Threatened Species. Based on review of the California Natural Diversity Data Base, the nearest special-status species is the southwestern pond turtle (species of special concern), reported in Santa Paula Creek 0.5 miles to east of the property. In any case, the proposed barn would have no direct or indirect effect on special-status species or their habitat.

Note that Southern California steelhead historically occurred in Sisar Creek but has been extirpated, see attached distribution map in California Department of Fish and Wildlife (2024), primarily due to the failed fish ladder just upstream of Santa Paula. Foothill yellow-legged frog has been extirpated in Ventura County, see attached pages from the listing petition from the Center for Biological Diversity (2016).

Wildlife Movement. The property zoning includes a habitat conservation wildlife corridor overlay. The proposed barn would be separated from wetlands by two existing fence lines and existing livestock pens holding goats and ponies. The proposed barn would not introduce a new land use, or other features (such as fences, walls or other barriers) that may affect wildlife movement along Sisar Creek.



Cumulative Impacts to Biological Resources. The proposed barn would not have any impacts; therefore, it would not incrementally contribute to cumulative impacts.

Conclusion. The proposed 90-foot setback of the barn from wetlands is adequate to avoid adverse impacts to the wetlands of Sisar Creek.

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Should you have any questions concerning the findings of this assessment, please contact me at mingamells@padreinc.com or 805/644-2220 ext. 413 at your earliest convenience.

Sincerely,

Padre Associates, Inc.

A handwritten signature in blue ink, appearing to read "Matt Ingamells", is written over the printed name.

Matt Ingamells
Senior Biologist

Attachments: Site Map

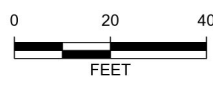
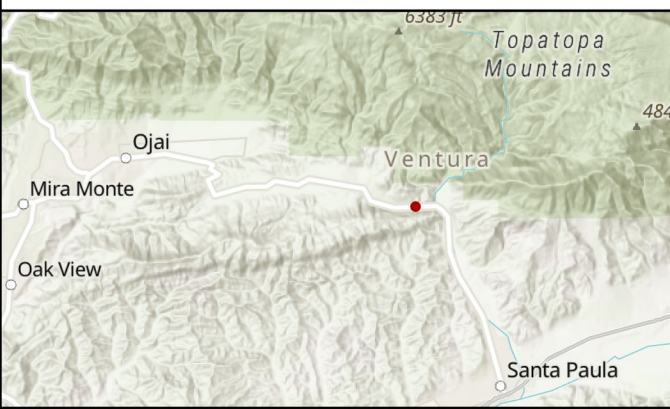
Steelhead distribution map

Pages from the foothill yellow-legged frog listing petition



- LEGEND:**
- Coast Live Oak
 - - - Existing Fence
 - Wetland Boundary
 - ▨ Proposed Barn

MAP EXTENT:



Source: Google Earth Imagery 5/11/23, County of Ventura
Coordinate System: NAD 1983 StatePlane California V FIPS 0405 Feet.
Notes: This map was created for informational and display purposes only.



PROJECT NAME: 15315 OJAI-SANTA PAULA ROAD VENTURA COUNTY, CA	
PROJECT NUMBER: 2402-1661	DATE: April 2024

SITE MAP

State of California
Natural Resources Agency
Department of Fish and Wildlife

**REPORT TO THE FISH AND GAME COMMISSION
CALIFORNIA ENDANGERED SPECIES ACT STATUS REVIEW OF
SOUTHERN CALIFORNIA STEELHEAD (ONCORHYNCHUS MYKISS)**

January 2024



Southern California Steelhead Rainbow Trout, CDFW photo

Prepared by
California Department of Fish and Wildlife



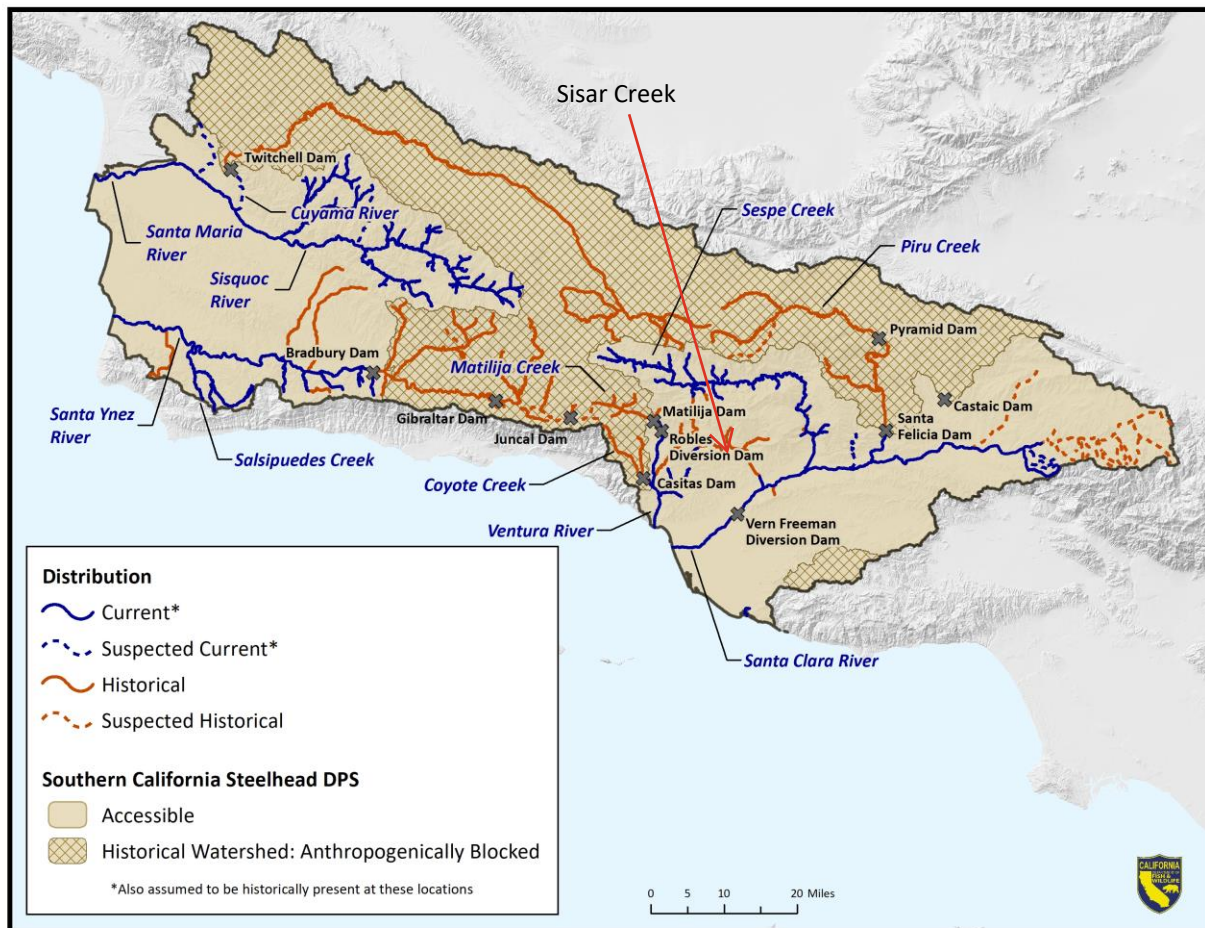


Figure 7. Map of the Monte Arido Highlands BPG depicting known and suspected current and historical distribution.

4.3.1.1 Santa Maria River

The Santa Maria River runs from the confluence of the Cuyama and Sisquoc rivers to the ocean and encompasses 1,790 square miles of watershed (Becker and Reining 2008). Historically, the Santa Maria River served mainly as a corridor for steelhead migrating to and emigrating from the Cuyama and Sisquoc rivers, rather than as habitat for spawning and rearing (Titus et al. 2010).

Hatchery stocking of *O. mykiss* occurred in the early 1930s in the Sisquoc and Cuyama watersheds (Titus et al. 2010). However, local newspaper records from the late 1800's reported abundant harvests of *O. mykiss* in the Sisquoc River watershed well before hatchery stocking occurred (Camm Swift, Emeritus, Section of Fishes, Natural History Museum of Los Angeles County, personal communication). In the early to mid-1940s, juvenile steelhead from the Santa Ynez River were rescued and translocated to the Santa Maria River. Tributaries of the Cuyama

BEFORE THE FISH AND GAME COMMISSION

**Petition to List the Foothill Yellow-Legged
Frog (*Rana boylei*) As Threatened
Under the California Endangered Species Act**



Photo by Todd Steiner (used with permission)

Submitted To: California Fish and Game Commission
1416 Ninth Street
Box 944209
Sacramento, CA 94244-2090
FGC@fgc.ca.gov

Submitted By: Center for Biological Diversity

Date: December 14, 2016

EXECUTIVE SUMMARY

The Center for Biological Diversity petitions the California Fish and Game Commission to list the foothill yellow-legged frog (*Rana boylei*) as a threatened species under the California Endangered Species Act.

Adult foothill yellow-legged frogs are moderately-sized (1.5 to 3 inches) with a distinctive lemon-yellow color under their legs. They inhabit partially shaded, rocky perennial streams and rivers at low to moderate elevations in Pacific Coast drainages as well as the lower western slopes of the Sierra Nevada mountains. The foothill yellow-legged frog life cycle is synchronized with the seasonal timing of streamflow conditions. Adult frogs move throughout stream networks from winter refugia to mating habitat where eggs are laid in spring and tadpoles rear in summer. For breeding they require streams with riffles containing cobble-sized or larger rocks as substrate to be used as egg laying sites. Non-breeding habitat is characterized by perennial water where they can forage through the summer and fall months.

In California, foothill yellow-legged frogs were once found from the Oregon border to at least as far south as the Upper San Gabriel River, Los Angeles County; the species also possibly occurred historically as far south as Orange County, southwestern San Bernardino County and San Diego County. Different regions of California may contain distinct populations or subspecies of foothill yellow-legged frogs. Foothill yellow-legged frogs have now disappeared from more than half of their historically occupied locations throughout California and Oregon, resulting in a range contraction in northern and southern California.

The survival of the foothill yellow-legged frog in California is threatened by a combination of factors, including habitat alteration and destruction from: dams, water development and diversions; logging; marijuana cultivation; mining; roads and urbanization; recreation; and off-road vehicles. Frogs are also threatened by impacts from invasive species, disease, climate change, and pollution.

The foothill yellow-legged frog is now extirpated from all of southern California south of San Luis Obispo County and is nearly extirpated from the south coast region. The species is extirpated or near extirpation in many areas of the central coast and the San Francisco Bay Area, with declines in many drainages in these regions. Significant populations remain in the Diablo Range and throughout Sonoma County. There have been documented declines of frog populations in the upper Sacramento River basin although some significant and many small populations remain in the upper Sacramento River basin.

The largest foothill yellow-legged frog populations in California are in the north coast range, with healthy populations scattered throughout the region. The strongholds for the species are in the Smith River, Red Cap Creek tributary of the Klamath River, South Fork Trinity River, South Fork Eel River; Redwood Creek, coastal tributaries in Mendocino County, and Russian River tributaries. However, only 6 sites in northern California have large populations exceeding 100 breeding females per kilometer of river, with an additional 9 sites having more than 50 breeding females per km. There have been documented declines in the northern coastal California region, with frogs lost from 39 of 165 historical sites (24%) in the north coast.

Foothill yellow-legged frogs are nearly extirpated from the southern portion of the Sierra Nevada. They have disappeared from Yosemite, Sequoia and Kings Canyon National Parks, and are near extirpation in Sequoia and Sierra National Forests, with few remaining populations and limited distribution. Although populations persist in many river basins in the northern and central Sierras, including the American, Clavey, Cosumnes, Feather, Merced, Mokelumne, Stanislaus, Tuolumne, and Yuba rivers, many former populations have been lost and the majority of recent observations in Sierran national forests are of small and scattered populations, with limited evidence of successful reproduction. At least half of the known historical locations have been lost in every northern and central Sierra county except Plumas County.

The overall population trend for the foothill yellow-legged frog in California is distressing. The species had disappeared from 45 percent of its historic range in California by 1991, and frog numbers at many of the formerly large frog populations have crashed in recent years. While the number of populations is important, population size is also critical; as of 2005 only 30 of the 213 sites in California with foothill yellow-legged frogs (14%) had populations estimated to be 20 or more adult frogs.

Existing federal and state management and regulatory mechanisms - such as occurrence on federally protected lands, consideration under the National Environmental Policy Act or Clean Water Act or California Environmental Quality Act, or coverage under federal Habitat Conservation Plans and state Natural Community Conservation Plans - have proved inadequate to prevent the decline of foothill yellow-legged frogs.

Recommended management actions for foothill yellow legged frogs will vary depending on the type of river system where a given extant population remains, either with flows regulated by dams or in free-flowing systems that may be subject to other forms of human perturbation (such as illegal diversion of flows in summer for *Cannabis* cultivation or excessive sedimentation and hillslope erosion due to road building and other types of land use in the upland portions of the watersheds). In rivers with dams, avoiding aseasonal flow fluctuation which could cause the stranding and scouring of egg masses and tadpoles should be avoided. Maintaining thermal regimes conducive to larval survival and rapid development will also be important. Recovery actions for the foothill yellow-legged frog should include management of non-native bullfrogs which are predators as adults, competitors as tadpoles and reservoir hosts for parasites and disease organisms that have been shown to have negative effects on foothill yellow legged frogs. Management of non-native fish and crayfish which are predators of frogs, tadpoles, and egg masses will also be important elements of recovery. Reintroduction into stream systems with appropriate habitat should also be considered.

A PETITION TO THE STATE OF CALIFORNIA FISH AND GAME COMMISSION

For action pursuant to Section 670.1, Title 14, California Code of Regulations (CCR) and Sections 2072 and 2073 of the Fish and Game Code relating to listing and delisting endangered and threatened species of plants and animals.

I. SPECIES BEING PETITIONED:

Common Name: Foothill yellow-legged frog

Scientific Name: (*Rana boylei*)

II. RECOMMENDED ACTION:

(Check appropriate categories)

a. List X

b. Change Status ☐

As endangered

As threatened X

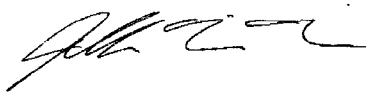
c. Or Delist ☐

III. AUTHOR OF PETITION:

Name: Jeff Miller
Address: 1212 Broadway, Suite 800
Oakland, CA 94612
Phone Number: (510) 499-9185

I hereby certify that, to the best of my knowledge, all statements made in this petition are true and complete.

Signature:



Date: December 14, 2016

Attachement B: Petition to List Foothill Yellow-Legged Frog (<i>Rana boylei</i>) As Threatened Under the California Endangered Species Act December 14, 2016

BEFORE THE FISH AND GAME COMMISSION

**Petition to List the Foothill Yellow-Legged
Frog (*Rana boylei*) As Threatened
Under the California Endangered Species Act**



Photo by Todd Steiner (used with permission)

Submitted To: California Fish and Game Commission
1416 Ninth Street
Box 944209
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Submitted By: Center for Biological Diversity

Date: December 14, 2016

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In California, foothill yellow-legged frogs were once found from the Oregon border to at least as far south as the Upper San Gabriel River, Los Angeles County; the species also possibly occurred historically as far south as Orange County, southwestern San Bernardino County and San Diego County. Different regions of California may contain distinct populations or subspecies of foothill yellow-legged frogs. Foothill yellow-legged frogs have now disappeared from more than half of their historically occupied locations throughout California and Oregon, resulting in a range contraction in northern and southern California.

The survival of the foothill yellow-legged frog in California is threatened by a combination of factors, including habitat alteration and destruction from: dams, water development and diversions; logging; marijuana cultivation; mining; roads and urbanization; recreation; and off-road vehicles. Frogs are also threatened by impacts from invasive species, disease, climate change, and pollution.

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Existing federal and state management and regulatory mechanisms - such as occurrence on federally protected lands, consideration under the National Environmental Policy Act or Clean Water Act or California Environmental Quality Act, or coverage under federal Habitat Conservation Plans and state Natural Community Conservation Plans - have proved inadequate to prevent the decline of foothill yellow-legged frogs.

Recommended management actions for foothill yellow legged frogs will vary depending on the type of river system where a given extant population remains, either with flows regulated by dams or in free-flowing systems that may be subject to other forms of human perturbation (such as illegal diversion of flows in summer for *Cannabis* cultivation or excessive sedimentation and hillslope erosion due to road building and other types of land use in the upland portions of the watersheds). In rivers with dams, avoiding aseasonal flow fluctuation which could cause the stranding and scouring of egg masses and tadpoles should be avoided. Maintaining thermal regimes conducive to larval survival and rapid development will also be important. Recovery actions for the foothill yellow-legged frog should include management of non-native bullfrogs which are predators as adults, competitors as tadpoles and reservoir hosts for parasites and disease organisms that have been shown to have negative effects on foothill yellow legged frogs. Management of non-native fish and crayfish which are predators of frogs, tadpoles, and egg masses will also be important elements of recovery. Reintroduction into stream systems with appropriate habitat should also be considered.

NOTICE OF PETITION

Center for Biological Diversity
1212 Broadway, Suite 800
Oakland, CA 94612
Contact: Jeff Miller
Phone: (510) 499-9185
E-mail: jmiller@biologicaldiversity.org

Petitioner Center for Biological Diversity formally requests that the California Fish and Game Commission list the Foothill yellow-legged frog (*Rana boylei*) as a threatened species under the California Endangered Species Act ("CESA"), Fish and Game Code §§ 2050 et seq. This petition sets in motion a specific administrative process as defined by Fish and Game Code §§ 2070-2079, placing mandatory response requirements on the Commission and very specific time constraints upon those responses.

Petitioner Center for Biological Diversity is a national nonprofit organization with more than 1.1 million members and online activists dedicated to the protection of endangered species and wild places, through science, policy, education, citizen activism and environmental law.

Contents:

Executive Summary.	1
Notice of Petition.	3
 NATURAL HISTORY AND STATUS OF FOOTHILL YELLOW-LEGGED FROG	 5
NATURAL HISTORY.	5
Description.	5
Taxonomy.	6
Range in California and Documented Range Contraction.	6
Habitat Requirements	11
Life History.	11
Natural Mortality.	14
CHANGES IN DISTRIBUTION AND ABUNDANCE	14
Southern California	16
South Coast.	21
Central Coast	23
Bay Area	31
Upper Sacramento River	43
Northern Coastal California	49
Southern Sierra Nevada	65
Central/ Northern Sierra Nevada	75
Population Trends	95
 FACTORS AFFECTING ABILITY TO SURVIVE AND REPRODUCE.	 100
Climate Change and UV-Radiation	101
Dams, Water Development and Diversions	102
Disease	105
Invasive Species	107
Livestock Grazing	108
Logging	108
Marijuana Cultivation	109
Mining	109
Off-road Vehicles	110
Pollution	110
Recreation	112
Roads and Urbanization	112
 INADEQUACY OF EXISTING REGULATORY MECHANISMS	 113
RECOMMENDED MANAGEMENT AND RECOVERY ACTIONS	119
 BIBLIOGRAPHY OF LITERATURE CITED.	 121

NATURAL HISTORY AND STATUS OF FOOTHILL YELLOW-LEGGED FROG

A. NATURAL HISTORY

Description

Adult foothill yellow-legged frogs are moderate-sized (37 to 82 mm snout-urostyle length) ranid frogs with indistinct dorsolateral folds, fully webbed feet, slightly expanded toe tips, and rather thick, rough pebbly skin (Stebbins 1951, 2003; Zweifel 1955). Dorsal color is highly variable and is usually light and dark mottled gray, olive, or brown, but variable amounts of brick red are often present, and a pale triangle is often located between the eyes and the snout (Zweifel 1955; Nussbaum et al. 1983; Jones et al. 2005). The undersurfaces of the posterior abdomen and ventral surfaces of the rear legs are varying shades of yellow, which fades to white anteriorly on the belly (Stebbins 1951; Zweifel 1955). Females attain larger sizes than males (Jennings and Hayes 1994). Mature males have a dark swollen bump or nuptial pad on the dorso-medial surface of each thumb that becomes darker, slightly larger, and rougher to the touch during the breeding season (Hayes et al. 2016). Males also have proportionally larger forearm muscles and narrower waists than females (Hayes et al. 2016).

Juvenile foothill yellow-legged frogs look similar to adults except for their smaller size (14 to 36 mm snout-urostyle length), more contrasting dorsal coloration, and lack of significant yellow on their undersurfaces (Stebbins 1951; Zweifel 1955; Nussbaum et al. 1983; Jones et al. 2005). Undersurfaces of the youngest juveniles are cream or flesh colored and the yellow color makes its first appearance on the calves and thighs, expanding anteriorly and posteriorly as juveniles grow in size (Hayes et al. 2016).

Newly hatched tadpoles are dark brown to black and typically measure 7 to 8 mm in total length (Storer 1925; Zweifel 1955). As tadpoles grow, their coloration turns an olive color with coarse brown mottling dorsally. The ventral surface of the body is silvery and nearly opaque, and the coiled intestine is barely visible. The body is more flattened, and the tail fin, tallest at its mid-portion, has a relatively broad musculature (Zweifel 1955). When viewed from above, the eyes of foothill yellow-legged frog tadpoles are dorsally positioned so they are located within the outline of the head in bird's eye view (Hayes et al. 2016). Tadpoles have a large, downward-oriented, almost suction-like mouth with several rows of denticles or labial teeth, with the number of rows increasing with development (Hayes et al. 2016).

Egg masses contain from about 100 to more than 3,000 eggs, depending on the size of the female and geographic variation among populations (Kupferberg et al. 2009b). Upon deposition, the mass is compact and the jelly is highly transparent and has a hyaline blue tint. Within 6 hours, the egg mass absorbs water, loses the bluish tint, expands to a long-axis diameter of 45 to 90 mm, and resembles a cluster of grapes (Hayes et al. 2016). Each ovum is dark brown to black in appearance and surrounded by three jelly envelopes. Individual eggs range from 1.0 to 2.3 mm in diameter, and the outermost of the three jelly envelopes ranges from 3.9 to more than 6 mm in diameter (Storer 1925; Zweifel 1955).

Taxonomy

Rana boylei was named after Dr. Charles Elisha Boyle, a California “49er” that collected the type specimens in 1850 (Jennings 1987). The foothill yellow-legged frog was first described as a species by Baird (1854). A half-century of taxonomic uncertainty followed with several name changes (Zweifel 1968). Since 1955, the foothill yellow-legged frog has been recognized as a distinct species in the family Ranidae (Zweifel 1955; Collins 1990). The “boylei” group of western ranids seems to have diverged from other ranids about 8 million years ago (Macey et al. 2001). Based on morphological analyses, *R. boylei* was thought to be most closely related to *R. muscosa*, the mountain yellow-legged frog (Zweifel 1955). However, a recent phylogenetic analysis allied it most closely to *R. pretiosa*, the Oregon spotted frog (Macey et al. 2001).

Several studies have detected intraspecific genetic variation (Case 1978a,b; Lind 2005; Dever 2007). Recent mitochondrial DNA analysis by Lind (2005) and Lind et al. (2011) identified significant genetic partitioning between coastal and Sierra Nevada foothill yellow-legged frog populations, two distinct more northerly groupings, and significant differentiation of a single sample in the southern Sierra Nevada from samples in the central and northern Sierra (a pattern congruent with that of other herpetofauna species widespread across the Sierra, e.g., Macey et al. 2001). Conclusions about evolutionary relationships did not involve formal taxonomic description for any of these groups in either of these studies, but Lind (2005) and Lind et al. (2011) noted that some populations may ultimately be regarded as deserving formal taxonomic recognition.

Range in California and Documented Range Contraction

The overall range of the foothill yellow-legged frog historically included lower elevation streams draining the Pacific slope, from the upper reaches of the Willamette River system, Oregon, south to northwestern Baja California (NatureServe 2011; Hayes et al. 2016). In California, foothill yellow-legged frogs were found from the Oregon border to at least as far south as the Upper San Gabriel River, Los Angeles County; the species also possibly occurred historically as far south as Orange County, southwestern San Bernardino County and San Diego County.

The species has now disappeared from more than half of its historically occupied locations (Lind 2005) throughout its range in California and Oregon, resulting in a range contraction at the northern and southern ends of the range. The decline is especially severe in the northern portion of the range and in southern California (see Sweet 1983; Jennings and Hayes 1994; Hayes and Jennings 1988; Jennings 1995). Maps produced by Lind (2005), Olson and Davis (2009), U.S. Forest Service (2011), Hayes et al. (2016), and Thomson et al. (2016) illustrate the range contraction (Figures 1 through 4).

Howard et al. (2015) compiled the California Freshwater Species Database from nearly 500 sources (TNC 2015), the first comprehensive geospatial database of California’s freshwater species standardized into single format, providing a single source for geodata covering the plants and animals that rely on California’s freshwater resources to survive. A description of the methods used by Howard et al. (2015) to compile the data is available in a recent study published in PLoS ONE. Mapping the *R. boylei* occurrence data from the California Freshwater Species Database would also illustrate the overall range contraction of the species.

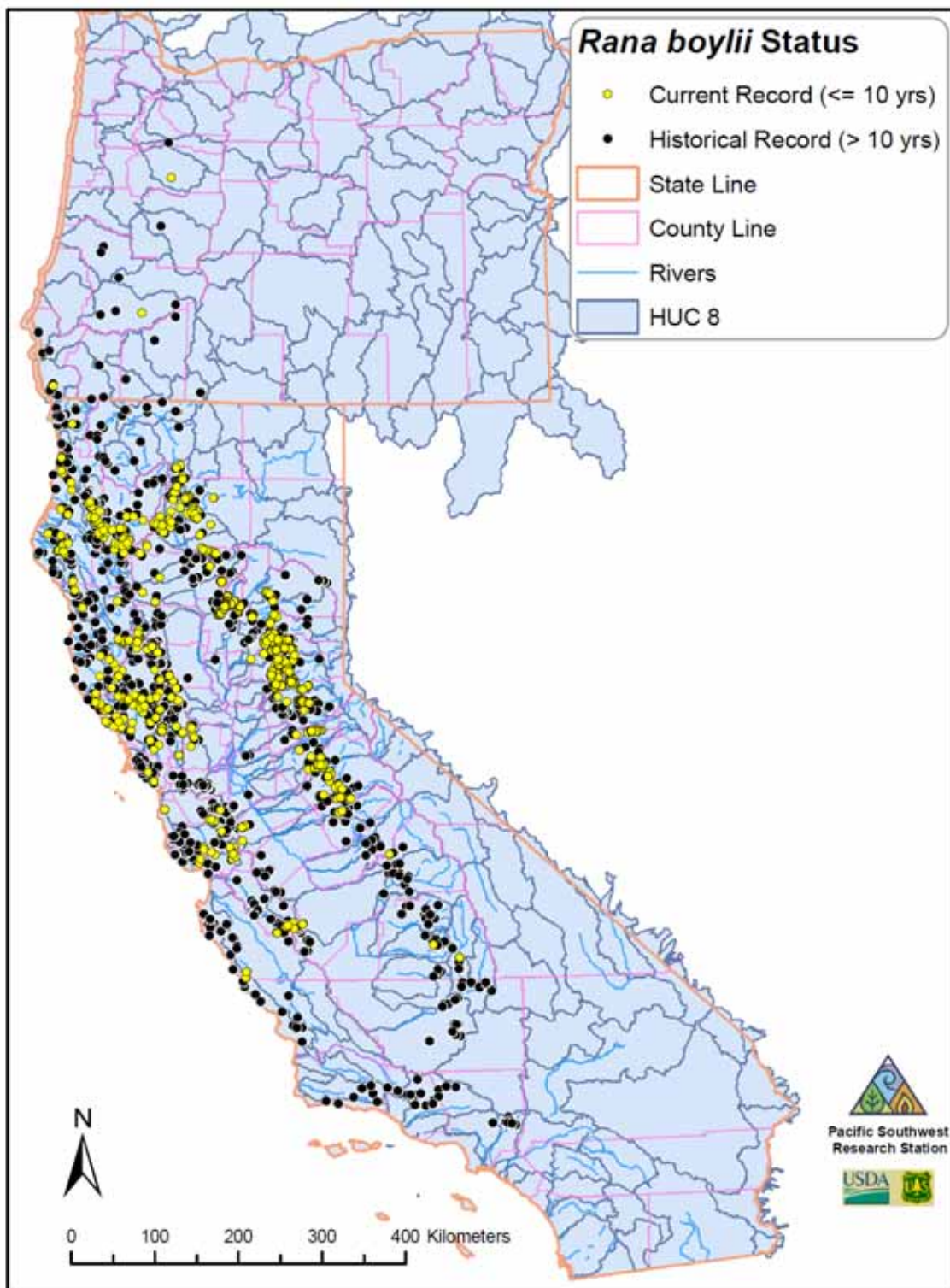


Figure 1. Range contraction map from USFS (2011) comparing historic (> 10 yrs) and recent (< 10 years) locality records. The map was developed using over 6,000 locality records from museum collections, research projects, technical reports, and government databases.

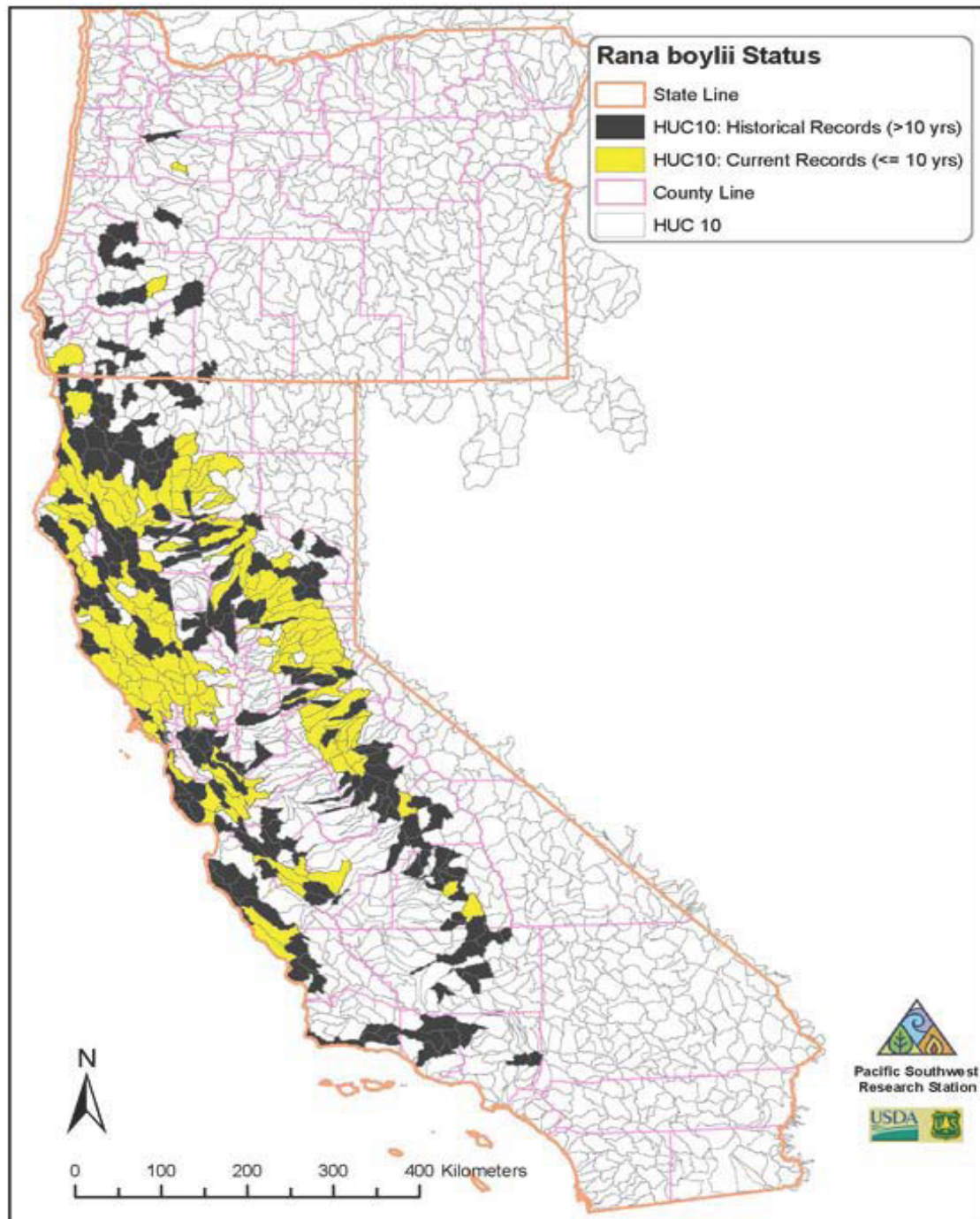


Figure 2. Range contraction map from USFS (2011) comparing historic (> 10 yrs) and recent (< 10 years) presence by watershed. The map was developed using over 6,000 locality records from museum collections, research projects, technical reports, and government databases.

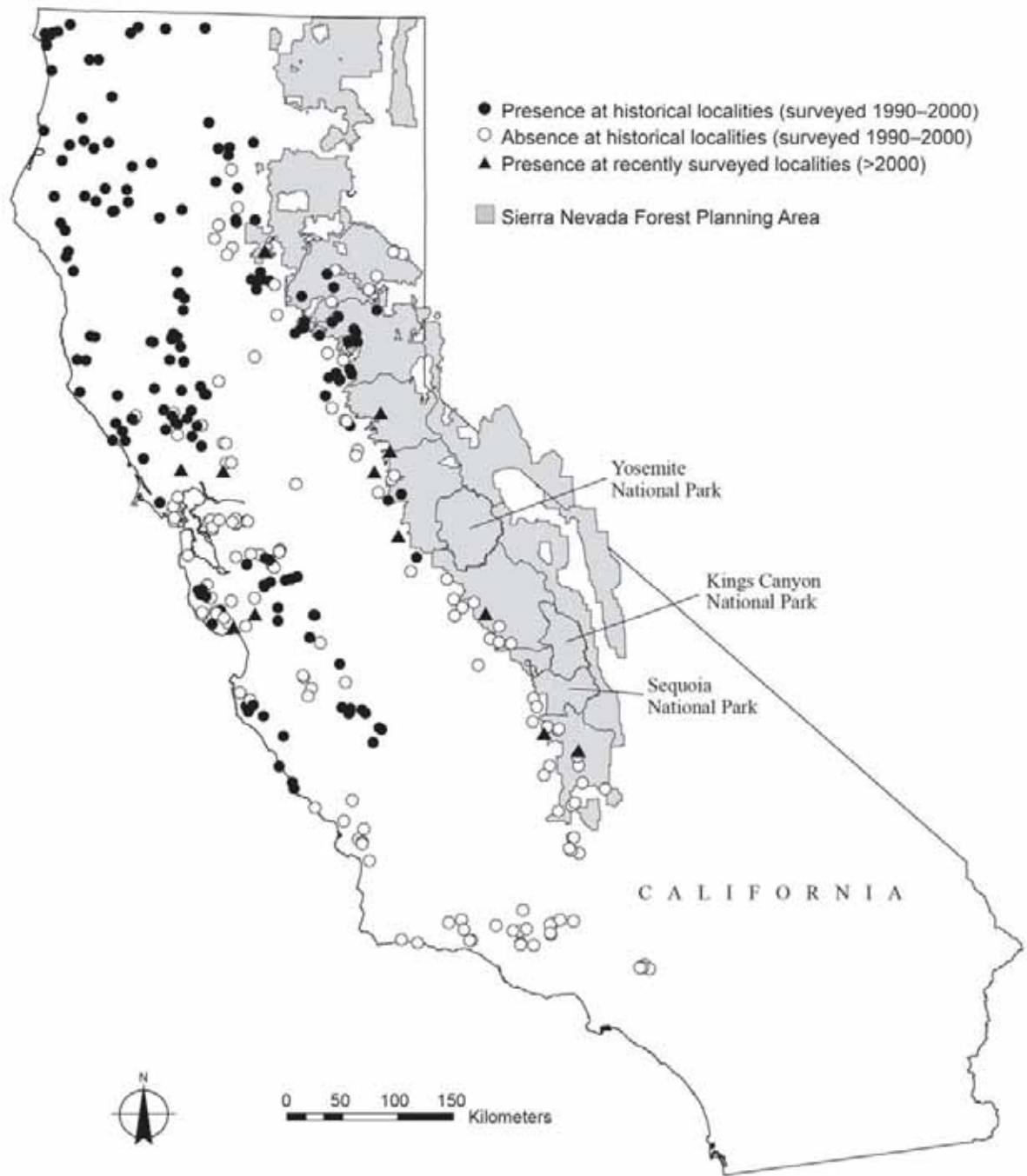


Figure 3. Distribution and status map by Hayes et al. (2016) of the foothill yellow-legged frog (*Rana boylei*) in California, based on an update of Lind (2005). Lind (2005) assessed status prior to year 2000. The Hayes et al. (2016) map overlays new sightings from post-2000 localities on that previous analysis.

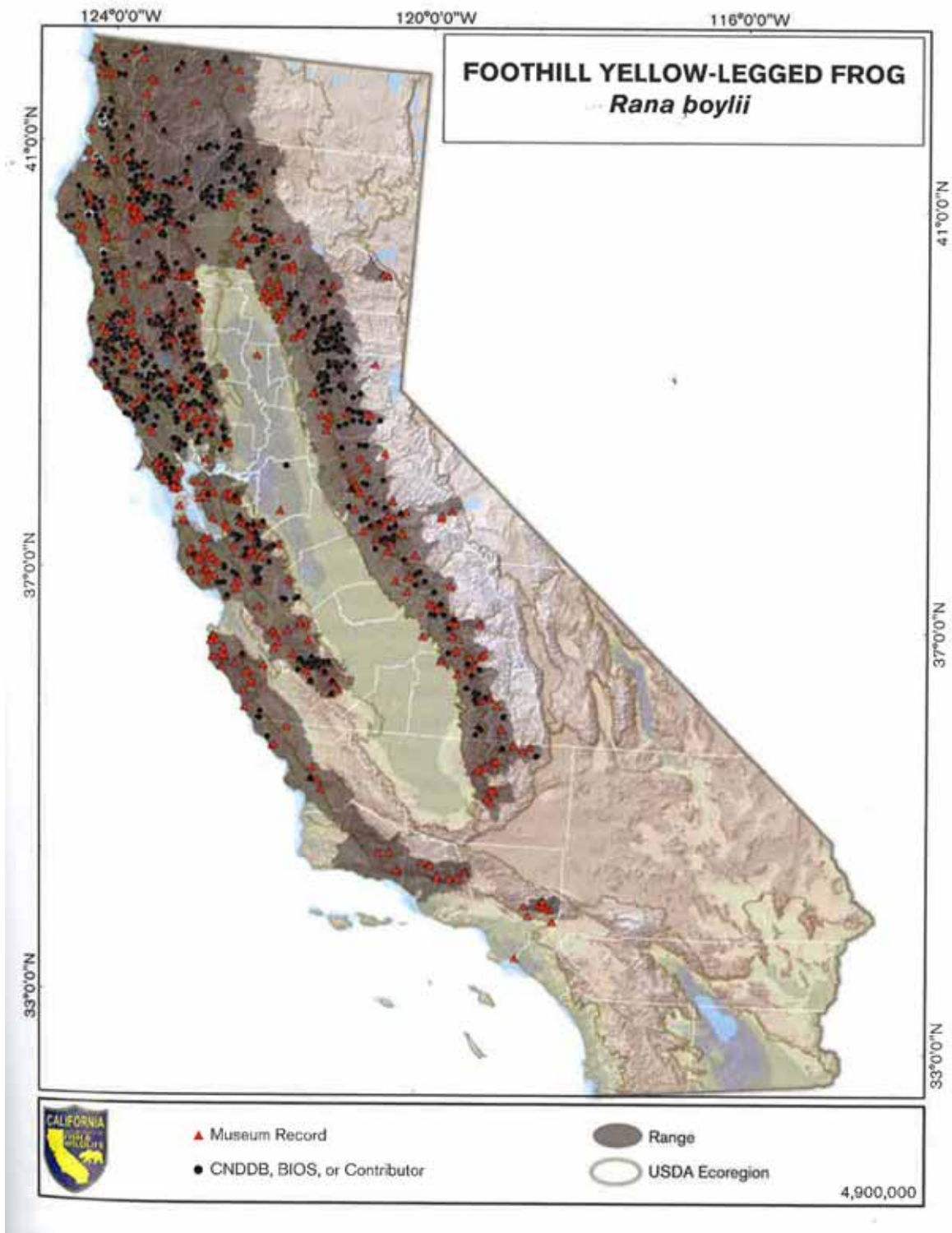


Figure 4. Distribution map by Thomson et al. (2016) of museum records and CNDDb/BIOS records of the foothill yellow-legged frog in California

Habitat Requirements

Foothill yellow-legged frogs inhabit partially shaded, rocky perennial streams and rivers at low to moderate elevations, across a range of vegetation types including chaparral, oak woodland, mixed coniferous forest, riparian sycamore and cottonwood forest, and wet meadows (Nussbaum et al. 1983; Stebbins 1985; Hayes and Jennings 1988). Foothill yellow-legged frogs are primarily stream dwelling. Stebbins (1985) describes foothill yellow-legged frogs as stream or river frogs found mostly near water with rocky substrate, often found in or near riffles, and on open, sunny banks. Other authors have expanded this description, and offer variations (e.g. Storer 1925; Fitch 1938; Zweifel 1955; Hayes and Jennings 1988; Kupferberg 1996a; Lind et al. 1996; Van Wagner 1996). Jennings and Hayes (1994a) noted that suitable streams have riffles containing cobble-sized (7.5 cm diameter) or larger rocks as substrate. Habitat suitable for egg laying where flow velocities are less than those found in riffles varies among regions and size of river (Bondi et al. 2013). Within a single watershed, these frogs can occupy a wide range of stream sizes (from 1st to 7th stream order, Bury and Sisk 1997), but occupied streams are generally small to mid-sized with some shallow, flowing water (Hayes and Jennings 1988). Fuller and Lind (1992) observed subadults on partly shaded (20%) pebble/cobble river bars near riffles and pools. Less typically, occupied streams lack a rocky, cobble substrate (Fitch 1938). Other types of riparian habitats include isolated pools and vegetated backwaters (Hayes and Jennings 1988, Ashton et al. 1998). Habitat requirements vary seasonally and with life stage, but this species is absent, or occurs at low density even in suitable physical habitat if introduced aquatic predators (e.g. bullfrogs, bass) are present (Hayes and Jennings 1986, 1988; Kupferberg 1997a).

Life History

The foothill yellow-legged frog life cycle is synchronized with the seasonal timing of streamflow conditions. Radiotelemetry studies show that adult frogs move throughout dendritic networks of streams from winter refugia, such as small tributary streams, where they can avoid mortality due to flooding (Bourque 2008; Gonsolin 2010) to mating habitat in wider and more sunlit mainstem channels where eggs are laid in spring and tadpoles graze on algae in summer (Wheeler and Welsh 2008). Non-breeding habitat is characterized by perennial water where they can forage through the summer and fall months. Springs, seeps, or other moist habitats such as woody debris and clumps of sedges occurring at high-water lines may serve as refugia during periods of high stream flow in winter (Van Wagner 1996; Rombough 2006).

Breeding is triggered by warming water temperatures, decreasing streamflows, and increasing daylength during the transition between the wet and dry season. Breeding sites are generally (but not always) located in low-gradient stream reaches at depositional features such as lateral point bars and pool tail-outs (Kupferberg 1996a; Wheeler and Welsh 2008). Breeding may commence as early as March in warm coastal locations and as late as July in snowmelt dominated rivers (Storer 1925; Zweifel 1955; Ashton et al. 1998; AmphibiaWeb 2012; Wheeler et al. 2014).

Like most ranid frogs, males probably defend areas around themselves during breeding season (Martof 1953; Emlen 1968). Foothill yellow-legged frog vocalizations are seldom heard. The voice is a guttural, grating sound on one pitch or with rising inflection, a

single croak lasting $\frac{1}{2}$ to $\frac{3}{4}$ of a second. Four or five croaks may be given in rapid series followed by a rattling sound, the entire sequence lasting about 2.5 seconds (Stebbins 1985). While much of the mate calling occurs underwater (MacTague and Northern 1993), males also call from above water. Above water calls are faint and are not generally heard over distances greater than 50 meters (Ashton et al. 1998). Examples of both above water, and underwater calls are documented and described on Frog and Toad Calls of the Pacific Coast (Davidson 1995).

Females often oviposit eggs in shallow water toward the margin of streams, but in some large rivers they oviposit at depths greater than 1 m and distances up to 20 meters from the water's edge (Mokelumne River, unpublished data from Garcia and Associates for PG&E). Clutches of eggs are often attached to the flow-protected lee sides of rocky substrates within a narrow range of flow velocities (Kupferberg 1996a; Bondi et al. 2013). Tadpoles disperse from egg laying sites but similarly require protection from swift currents, especially when they approach metamorphosis and are poor swimmers (Kupferberg et al. 2011). Cobble and pebble are the preferred substrate for egg mass attachment, but egg masses have been found attached to aquatic vegetation, woody debris, gravel and bedrock (Fuller and Lind 1992; Ashton et al. 1998, Bondi et al. 2013). Females lay a distinct cluster of eggs, with average clutch sizes ranging from 100 to 1,100 eggs (Storer 1925; Wright and Wright 1949) but can reach more than 3,000 eggs (Kupferberg et al. 2009c). Larger and more fecund females tend to breed first and small females oviposit clutches with fewer eggs as the breeding season progresses (Kupferberg et al. 2009c, Gonsolin 2010).

Eggs hatch in 5 to 30 days, or more (Zweifel 1955). In the mainstem Trinity River, eggs hatch in 27 to 36 days (Ashton et al. 1998). The slower development is probably due to colder temperatures from dam released water (Wheeler et al. 2015; Railsback et al. 2016). At the time of hatching, the embryos are at a Gosner stage of 20 to 22 (Ashton et al. 1998). In the absence of disturbance, the tadpoles will remain associated with the egg mass for several days after hatching then disperse to local interstices of the gravel bed, often moving downstream in areas of moderate flow (Ashton et al. 1998). Larval growth rate and survival depend on water temperature and interactions between temperature and quality of algal food resources (Catenazzi and Kupferberg 2013; Furey et al. 2015; Railsback et al. 2016). Tadpoles actively thermoregulate (Brattstrom 1962) and prefer warm temperatures at or above approximately 20C (Kupferberg et al. 2013). Larval growth rate and survival to metamorphosis are inversely proportional to flow velocity conditions (Kupferberg et al. 2011), *i.e.* growth is slower at higher velocities. Metamorphosis generally occurs in three to four months. Male foothill yellow-legged frogs can reach sexual maturity at age 1 to 2 years, at a length of about 40 mm (Zweifel 1955; Gonsolin 2010), whereas females mature at two or three years of age depending on latitude and elevation of the population (Kupferberg et al. 2009c; Gonsolin 2010), with Central Coast populations maturing at earlier ages. Some individuals may reproduce as early as 6 months after metamorphosis (Jennings 1988).

Rana boylei tadpoles feed on periphyton scraped from rocks or plants. They seem to grow fastest feeding on epiphytic diatoms growing on filamentous algae such as *Cladophora* sp., and have been observed to preferentially graze on this algal type (Jennings and Hayes 1994; Ashton et al. 1998; Kupferberg 1997b). Tadpoles have been observed actively congregating on dead tadpoles and dead, open bivalves (Ashton et al. 1998). Metamorphosed frogs feed primarily on terrestrial invertebrates, but also eat

some aquatic invertebrates (Fitch 1936; Zeiner et al. 1988; Hothem et al. 2009). Adult diet includes flies, moths, mosquitoes, hornets, ants, beetles, grasshoppers, water striders, and snails (Fitch 1936; Nussbaum 1983; Csuti et al. 2001). Van Wagner (1996) provided a thorough literature review and a detailed diet analysis of post-metamorphic *R. boylei*. Analysis of 63 post-metamorphic *R. boylei* found terrestrial arthropods to be the primary (~90%) prey items year round, comprised of 87.5% insects and 12.6% arachnids (Van Wagner 1996). Foothill yellow-legged frogs capture their prey by waiting along stream edges and pouncing (Airola 1980).

The foothill yellow-legged frog is primarily diurnal and may be active year-round in locations where winter temperatures permit, with peak activity in April and May (Airola 1980).

Home ranges and dispersal patterns of the foothill yellow-legged frog are poorly understood (Jennings and Hayes 1994), but a handful of mark-recapture and telemetry studies exist. Frogs have been found 50 m (Nussbaum et al. 1983; Csuti et al. 2001) to 70-80 m (C. Rombough, pers. comm., as cited in Olson and Davis 2009) from water. Metamorphs have been found in pitfall traps at greater distances upslope from the water's edge (Twitty 1967). Along streams, Van Wagner (1996) reported seasonal movements of about 450 m for this species in California, and an 800 m movement distance is known from Oregon (C. Rombough, pers. comm., as cited in Olson and Davis 2009). A telemetry study by Bourque (2008) in Tehama County, California documented movement distances to and from breeding sites of 0.65 km and as far as 7.04 km for male and female foothill yellow-legged frogs, respectively, with median travel distances of 65.7 and 70.7 meters/day. Frogs used watercourses as movement corridors and rarely moved > 12 m from the stream channel. In Coyote Creek (Santa Clara Co., CA) radio-telemetry revealed that 60% of radio-tagged frogs were found underwater and under substrate and would not have been detected without the use of a transmitter (Gonsolin 2010). Gonsolin (2010) also reported that movements were restricted seasonally to the period between Feb and May for adults as they migrated to and from breeding sites. Adult females moved an average of 744 m, adult males moved 485 m, and juveniles moved 305 m.

During breeding season and summer, foothill yellow-legged frogs are rarely encountered far from permanent water. Adults congregate around breeding pools in spring, with month varying by latitude and elevation of the river. In late summer adults were found to be scarce along the main stem of the Trinity River, indicating that they may be dispersing into the vegetation, moving up tributaries, or reducing diurnal activity (Ashton et al. 1998). Recently metamorphosed frogs show a strong tendency to migrate upstream (Twitty 1967). This may be an evolutionary mechanism to repatriate individuals washed downstream from suitable habitat during the larval stage. During the winter, frogs have been observed in abandoned rodent burrows and under logs as far as 100 m from streams (Zeiner et al. 1988; Welsh 1994).

Movements of marked animals were not noted to occur November through March in Oregon (C. Rombough, pers. comm., as cited in Olson and Davis 2009). Radio telemetry tracking of post-breeding adult females in California documented dispersal distances from 0 to 7,043 m (R. Bourque, pers. comm., as cited in Olson and Davis 2009) where, over the course of 60 days, one female traveled upstream along the main channel of a perennial stream, then up intermittent and dry tributary channels, then over a ridge eventually working her way downstream to perennial waters in an adjacent watershed

(R. Bourque, pers. comm., as cited in Olson and Davis 2009). Other ranids have capabilities of dispersing kilometers overland; however, according to Nussbaum et al. (1983) this species is likely restricted to movements along streams or stream-riparian corridors. Their likely restriction to riparian corridors needs further study because of the low detectability of frogs in uplands. Dever's (2007) genetic study suggested that a distance of 10 km may effectively isolate frog populations along a river system (i.e., frogs this distance apart on a river are not part of a single interbreeding population). In river systems altered by human activities however, when populations are separated by dams, reservoirs, and reaches where flows artificially fluctuate daily, genetic isolation becomes apparent at much shorter distances of separation (Peek 2010, 2012).

Ashton et al. (1997) summarizes additional information on the natural history of the foothill yellow-legged frog. A more recent but less detailed account is provided by Morey (2007).

Natural Mortality

A number of native vertebrates (e.g. birds and snakes) and aquatic invertebrates (e.g. dragonfly nymphs) feed on foothill yellow-legged frogs, their tadpoles and eggs (Fitch 1936, 1941; Everdon 1948; Zweifel 1955; Milne and Milne 1980; Nussbaum et al. 1983; Jennings and Hayes 1994; Lind and Welsh 1994; Duellman and Trueb 1986; Jennings 1988; Moyle and Brown 1997; Ashton et al. 1998; Fellers 2005; Olson and Davis 2009). Among the documented predators of foothill yellow-legged frogs at various life stages are signal crayfish (*Pacifastacus leniusculus*) (Rombough and Hayes 2005); aquatic insects including caddisfly larvae (Limnephilidae), waterstriders (Gerridae), and veliid bugs (Veliidae) (Kupferberg 1996a; Rombough and Hayes 2005c); California tiger salamander (*Ambystoma californiense*) larva (Fidenci 2006) and the rough-skinned newt (*Taricha granulosa*, Everdon 1948); garter snakes, predominantly the aquatic garter snake (*Thamnophis atratus*) and the Sierran garter snake (*T. couchii*) (Fitch 1936, 1940, 1941; Stebbins 1951; Zweifel 1955; Nussbaum et al. 1983; Jennings and Hayes 1994; Lind and Welsh 1994); North American river otters (*Lutra canadensis*) (Hayes et al. 2016); mallard ducks (*Anas platyrhynchos*) (Rombough et al. 2005b). Foothill yellow-legged frogs are vulnerable to predation by fishes, both native and non-native including bass (*Micropterus* sp.) and Sacramento pikeminnow (*Ptychocheilus grandis*) (Brown and Moyle 1997; Corum 2003; Ashton and Nakamoto 2007; Paoletti et al. 2011). Other fish species are suspected to be predators (Hayes and Jennings 1988; Rombough and Hayes 2005c).

CHANGES IN DISTRIBUTION AND ABUNDANCE

The occurrence data discussed below are derived from an exhaustive search of museum specimen collection records, published literature on the species, environmental review documents, agency survey data, and all observation records in the California Natural Diversity Database as of December 9, 2016 (CNDDDB 2016). The information is organized geographically by region, then by county (as well as by National Forest for Sierra Nevada populations), then by watershed. All known historical and recent distribution and abundance data is given or summarized, as well as a summary of the recent status for each county. Many of the "small" populations discussed below are observations of single or scattered individuals, or small numbers of frogs with no

evidence of recruitment success. Information on locations, dates and numbers of frogs can be found in the footnotes for each section.

Determining the size of *R. boylei* populations for the purposes of making comparisons among locations, detecting trends, and assessing long term viability is a problematic task. Because of their cryptic coloration and their behavior, simple counts from visual encounter surveys may not accurately reflect the abundance of frogs comprising a breeding population of *R. boylei*. Outside of the breeding season, frogs can be dispersed across a range of channel sizes throughout a dendritic stream network (Welsh et al. 2005). Telemetry studies have shown that 60% of radio-tagged foothill yellow-legged frogs were found underwater and under substrate and would not have been detected without the use of a radio-transmitter (Gonsolin 2010). Clutches of eggs on the other hand, are comparatively visible and tractable to count (Figure 5).



Figure 5. Photo showing cryptic nature of *Rana boylei* in the stream channel relative to an egg mass. The arrow points to a male near the bank with its head out of the water.

Since each female *R. boylei* lays one discrete clutch of eggs, it has become standard practice to use egg mass counts, standardized by the length of the river reach searched, as a proxy for population density in lieu of more labor and time-intensive methods such as mark-recapture studies. A compilation of breeding season censuses of *R. boylei*

(Kupferberg et al. 2009c), indicated that the average abundance in free-flowing rivers is approximately 32 breeding females per river kilometer while genetic evidence demonstrates that there is significant isolation by distance between individuals occurring more than 10 km away from each other (Dever 2007). Thus Kupferberg et al. (2009c) chose 320 females in a 10 km reach to represent a typical *R. boylei* population and to use this starting number as a reference for comparison in a matrix projection model of population viability. When starting population size was set at 2 females per km (the low end of extant populations in regulated rivers), the model predicted a 13 fold increase in risk of extinction over a thirty year period in the absence of other anthropogenic stressors. When small starting population sizes were combined with hydrologic stressors such as ill-timed pulsed flows which cause eggmass and tadpole mortality, risk of extinction increased 20 to 45 times above the reference model. In contrast, when starting population size was set at 107 females per river km (an observed density on the SF Eel River), risk of extinction decreased 5 fold to less than a 1% chance of occurring. Given these modeling scenarios, we evaluate the number of frogs observed in the locality accounts listed below as follows. Populations with 100's of breeding adults are considered robust while populations with densities in the single digits are considered to be at high risk of local extinction.

Southern California

The foothill yellow-legged frog once occurred south of its current range, with documented historical localities in Santa Barbara, Ventura and Los Angeles counties. Based on museum specimens, the species possibly occurred as far south as Orange, southwestern San Bernardino and San Diego counties. However, foothill yellow-legged frogs are now extirpated from all of southern California south of San Luis Obispo County.

The foothill yellow-legged frog was formerly widespread and fairly common in the southern California coastal mountains, but has not been seen in or south of the Transverse Ranges since 1977 despite repeated searches (Sweet 1983; Jennings and Hayes 1994; Adams in prep.), and has completely disappeared from southern California south of Santa Barbara (Jennings 1995). High water conditions from 500-year frequency floods that occurred over much of southern California in 1969 are thought to have been one factor in the extirpation of the species from the region (Sweet 1983).

There has been some confusion regarding the species identification of historical collections of yellow-legged frogs in southern California and potential mislabeling of southern California mountain yellow-legged frogs (*Rana muscosa*) as foothill yellow-legged frogs (*R. boylei*). Historical locations of the mountain yellow-legged frog in southern California were from creeks and drainages in the San Gabriel, Big Bear, and San Jacinto Mountains of Los Angeles, San Bernardino, and Riverside Counties, as well as from an isolated population on Palomar Mountain in San Diego County (USFWS 2002). Mountain yellow-legged frogs in southern California historically inhabited a wide elevation range of localities, from 370 m (1,220 feet) to 2,290 m (7,560 feet) (USFWS 2002). The elevational range of the foothill yellow-legged frog (*R. boylei*) is from sea level to 1,830 m (6,000 feet) (Stebbins 1985; Stebbins and McGinnis 2012). *Rana boylei* and *R. muscosa* rarely co-occurred but did so at a couple of sites in the San Gabriel Mountains (Schoenherr 1976).

San Diego County

Although San Diego County is considered outside of the known historical range for *R. boylei*, there are historical collection records of yellow-legged frogs from a few scattered locations in San Diego County from 1928 to 1963, which deserve examination. There is some debate as to whether collection specimens from San Diego County labeled *R. boylei* were mislabeled (California Herps 2015).

The U.C. Museum of Vertebrate Zoology (UCMVZ 2001) had a collection specimen originally labeled *R. boylei* from 1951 in Doane Valley, Palomar Mt. State Park, at an elevation around 6,000 feet. An isolated population of the mountain yellow-legged frog (*R. muscosa*) was known to occur historically at Palomar Mountain (Vredenburg et al. 2007), and this specimen is now apparently correctly identified as *R. muscosa* (California Herps 2015; UCMVZ 2015).

The University of Kansas Museum of Natural History (UKMNH 2001) has a collection specimen from 1928 labeled *R. boylei* from Boulder Park in Jacumba (elevation 3,000 feet or 910 m), in the southeastern part of San Diego County near the U.S./Mexico border; but this specimen is thought to be misidentified (California Herps 2015). The University of Kansas Museum of Natural History also has 3 collection specimens labeled *R. boylei* from September 1928 from an unidentified location in San Diego County; however since the location cannot be determined, it is possible these were also misidentified mountain yellow-legged frogs (UKMNH 2001). Genetic and morphological analysis of these 4 UKMNH specimens would be useful.

The California Academy of Sciences has a specimen from 1963 labeled *R. boylei* which was collected from Carroll Canyon, just north of San Diego (CAS 2001); this would have been in the Penasquitos Creek watershed, near sea level and well below the known historical elevational range for *R. muscosa*.

Recent status: There are no recent foothill yellow-legged frog records from San Diego County. If the species did occur in San Diego County historically, it certainly has been extirpated.

Orange County

Although Orange County is considered outside of the known historical range for *R. boylei*, Bryant and Remington (1990) reported that foothill yellow-legged frogs were found in the 1940s (apparently as late as 1942) in the upper Newport Bay area. This would have been in the San Diego Creek/Bonita Creek watershed, near sea level and well below the known historical elevational range for *R. muscosa*.

Recent status: There are no known foothill yellow-legged frog records from Orange County. If the species did occur in Orange County historically, it certainly has been extirpated.

San Bernardino County

Although San Bernardino County is considered outside of the known historical range for *R. boylei*, there are historical collection records of yellow-legged frogs from the Santa Ana River drainage, in southwestern San Bernardino County, which deserve

examination.

Santa Ana River

The U.C. Museum of Vertebrate Zoology had specimen collection records of yellow-legged frogs originally labeled *R. boylei* (UCMVZ 2001) that were collected from the Santa Ana River drainage in San Bernardino County from 1905 to 1923; however, these specimens have more recently been identified as mountain yellow-legged frog (*R. muscosa*) (UCMVZ 2015). The University of Michigan Museum of Zoology (UMMZ 2001) has 12 frogs labeled *R. boylei* that were collected 2 miles from Santa Ana Canyon on a single day in August 1940. Santa Ana Canyon (or Santa Ana Narrows) is the water gap where the Santa Ana River passes between the Santa Ana Mountains and the Chino Hills, near the intersection of Orange, Riverside, and San Bernardino Counties, at less than 700 feet elevation. If these frogs were indeed taken in San Bernardino County, 2 miles from the Canyon, this would be the vicinity of Chino Hills State Park, in the lower Santa Ana River, below the known elevational range for *R. muscosa*.

Recent status: There are no recent records of foothill yellow-legged frogs from San Bernardino County. If the species did occur in San Bernardino County historically, it certainly has been extirpated.

Los Angeles County

There are historical collection records of foothill yellow-legged frogs from 1903 to 1977 in Los Angeles County, throughout the southern foothills of the San Gabriel Mountains in the San Gabriel River drainage, in Little Rock Creek in the northern San Gabriel Mountains, in a few scattered localities in the floodplain of what is now urbanized Los Angeles, and in extreme northwestern Los Angeles County in Piru Creek and Castaic Creek, tributaries to the Santa Clara River.

The mountain yellow-legged frog (*R. muscosa*) was known to occur historically in the San Gabriel Mountains of Los Angeles County (Vredenburg et al. 2007). Museum specimens collected from the San Gabriel Mountains are currently being evaluated by the USGS and the Los Angeles County Museum herpetological curator, as to whether they were actually *R. boylei* or *R. muscosa*.

San Gabriel River/Tujunga Creek

There are historical collection records labeled *R. boylei* from 1908-1952 in the San Gabriel Mountains, in streams draining into urbanized Los Angeles (CAS 2001; UMMZ 2001; UCMVZ 2015).¹ The West Fork, North Fork and East Fork of the San Gabriel River are all above San Gabriel Reservoir, which is at 1,145 ft (349 m). The U.C.

¹ In the San Gabriel River drainage – an unidentified location in the San Gabriel Mountains in 1915, the East Fork in 1935, unidentified location on the San Gabriel River in 1940, San Gabriel River above the junction of the North and West Forks in 1940, Crystal Lake Park (up a tributary to the North Fork) in 1946, the West Fork in 1950 (30 frogs from N of Camp Rincon on a single day in June) and 1951, the junction of the North and West Forks in 1951, the North Fork in 1951, and unidentified location on the San Gabriel River in 1951; from the Santa Anita Wash drainage - Big Santa Anita Canyon near Sierra Madre in 1908, Little Santa Anita Canyon in 1909, near Mt. Wilson in 1913, Little Santa Anita Canyon in 1918, and 8 frogs from near Mt. Wilson in October 1944; from the Tujunga Creek drainage - Mill Creek, near Big Tujunga Creek in 1952 (CAS 2001; UMMZ 2001; UCMVZ 2015).

Museum of Vertebrate Zoology currently list 59 *R. boylei* specimens collected from the San Gabriel Mountains in Los Angeles County from 1944 through 1951 (UCMVZ 2015). All museum specimens collected from the San Gabriel Mountains should be evaluated as to whether they were actually *R. boylei* or *R. muscosa*. Dunn et al. (1988) presumed *R. boylei* persisted in the San Gabriel Mountains, but there were no recent documented occurrences.

Los Angeles Floodplain

There are historical collection records labeled *R. boylei* from 1907-1930 in the greater Los Angeles floodplain, at low elevation (HMCZ 2001; CAS 2001; Cornell University 2002).² The U.C. Museum of Vertebrate Zoology had a yellow-legged frog specimen originally labeled as *R. boylei*, collected from Arroyo Seco Canyon near Pasadena in 1903 (UCMVZ 2001); this specimen is now labeled as *R. muscosa* (UCMVZ 2015). However, this locale is at low elevation within the floodplain of urban Los Angeles, and well below the known elevational range for *R. muscosa* in southern California; for which the lowest known occurrence is above 1,220 feet (USFWS 2002).

Santa Clara River

There are historical collection records labeled *R. boylei* and reports of observations from 1967-1977 in Piru Creek in the Santa Clara River drainage (Jennings and Hayes 1994; Stephenson and Calcarone 1999; LSUMNS 2001).³ The last reliable observation of *R. boylei* in this region was from Piru Creek in 1977 (Jennings and Hayes 1994).

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at any of 3 historical locations in Los Angeles County. There are no records of foothill yellow-legged frogs from Los Angeles County since 1977. The species is extirpated from Los Angeles County.

Ventura County

Santa Clara River

There are historical collection records of foothill yellow-legged frogs from 1914 to 1970 in Ventura County, all within the Santa Clara River drainage and its tributaries, including Piru Creek, Hopper Canyon, Sespe Creek, and Santa Paula Creek (Jennings and Hayes 1994; Stephenson and Calcarone 1999; UMMZ 2001; CAS 2001; CNDDB 2016; UCMVZ 2015).⁴ The species was apparently relatively common in Hopper Creek

² Claremont in 1907; the city of Los Angeles in 1911; La Crescenta in 1915; and Monrovia in 1930 (HMCZ 2001; CAS 2001; Cornell University 2002).

³ Specimens were taken from Elizabeth Lake Canyon (Stephenson and Calcarone 1999), Piru Creek south-southeast of Caswell in 1967 (LSUMNS 2001) and upstream from Piru Gorge (now flooded under Piru Lake) in 1970 (Jennings and Hayes 1994). The last reliable observation of the species in this region was 1-2 km south of Frenchman's Flat along Piru Creek in July 1977 (Jennings and Hayes 1994).

⁴ Collections records include: 1 frog from Sespe Canyon, at the head of Sespe Creek on the Ventura/S Barbara border in 1914 (CAS #39253); 1 frog (CAS #10223) from Sespe Gorge in 1948; 1 frog (CAS #10229) in 1949 and 1 larva (CAS #11549) in 1950 from the junction of Lion Canyon and Sespe Creek; an unknown location along Sespe Creek in 1940; 5 frogs and 1 larva (CAS #10224-10228, 11550) from Piru Creek in 1949; 11 adults and juveniles (MVZ #33664-33673; 54519) from Hopper Creek in 1940; 2 juveniles

(possibly named after the species), where 11 adult/juvenile frogs were collected on a single day in May 1940, and in Piru Creek, where 5 frogs and larva were collected in May 1949 (CAS 2001; UCMVZ 2015). The last known specimens in Ventura County were collected from Piru Creek, 10 miles N of the Temescal Ranger Station, in 1970 (Jennings and Hayes 1994).

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at any of 11 historical locations in Ventura County. There no reports of foothill yellow-legged frogs from Ventura County since 1970 (Jennings and Hayes 1994). The species is extirpated from Ventura County.

Santa Barbara County

Santa Ynez River

There are historical collections from 1933-1966 in the upper Santa Ynez River drainage and tributaries including Big Canyon Creek, Indian Creek, Juncal Creek, and Santa Cruz Creek (Jennings and Hayes 1994; Stephenson and Calcarone 1999; SDNHM 2001; UMMZ 2001; CAS 2001; CNDDDB 2016; UCMVZ 2015).⁵ All of the historical collections were of small numbers of frogs or individual frogs.

There are no recent records from the Santa Ynez River drainage.

Small coastal streams

There are historical collections records from Gaviota Creek in 1922 and Refugio Creek in 1974, small coastal streams along the southern coast of Santa Barbara County, tributary to the ocean (SBMNH 2001; UMMZ 2001; CNDDDB 2016).⁶

There are no recent records from southern coastal Santa Barbara County.

Recent status: Jennings and Hayes (1994) were unable to locate any foothill yellow-legged frogs during resurvey efforts from 1988-1991 at any of 7 historical locations in Santa Barbara County. There are no foothill yellow-legged frog records from Santa Barbara County since 1974 in the California Natural Diversity Database (CNDDDB 2016), the last record of foothill yellow-legged frog from Santa Barbara. The species is extirpated from Santa Barbara County.

and 1 adult (CAS #50470-50472) from Sespe Creek 3 miles N of Fillmore in 1921; 1 frog (MVZ #42611) from Santa Paula Canyon, near East Fork in 1946 and 4 frogs (ASUHERPS #4431-4434) from the same location in 1962; Santa Paula Creek in 1942; and 6.4 miles NE of Ventura in 1940 (UMMZ 2001; CAS 2001; CNDDDB 2016; UCMVZ 2015).

⁵ Historical collection records include: single frogs collected from the Santa Ynez River at Juncal Campground in 1961 (CAS #181275) and Big Canyon Creek at Juncal Road in 1966 (CAS #181276); Indian Creek at the base of Mono Dam in 1940; 1 frog (MVZ #35222) from Indian Canyon, 2 miles southeast of Bluff Camp in 1940; 2 adults, 2 juveniles and 2 frogs of unknown age (CAS #181269-181274) from Santa Cruz Creek near Santa Cruz Guard Station in 1960; and 1 frog (SDNHM #20776) from Bear Canyon on the north slope of Santa Ynez Mountain SE of Lake Cachuma in 1933 (SDNHM 2001; UMMZ 2001; CAS 2001; CNDDDB 2016; UCMVZ 2015).

⁶ Three yellow-legged frogs were taken from Gaviota Creek in 1922 (UMMZ 2001); and 1 frog (SBMNH #HE 243) was collected from Refugio Creek in 1974 (SBMNH 2001; CNDDDB 2016).

South Coast

The South Coast includes coastal drainages south and west of the Santa Lucia Range in Monterey County, south to San Luis Obispo County. The foothill yellow-legged frog is now nearly extirpated from Monterey and San Luis Obispo counties and the south coast region.

San Luis Obispo County

Cuyama River

There is a foothill yellow-legged frog collection record from the Alamo Creek tributary of the Cuyama River in 1940. The Cuyama River joins the Santa Maria River in southern San Luis Obispo County.

Arroyo Grande Creek

There are historical collection records from 1943-1963 in Arroyo Grande Creek (SBMNH 2001; CNDDDB 2016; UCMVZ 2015).⁷

San Luis Obispo Creek

There are historical collection records from 1939 and 1953 in San Luis Obispo Creek (UCMVZ 2015).⁸

The species was presumed to occur in Poly Canyon in the Brizzolari Creek watershed, a tributary of Stenner Creek and San Luis Obispo Creek, on land owned by Cal Poly University (Cal Poly 2001), but there were no documented records.

Northern coastal streams

There are historical collection records from northern coastal San Luis Obispo County streams: San Carpoforo Creek in 1940 and Santa Rosa Creek in 1948 (CAS 2001; CNDDDB 2016).⁹

The species persisted in several coastal drainages in very northern San Luis Obispo County in the 1990s (Jennings and Hayes 1994; Stephenson and Calcarone 1999). As discussed above, small numbers of frogs were located in San Carpoforo Creek in northern San Luis Obispo/southern Monterey Counties in 1999 (LPNF 2001), and a single adult frog (deposited at UCSB) was captured from Little Pico Creek (near Highway 1, just ESE of San Simeon) in April 1999 (CNDDDB 2016).

⁷ A single frog from Arroyo Grande Creek in 1943; 1 frog (MVZ #58422) from Lopez Canyon in Arroyo Grande Creek, 6 miles E of San Luis Obispo, in 1953; and 5 frogs (SBMNH #HE 136-140) from Lopez Canyon in 1963 (SBMNH 2001; CNDDDB 2016; UCMVZ 2015).

⁸ There are collection records of single frogs from Reservoir Canyon, in San Luis Obispo Creek, 2 miles E of San Luis Obispo, in 1939 (MVZ 31615) and 1953 (MVZ 59660) (UCMVZ 2015).

⁹ Six frogs (CAS #159503-159508, and larvae) were collected from Santa Rosa Creek near Cambria in September 1948; and single frogs in 1940 from the lagoon of San Carpoforo Creek, and 2 miles from Santa Ana Canyon (CAS 2001; CNDDDB 2016).

Recent status: Jennings and Hayes (1994) were able to relocate the species during resurvey efforts from 1988-1991 at only 3 of 11 historical locations (27%) in San Luis Obispo County. There have been no documented foothill yellow-legged frog observations in San Luis Obispo County since 1999. The foothill yellow-legged frog is now nearly extirpated from San Luis Obispo County.

Monterey County

Malpaso Creek

Two frogs were collected from Malpaso Creek, north of Garrapata State Park, in 1915 (CAS 2001).

Bixby Creek

There is a historical collection record of 1 frog from the Turner Creek tributary near Devils Peak in 1939 (UCMVZ 2015).

Little Sur River

There are historical collection records from 1933-1948 in the Little Sur River (CAS 2001; UCMVZ 2015).¹⁰

Big Sur River

There are historical collection records from 1929-1946 in the Big Sur River (CAS 2001; UCMVZ 2015).¹¹

Foothill yellow-legged frogs still occurred in the 1990s in the Big Sur River (Jennings and Hayes 1994; Stephenson and Calcarone 1999).

Big Creek

Frogs were not located during surveys in the 1970s of the Landels-Hill Big Creek Reserve, in the Big Creek coastal drainage (Carothers et al. 1980).

Willow Creek

A collection was made at the mouth of Willow Creek in 1948 (CAS 2001).

Foothill yellow-legged frogs still occurred in the 1990s in Willow Creek (Jennings and Hayes 1994; Stephenson and Calcarone 1999).

¹⁰ Two frogs from Pine Creek (at the Little Sur River) in 1933; 1 frog from Skinner Creek near Devils Peak in 1939; and 1 frog from the mouth of the Little Sur River in 1948 (CAS 2001; UCMVZ 2015).

¹¹ Collected at Idyllwild Park from 1929 to 1930; 6.5 miles above the mouth in 1937; and Big Sur State Park in 1946 (CAS 2001; UCMVZ 2015).

Dutra Creek/San Carpoforo Creek

Small numbers of frogs were documented from 1995-1999 in Dutra Creek and San Carpoforo Creek, within Monterey County (CNDDDB 2016).¹² However, Kupferberg and Adams searched several reaches of San Carpoforo Creek on public lands (USFS lands in the upper watershed and BLM lands in the lower watershed), and the Dutra Creek tributary (where A. Lind had previously collected samples) in July 2014, and did not locate any foothill yellow-legged frogs (S. Kupferberg, pers. comm., 2015).

Recent status: Unknown, but likely near extirpation. Foothill yellow-legged frogs still occurred in the 1990s in several coastal drainages in Monterey County but there are no foothill yellow-legged frog records from coastal drainages in Monterey County south and west of the Santa Lucia Range since 1999.

Central Coast

The species appears to be extirpated from western San Joaquin County and may be near extirpation in western Merced, Santa Cruz and San Mateo counties. The species has declined in many of the drainages in western Stanislaus, western Fresno and Monterey counties. There appear to be significant populations of foothill yellow-legged frogs remaining in the Diablo Range in San Benito County.

Monterey County

Salinas River

There are historical collection records from 1919-1955 in the Salinas River drainage, including Arroyo Seco, Nacimiento Creek, Reliz Canyon, Salinas River and Santa Lucia Creek (Zweifel 1955; CAS 2001; UCMVZ 2015).¹³

Small populations were observed in the 1990s along the Monterey/San Benito County line in Lewis Creek, tributary to San Lorenzo Creek, a northern tributary to the Salinas River, and in the Nacimiento River at Fort Hunter Liggett (CAS 2001; CNDDDB 2016).¹⁴

¹² Two adult frogs were observed July 24, 1995 along Dutra Creek, 1 adult and 30 larvae were seen at this same spot on July 25, 1995 (Fellers site S-734B); a single adult was observed along San Carpoforo Creek about 0.4 mile NE of the Dutra Creek confluence on July 25, 1995; and an unknown number of tadpoles were observed and 5 tadpoles were collected (deposited in UCSB collection) on August 20 and 26, 1999 from the Dutra Creek tributary (CNDDDB 2016). The Los Padres National Forest noted that foothill yellow-legged frogs were located in San Carpoforo Creek in northern San Luis Obispo/southern Monterey Counties in 1999 (LPNF 2001).

¹³ Frogs were collected in the Salinas River drainage from Lopez Canyon (Zweifel 1955). Frogs were collected in the Arroyo Seco tributary drainage from: Abbotts Ranch in 1919; Reliz Canyon, W of Greenfields, in 1938; W of Tassajara Springs in the Tassajara Creek tributary in 1940; and Camp Calatraz, in the Santa Lucia Creek tributary, in 1951 (CAS 2001; UCMVZ 2015). There are also historical collection records from the upper Salinas River drainage within San Luis Obispo County: 1 frog (CAS #43336) from Santa Margarita in 1917; and 6 frogs from the Nacimiento Creek tributary in 1909 (CAS 2001).

¹⁴ In the Lewis Creek tributary, 1 subadult frog was collected (CAS #195440) in February 1994 from Priest Valley just W of the Fresno County line; 3 adults and 7 juveniles were observed in August 2001 7 miles SW of Hernandez Reservoir; and 7 juveniles were observed on April 1, 2002 1.25 miles SE of the Yaqui Creek confluence (CAS 2001; CNDDDB 2016). These populations were subject to livestock grazing. There is one CNDDDB record from the 1990s from the Nacimiento River tributary of the Salinas River drainage: 10 adults, 10 metamorphs, and an unspecified number of tadpoles observed in August 1999 in Los Burros Creek, upstream from the confluence with Little Burros Creek, at Fort Hunter Liggett (CNDDDB 2016).

Carmel River

There are historical collections records before 1900-1975 in the Carmel River drainage (CAS 2001; FMNH 2001; UCMVZ 2015).¹⁵ The species was apparently once relatively common in San Clemente Creek, where 11 frogs were collected on a single day in August 1939 (UCMVZ 2015).

Foothill yellow-legged frogs reportedly persist at the Hastings Reserve in the Finch Creek tributary (UCNRS 2015), but there are no known documented sightings.

Santa Lucia Range

There are historical collection records from the Santa Lucia Range in 1902, from Santa Lucia Peak and Cone Peak (UCMVZ 2015).

Recent status: Jennings and Hayes (1994) were able to relocate the species during resurvey efforts from 1988-1991 at only 5 of 12 (42%) historical locations in Monterey County. The species has clearly declined in the Carmel River drainage. There were small populations documented in Salinas River tributaries in the late 1990s and early 2000s. There are no foothill yellow-legged frog records from Monterey County in the California Natural Diversity Database since 2002 (CNDDB 2016).

San Benito County

There are numerous historical collection records for foothill yellow-legged frogs from Pinnacles National Monument in Chalone Creek and other northern tributaries to the Salinas River; from the headwaters and upper reaches of the San Benito River drainage, which flows NW to the Pajaro River; and from Panoche Creek, which flows E into the San Joaquin Valley.

Salinas River/Pinnacles NM

There are historical collection records from Pinnacles National Monument from 1918-1938 (Banta and Morafka 1967; SDNHM 2001; CAS 2001; UCMVZ 2015).¹⁶ Foothill yellow-legged frogs were apparently common historically in Pinnacles, with large numbers of specimens taken from collection sites on a single day: 14 frogs from Clear Creek in May 1960; and 11 frogs from Pinnacles in July 1938 (CAS 2001; SDNHM 2001; UCMVZ 2015). *Rana boylei* was still “abundant” in Pinnacles National Monument in 1953 (Banta and Morafka 1967) and “quite common” along streams and waterways

¹⁵ Collection localities include: Pacific Grove before 1900; Pine Valley, at the head of the Carmel River in 1902; the Carmel River about 5 miles from the mouth in 1904; San Jose Creek near Carmel in 1907; the San Clemente Creek tributary in 1939; and Carmel Valley above P.I. County Ranch, unknown date (CAS 2001; FMNH 2001; UCMVZ 2015). A tributary to Tularcitos Creek in the Carmel River Valley named Rana Creek, likely had ranid frogs in it, based on the name. Single frogs were collected from Big Creek (MVZ 134089) in 1974 and Blomquist Pond (MVZ 134090) in 1975, tributaries to Cachuga Creek (an upper tributary of the Carmel River) in the Hastings Natural History State Reserve.

¹⁶ Seven frogs taken from 7 miles SW of the Cook Post Office in July 1918; one frog in August 1918; 3 frogs from Vancouver, Pinnacles in August 1918; 11 frogs taken on a single day in July 1938; and historical specimens in the Stanford University collection taken from along Chalone Creek (Banta and Morafka 1967; SDNHM 2001; CAS 2001; UCMVZ 2015).

throughout the Monument in the late 1950s (Wauer 1958). The species was present in the mid 1960s (De Foe 1963; Morafka 1965), but was not observed during sampling by Banta and Morafka (1967). By the mid 1980s *R. boylei* was considered “rare” in the Monument, found only in the vicinity of flowing streams such as Chalone Creek (Fellers 1986).

Intensive amphibian surveys by Ely (1993, 1994) during 1992-1994 covered stream reaches with suitable *R. boylei* habitat within Pinnacles National Monument, including Chalone Creek and tributaries such as West Fork Chalone Creek, Bear Creek, and Frog Canyon Creek. *Rana boylei* could not be found, even at historical locations such as Pinnacles Caves. Ely (1993, 1994) documented extensive damage to riparian vegetation, stream structure, and shorelines within the Monument by feral pigs, cited likely predation of frog larvae and eggs by introduced mosquito fish (*Gambusia affinis*) and Sacramento perch (*Archoplites interruptus*) in Chalone and Bear Creeks, and presumed that Bear Gulch Reservoir may have eliminated occupied *R. boylei* habitat in Bear Creek. *Rana boylei* is now considered extirpated from Pinnacles National Monument (Fesnock and Johnson 2002; NPS 2015).

San Benito River

There are historical collection records from 1936-1986 in the San Benito River drainage, including Clear Creek, Laguna Creek, San Benito River and Tres Pinos Creek (CAS 2001; UCMVZ 2015).¹⁷

“Large” populations of *R. boylei* were observed on Bureau of Land Management land in the upper San Benito River watershed during surveys in 1992 (Ely 1992). The species was found in “moderate to good numbers” in every creek system surveyed upstream from Hernandez Reservoir; in several sections of the San Benito River and in tributaries such as Clear, Pacheco, San Carlos, Sawmill, and White Creeks (Ely 1992, 1993; CNDDDB 2016). Pacheco Creek had an “excellent” population, with tadpoles found in “exceptional” numbers in July 1992 (Ely 1992). *Rana boylei* were also found in the San Benito River downstream of Hernandez Reservoir to Long Canyon and a very large population with “very impressive numbers” (112 frogs seen in one visit) was noted in the Laguna Creek tributary to the reservoir. This high density of frogs was found in a gorge inaccessible to livestock. Ely (1992) thought that Laguna and Arroyo Leona Creeks had perhaps the greatest population densities of the species remaining in the entire south coast range. Ely (1992) noted impacts to frog habitat and threats to the species on these BLM lands and nearby private lands from heavy off-road vehicle use, excessive cattle grazing, and impacts from past mining. Ely (1992) noted that water releases from Hernandez Reservoir in the summer appear to have adversely impacted frog reproduction in the San Benito River downstream of the dam, and presumed the river

¹⁷ Eight frogs collected from 1 mile SE of the summit of San Benito Mt. in 1936 and 4 frogs and tadpoles in 1944; 15 frogs from 1 mile S of the summit of Santa Rita Peak in 1936; 3 frogs from Horsethief Canyon, 3.5 miles SSE of Hernandez; single frogs from Laguna Ranch, 4 miles S of Hernandez, and San Benito River, 2 miles ESE of Hernandez, in 1936; 14 frogs from Clear Creek, 15 miles SW of New Idria and 2 miles SE of Hernandez, in 1960; 1 frog from Laguna Creek at Coalinga Road in 1986; 1 frog from San Benito Road, 7.5 miles S of Bitterwater Road in 1948; 6 frogs from San Benito River, 9 miles N of Pinnacles in 1951; 6 frogs from Tres Pinos Creek, 15 miles ESE of Paicines in 1953; 3 frogs from San Benito River, 10 miles SSE of Paicines in 1953; and 6 frogs from near the San Benito River Bridge on Hwy. 25 in 1968 (CAS 2001; UCMVZ 2015).

section probably supported much larger numbers of the species before the reservoir was built.

Foothill yellow-legged frogs remained locally abundant through 2009 in some of the streams in the Clear Creek Management Area administered by the U.S. Bureau of Land Management, including San Carlos Creek, Clear Creek, and Sawmill Creek (USBLM 2009, 2013). During 13 surveys from 1996 to 2008 along Clear Creek, about 0.7 miles upstream from the San Benito River confluence, Fellers observed an annual average of 17 adults, 143 juveniles and 131 egg masses (CNDDDB 2016). Fellers documented modest numbers of adults and juveniles in San Carlos Creek in 1993, 1996-2000, and in 2009 (CNDDDB 2016).

Panoche Creek

Twenty-one frogs were collected from Panoche Creek (2 miles SE of Panoche) in July 1936 (CAS 2001; UCMVZ 2015).

Small to moderate populations were documented in the early 1990s in the Silver Creek drainage, which flows into Panoche Creek: in the tributaries Larious Creek, Sampson Creek, San Carlos Creek and White Creek (CAS 2001; CNDDDB 2016). There are no reports of *R. boylei* from upper Panoche Creek tributaries in the last two decades.

Recent status: Fellers (1994) reported healthy, reproducing populations in the Diablo Range in San Benito County. Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 3 of 11 historical locations (27%) in San Benito County. The species is extirpated from Pinnacles and the Salinas River drainage in San Benito County. Significant populations were documented in the early 1990s in the San Benito River and tributaries above Hernandez Reservoir, particularly in Clear Creek, Laguna Creek and Pacheco Creek; and also in the Silver Creek drainage. The species remained locally abundant through 2009 in San Benito River tributary streams in the BLM's Clear Creek Management Area.

Santa Cruz County

San Lorenzo River

There are historical collection records from 1891-1967 in the San Lorenzo River drainage and tributaries Boulder Creek and Bear Creek, in the Santa Cruz Mountains (Slevin 1928; LSUMNS 2001; CAS 2001; UCMVZ 2015).¹⁸

The species was "virtually extinct" in the Santa Cruz Mountains by the 1990s, with significant impacts from logging (R. Seymour and M. Westphal, pers. comm., 1996). There are no known recent occurrences at Henry Cowell State Park, but there have not been extensive surveys for the species (G. Gray, CA State Parks, pers. comm., 2001).

¹⁸ Collection localities include: Glenwood in 1891; Boulder Creek in 1892, before 1928, and 1939; near Zayante in the early 1930s; 1 mile N of the town of Boulder Creek in 1941; 3 miles NE of Boulder Creek in 1953; Cave Gulch in 1959; Bear Creek 3 miles NE of Boulder Creek in 1966; and Bear Creek, 4.4 miles E of Boulder Creek in 1967 (Slevin 1928; LSUMNS 2001; CAS 2001; UCMVZ 2015).

Pajaro River, Watsonville Slough

There are historical collection records from 1928 in Corralitos Creek, tributary to the Pajaro River, and 1970 in Harkins Slough, tributary of Watsonville Slough (UMMZ 2001; HMCZ 2001; UCMVZ 2015).¹⁹

Aptos Creek

Small numbers of frogs were reported from the Aptos Creek watershed within the Forest of Nisene Marks in 1998 (CNDDDB 2016).²⁰

Soquel Creek

Small to moderate populations were reported from 1992-2008 in the Soquel Creek drainage (CNDDDB 2016).²¹ There are no known recent occurrences in Big Basin Redwoods State Park, but there have not been extensive surveys for the species (G. Gray, CA State Parks, pers. comm., 2001).

Waddell Creek

There are historical collection records from 1892-1953 in Waddell Creek (UCMVZ 2015).²²

There are no known recent observations from Waddell Creek.

Recent status: Jennings and Hayes (1994) were able to relocate the species during resurvey efforts from 1988-1991 at 3 of 4 historical locations (75%) in Santa Cruz County. The species appears to be extirpated from much of Santa Cruz County, including the San Lorenzo River drainage, Pajaro River and Waddell Creek. Small to moderate populations appeared to persist in the Soquel Creek drainage through 2008.

San Mateo County

Pescadero Creek

There are historical collection records from 1937-1960 in the Pescadero Creek drainage (CAS 2001; UCMVZ 2015).²³

¹⁹ One frog collected from 3 miles N of Corralitos, in Corralitos Creek, tributary to the Pajaro River, in 1928; and 1 frog (MVZ 164868) collected by S. Sweet from Harkins Slough (tributary of Watsonville Slough) on October 23, 1970 (UMMZ 2001; HMCZ 2001; UCMVZ 2015).

²⁰ Three juveniles observed in Aptos Creek 0.6 mile downstream of Five Finger Falls and 1 individual in Bridge Creek, 0.5 mile upstream of the Aptos Creek confluence, in November 1998 (CNDDDB 2016).

²¹ Surveys of about 9 river miles of Soquel Creek upstream from Soquel and of about 1.25 miles of tributary Hinckley Creek upstream from the Soquel Creek confluence documented 1 adult in 1992; 2 adults in 1993; 20+ juveniles on 10/24/94; 2 adults, 25 juveniles and 11 egg masses on 4/27/96; 1 adult and 1 egg mass on 4/24/97; 3 adults on 10/24/97; about 20-30 adults and juveniles on 10/25/97; less than 50 frogs in October 1999; less than 50 frogs in October 2001; an unknown number of adults and juveniles in September 2002; "dozens" of frogs September-October 2003; and 3 juveniles in October 2008 (CNDDDB 2016).

²² Ten frogs collected from Waddell Creek in 1892; 3 frogs taken 2 to 5 miles from Ocean Beach in 1938; 5 frogs in 1941; and 2 frogs taken 2 miles from the mouth in 1953 (UCMVZ 2015).

²³ Ten frogs collected from 1 mile NE of Loma Mar in 1937; 4 frogs from the upper reaches of the tributary

Other than a single record from 1999 in Pescadero Creek County Park²⁴, there are no known recent observations from Pescadero Creek.

San Gregorio Creek

Historical collection records include 1 frog from La Honda in 1929 and 3 frogs from SW of La Honda in 1951 (CAS 2001; UCMVZ 2015).

There are no known recent observations from San Gregorio Creek.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 4 of 9 historical locations (44%) in San Mateo County. There are no CNDDDB records from San Mateo County since 1999 (CNDDDB 2016). The species is likely close to extirpation in San Mateo County.

Western Fresno County

Los Gatos Creek

There are historical collection records from 1938-1963 in Los Gatos Creek, which flows eastward to the San Joaquin Valley, and tributary Warthan Creek (Ely 1992; CAS 2001; UMMZ 2001; UCMVZ 2015).²⁵

No frogs were located in Warthan Creek during early 1990s surveys by Ely (1992). Small numbers of frogs were documented from 1992-1993 in Los Gatos Creek, the White Creek tributary on BLM land, and Warthan Creek (CAS 2001; CNDDDB 2016).²⁶ Threats to these frogs included legacy impacts from past heavy mining. There are no known observations in the Los Gatos Creek drainage since 1993.

Cantua Creek

During surveys of the east slope of the Diablo Range in the early 1990s, foothill yellow-legged frogs were found “in abundance” (more than 27 frogs) in Cantua Creek, with a “very large population” estimated at more than 88 frogs observed in the tributary Arroyo Leona Creek, in the Ciervo Hills on Bureau of Land Management and private lands (Ely 1992). A large population was documented consistently from 1992-1994 in Cantua

Oil Creek in 1937; 3 frogs from Portola State Park in 1951, 1 frog in 1958, 1 frog in 1959, and 2 frogs in 1960; and 2 frogs taken from 8 miles E of Pescadero in 1960 (CAS 2001; UCMVZ 2015).

²⁴ A single adult frog was observed in Pescadero Creek, between Jones Gulch and Harwood Creek in Pescadero Creek County Park, in September 1999 (CNDDDB 2016).

²⁵ Single frogs from Los Gatos Creek, 10 miles N of Coalinga in 1938; 7.9 miles SE of the mouth of Bear Canyon in 1941; 2 miles W of Coalinga in 1941; and 6 frogs collected from Los Gatos Creek, 8 miles NW of Coalinga in 1963 (CAS 2001; UMMZ 2001; UCMVZ 2015). There were a few historical records from Warthan Creek, a tributary of Los Gatos Creek near Coalinga (Ely 1992).

²⁶ In the White Creek tributary, 4 miles SSE of Santa Rita Peak, 7-10 tadpoles were observed on 6/28/92, 4 adults and 2 juveniles on 3/6/93, with one adult (CAS #190766) collected, and 4 adults and larvae on 6/29/93; a single adult frog (CAS #193924) was collected from Los Gatos Creek in the Alcalde Hills in October 1993 (CAS 2001; CNDDDB 2016). A single adult frog (CAS #193923) was collected from Warthan Creek in October 1993, 1.2 road miles E of Parkfield Road (CAS 2001; CNDDDB 2016).

Creek (CAS 2001; CNDDDB 2016).²⁷ In the 2000s, there was only one report of a small population of frogs at one locality in Cantua Creek (CNDDDB 2016).²⁸

Recent status: Fellers (1994) reported healthy, reproducing populations in western Fresno County. Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at any of 6 historical locations in western Fresno County. Small populations were documented in Los Gatos Creek and large numbers in Cantua Creek during surveys from 1992-1993, but there has been only report since 1993 and no evidence of resurveys.

Western Merced County

Los Banos Creek

There are historical collection records from 1911 and 1943 in Los Banos Creek (which flows west into the San Joaquin Valley) and tributary Romero Creek (UCMVZ 2015).²⁹ There were small populations documented from 1985-1988 in Los Banos Creek and North Fork Los Banos (CNDDDB 2016).³⁰

There are no known recent observations from Los Banos Creek.

Recent status: There are no CNDDDB records from western Merced County since 1988 and the status of the species in this county is unknown.

Western Stanislaus County

Del Puerto Creek

There are historical collection records from 1954-1987 in Del Puerto Creek, which flows east into the San Joaquin Valley (CAS 2001; UCMVZ 2015).³¹

Small to moderate populations were documented in Del Puerto Creek from 1993-1999

²⁷ Sample observations from this drainage by Ely, Christman, and Fellers submitted to the California Natural Diversity Database and collected by the California Academy of Sciences in the early 1990s included: 115+ frogs observed on 1/25/92; 6 adults and 2 juveniles on 4/26/92; 102 frogs, juveniles and 2 egg masses/tadpoles on 4/27/92; 3 adults and "impressive" numbers of tadpoles on 6/14/92; 20 adults and 15 egg masses, with 200-300 eggs/mass on 4/15/93; 20 adults, 2 juveniles and 15 egg masses on 4/25/93; 7 adults and one egg mass on 4/26/93; and 12 adults, 47 juveniles, 3,183 larvae and 5 egg masses in April 1994 (CAS 2001; CNDDDB 2016).

²⁸ In May 2003 one location in the Cantua Creek drainage had 10 adults, 5 juveniles and 42 larvae (CNDDDB 2016).

²⁹ Single frogs collected at Sweeney's Ranch, 22 miles S of Los Banos, in 1911; and in Romero Creek in 1943 (UCMVZ 2015).

³⁰ A single adult on the North Fork Los Banos Creek, 3 miles SW of San Luis Reservoir on 8/2/85; 10 adults at the Los Banos Detention Dam on the S end of Los Banos Reservoir on 8/19/85; 2 adults in North Fork Los Banos Creek, 2 miles SSW of San Luis Reservoir on 9/20/85; several frog observations along a 1-mile section of creek approximately 7 miles ESE of San Luis Reservoir on 9/30/85; and a single adult observed in Los Banos Creek ¾ mile SSE of San Luis Reservoir on 3/21/88 (CNDDDB 2016).

³¹ Single frogs were collected from Del Puerto Canyon, west of Patterson, in 1954, 1984 and 1986 and 7 frogs were collected in 1987 (CAS 2001; UCMVZ 2015).

(CAS 2001; CNDDDB 2016; UCMVZ 2015).³² Very small populations were documented in Del Puerto Creek from 2000-2008 (CNDDDB 2016).³³

Orestimba Creek

Small numbers of foothill yellow-legged were documented in 2004 and 2005 within Henry Coe State Park, along Robison Creek near its confluence with Orestimba Creek (CNDDDB 2016; HWCSP 2015).³⁴

Recent status: Fellers (1994) reported “healthy” reproducing populations in western Stanislaus County. There appeared to be small populations remaining in Del Puerto Creek through 2008 and Orestimba Creek through 2005.

Western San Joaquin County

Corral Hollow Creek

There are historical collection records from 1911-1971 lower Corral Hollow Creek in San Joaquin County (CMNH 2001; CNDDDB 2016; UCMVZ 2015).³⁵ Lower Corral Hollow Creek apparently had a large population historically, with 17 adult specimens collected on a single day in March 1911 (UCMVZ 2015). The last known sightings in lower Corral Hollow Creek within San Joaquin County were in 1971, and the remnant population in upper Corral Hollow Creek in Alameda County may be near extirpation as well (Jones & Stokes 2000).

Mokelumne River/San Joaquin River

Livezey (1963) reported an isolated population of *R. boylii* on the floor of the Central Valley in San Joaquin County, approximately midway between the known distribution of the species in the coast ranges to the west and the Sierra foothills to east. Livezey collected 4 specimens (now at the California State University Sacramento) on July 30, 1958 from the Mokelumne River drainage 8 km N of Lodi. However, it has been theorized that these frogs were perhaps strays from the Sierran foothills or that waif

³² A single frog was collected from the Adobe Creek tributary at the confluence with Del Puerto Creek in June 1994 (CAS 2001; UCMVZ 2015). Fellers observed 2 adults and 800 eggs on 4/9/93 in Del Puerto Creek at Adobe Creek and returned to collect 1 adult (CAS #196026) on 6/19/94; Fellers observed 14 larvae on 5/21/96 along lower North Fork Del Puerto Creek; multiple sightings about 1 mile S of the confluence with the North Fork, with 35 adults observed on 10/19/93, 6 adults and 2 metamorphs on 7/13/98 and 9/21/98, and 3 adults, 7 metamorphs and many larvae on 7/23/99. Along Del Puerto Creek about 0.6 miles W of Arkansas Canyon, Fellers observed 9 larvae on 6/2/98, 34 adults and 11 subadults on 5/25/99 08 (CNDDDB 2016).

³³ Fellers observed 2 subadults on 6/23/00 along Del Puerto Creek, just W of Slick Rock Canyon confluence; Fellers observed 1 adult and 1 larvae on 5/28/03 in Del Puerto Creek near Frank Raines Regional Park (this is near an off-road vehicle park); along Del Puerto Creek about 0.6 miles W of Arkansas Canyon, Fellers observed 1 adult, 1 subadult and 25 larvae on 7/23/00; and 49 larvae on 5/28/08 (CNDDDB 2016).

³⁴ Within the NE portion of Henry W. Coe State Park, 3 juveniles and 2 egg masses were observed in Robinson Creek on 3/20/04; and in South Fork Orestimba Creek, 1 adult and 2 juveniles were observed on 3/21/04 and 1 adult was observed on 5/15/05 (CNDDDB 2016).

³⁵ Foothill yellow-legged frogs were collected from lower Corral Hollow Creek in San Joaquin County, near Tracy, in 1911, 1943, 1967 and 1970 (CMNH 2001; UCMVZ 2015). The last known sightings in Corral Hollow Creek were in 1971, with 1 frog (MVZ 99240) collected on 4/24/71, 23 tadpoles (MVZ 98194) collected on 5/15/71 and 7 tadpoles (MVZ 98191) collected on 5/29/71 (CNDDDB 2016).

dispersal occurred in this case (Lind et al. 1996). The species was also found in a pond near the San Joaquin River, 18 miles W of Manteca, in 1960 (UCMVZ 2015).

Recent status: There are no recent reports from the CNDDDB or other sources of foothill yellow-legged frogs in western San Joaquin County. The species is likely completely extirpated from lower Corral Hollow Creek and western San Joaquin County.

Bay Area

The species is extirpated from western San Francisco County; and may be near extirpation in Contra Costa and San Mateo counties. The species has declined in many of the drainages in Napa, Marin, and Alameda counties; in Alameda County the largest populations have crashed recently. There appear to be significant populations of foothill yellow-legged frogs remaining in the Diablo Range in Santa Clara County and throughout Sonoma County.

San Francisco County

The California Academy of Sciences has a single collection record of *R. boylei* from before 1938 from an unknown location in San Francisco (CAS 2001).

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at the sole historical location in San Francisco County. There are no CNDDDB records and the foothill yellow-legged frog is extirpated from San Francisco County.

San Mateo County

SF Bay Tributaries

Two frogs were collected from the San Mateo Creek drainage (San Andreas Lake) before 1915; and 6 frogs were taken from Redwood City (likely Redwood Creek or Atherton Creek) in 1899 (Slevin 1928; CAS 2001; USNM 2001).

There are no known recent observations from San Francisco Bay tributaries in San Mateo County.

Santa Clara County

Historically, foothill yellow-legged frogs were probably present in virtually all of the larger perennial streams in Santa Clara County with the exceptions of the lower portions of Coyote Creek and the Guadalupe River (H.T. Harvey and Associates 1999).

Pajaro River

There are historical collection records in southern Santa Clara County from 1898-1965 in the Llagas Creek, Murphy Creek and Uvas Creek tributaries to the upper Pajaro River drainage (CAS 2001; UMMZ 2001; UCMVZ 2015).³⁶

³⁶ Single frogs collected from: Uvas Creek in 1909; Murphy Creek, tributary to Uvas Creek N of Mt. Madonna, in 1898; a tributary to Uvas Creek at Hecker Pass, 8 miles W of Gilroy in 1952; Llagas Creek in

Small populations were reported from 1998-2007 in Llagas Creek within Santa Clara County (CNDDDB 2016).³⁷

Mount Hamilton/Alameda Creek headwaters

There are historical collection records from 1900-1986 in headwater tributaries of the Alameda Creek drainage around Mt. Hamilton, including the upper reaches of Arroyo Mocho and Arroyo Valle, and Isabel and Smith Creeks and their tributaries above Calaveras Reservoir (UMMZ 2001; CAS 2001; Cornell 2002; UCMVZ 2015).³⁸

Small populations were documented from 1988-2004 in Mount Hamilton tributaries and the Alameda Creek headwaters, including upper Alameda Creek, Arroyo Hondo, Bonita Creek, Colorado Creek, Indian Creek, Smith Creek and Sulphur Creek (CNDDDB 2016).³⁹ A small population of foothill yellow-legged frogs was observed "several times" from 1997-2011 in Arroyo Hondo on the Blue Oak Ranch Reserve (CNDDDB 2016).⁴⁰

Coyote Creek

There are historical collection records from 1904-1975 in the Coyote Creek drainage and tributaries Fisher Creek and Upper Penetencia Creek (Slevin 1928; CDFG 1975; CAS 2001; UMMZ 2001; UCMVZ 2015).⁴¹ Foothill yellow-legged frogs were historically found

1922 and 1965; and Llagas Creek just above the head of Chesbro Reservoir in 1986 (UMMZ 2001; CAS 2001; UCMVZ 2015).

³⁷ In Llagas Creek, 2 adult frogs were reported 2 miles S of Calero Reservoir on 7/13/98, and 1 adult was observed here on 3/24/07; just above Chesbro Reservoir, 2 adults were observed on 3/22/03 and single adults were seen on 3/23/03, 3/30/03 and 4/11/03 (CNDDDB 2016).

³⁸ Historical collection localities include: unknown location on Mt. Hamilton in 1900; 12 frogs from 25.4 miles SE of Livermore (likely in the headwaters of Arroyo Mocho) in 1952; 2 frogs from Arroyo Mocho, 8.5 miles N of San Antonio in 1971; 2 juvenile/adult frogs from Blackbird Valley (tributary of upper Arroyo Valle) in 1986; 3 frogs from San Antonio Creek (tributary to upper Arroyo Valle), 10 miles from Mt. Hamilton, in 1939; 6 frogs from 2 miles N of the road from Lick Observatory (likely Smith Creek or Isabel Creek) in 1948; Isabel Creek near Mt. Hamilton in 1921; Smith Creek at Mt. Hamilton Road, unknown date; and Smith Creek Ranger Station, Mt. Hamilton Road, in 1950 (UMMZ 2001; CAS 2001; Cornell 2002; UCMVZ 2015).

³⁹ The species was observed in May 1990 in Upper Alameda Creek in Ohlone Regional Park, just within Santa Clara County (CNDDDB 2016). Numerous observations were made of all life stages of foothill yellow-legged frogs during surveys from March through July from 1988-91, in Smith Creek, Sulphur Creek and Indian Creek on the SW slopes of Mt. Hamilton within J.D. Grant County Park; more frogs were found upstream from Smith Creek at the confluence with Indian Creek, as well as the first 1/3 to 1/2 mile of Sulphur Creek in 1991; however in 1993 only 1 adult frog was found in this area, in Sulphur Creek (CNDDDB 2016). Single frogs were collected from Sulphur Creek, 2.5 miles upstream from the Smith Creek confluence in October 1992, and 0.2 miles upstream in May 1993 (CAS 2001). In Colorado Creek (tributary of upper Arroyo Valle), from Mines Road upstream into Blackbird Valley, Ely observed 1 adult on 4/20/93, and 17 adults and a "good number" of tadpoles upstream on 6/12/93; single adult frogs were collected from Colorado Creek by Ely on 11/3/92 (CAS #190556) and on 5/24/98 (CAS 205752) (CAS 2001; CNDDDB 2016). A single frog (CAS #195454) was collected from Indian Creek on 3/13/94 (CNDDDB 2016). The SFPUC reported 2 or more adults in Arroyo Hondo, just above Calaveras Reservoir, on 4/20/94 (CNDDDB 2016). 2 adult frogs were observed on 3/22/03 during a walking survey of a 1 mile section of Bonita Creek, a tributary of Isabel Creek (CNDDDB 2016).

⁴⁰ Arroyo Hondo on the Blue Oak Ranch Reserve (about 1.5 miles SSW of the Mt. Day Summit), with photos and a survey form indicating that 10 adults and 1 egg mass had been detected (CNDDDB 2016).

⁴¹ Historical collection localities include: Coyote Creek from before 1928; 6.54 road miles northeast of Gilroy (likely Coyote Creek near Anderson Lake) in 1953; 15 frogs from Coyote Creek in San Jose in 1922; Berryessa, tributary to Upper Penetencia Creek in 1904; Penetencia Creek, Alum Rock Park, in 1959; and

within Henry Coe State Park in the mainstem of Coyote Creek as well as the Middle Fork, and East Forks of Coyote Creek (Christman and Long unknown date).

Small to moderate populations were documented from 1986-2004 throughout the Coyote Creek drainage including mainstem Coyote Creek from Poverty Flat downstream to Gilroy Hot Springs, Middle Fork Coyote Creek, Little Coyote Creek, Water Gulch Creek, Grizzly Gulch Creek, and Soda Springs Canyon (PRA 1997; CAS 2001; CNDDDB 2016). Gonsolin (2010) documented breeding populations of foothill yellow-legged frog from 2004-2006 in upper Coyote Creek (upstream of Coyote Reservoir, from the inundation zone to Gilroy Hot Springs) and its tributary Dexter Creek and an unnamed tributary below Sheep Ridge. 2016 surveys of a 6 mile segment of Coyote Creek, from China Hole (Middle Fork Confluence) downstream to the Coyote Gate entrance of Henry W. Coe State Park, documented 9 adults, 1,000+ tadpoles and 121 egg masses (CNDDDB 2016). Foothill yellow-legged frogs currently persist within Henry Coe State Park in Coyote Creek and some of its tributaries, including Water Gulch, though their presence on the East Fork is minimal (HWCSP 2015). The park reports the species can be seen reliably along the Middle Fork of Coyote Creek, including Soda Springs Canyon, but is not found within the park in streams such as Pacheco Creek, Mississippi Creek, Coon Creek or Red Creek (HWCSP 2015).

Guadalupe River

Historical collection localities in the Guadalupe River drainage include from tributary Los Gatos Creek (at Los Gatos) in 1898 and Almaden Creek in 1950 (CAS 2001).

The Santa Clara Valley Water District reported small numbers of frogs in 2000 from two locations on Guadalupe Creek and tributary Rincon Creek (CNDDDB 2016).⁴²

Saratoga Creek

Foothill yellow-legged frogs were collected in the Saratoga Creek drainage from Saratoga on an unknown date before 1952; and from 2.6 miles WSW of Saratoga in 1953 (CAS 2001; UCMVZ 2015).

There are no known recent observations from Saratoga Creek.

Stevens Creek

Foothill yellow-legged frogs were collected in the Stevens Creek tributary in 1893 and 1939 (CAS 2001).

There are no known recent observations from Stevens Creek.

Upper Penetencia Creek and Fisher Creek in 1975 (Slevin 1928; CDFG 1975; UMMZ 2001; CAS 2001; UCMVZ 2015).

⁴² Two adult frogs from Guadalupe Creek downstream from Guadalupe Reservoir on 8/28/00; and 1 individual frog from Rincon Creek, upstream of the Guadalupe Creek confluence on 8/24/00 and 8/29/00 (CNDDDB 2016).

San Francisquito Creek

San Francisquito Creek forms the border between Santa Clara and San Mateo counties. Collections and records from this watershed are discussed here, as most records are from the Santa Clara side of the creek. There are historical collections from 1897-1940 in the San Francisquito Creek drainage (Slevin 1928; CAS 2001; FMNH 2001; USNM 2001).⁴³ Foothill yellow-legged frogs were reported to be “fairly common” in the San Francisquito drainage in the 1960s (Launer et al. 1999). Dyrkacz (1981) reported on an albinistic foothill yellow-legged frog collected from Portola Valley.

There are no known recent observations from San Francisquito Creek.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at 8 of 14 historical locations (57%) in Santa Clara County. H.T. Harvey and Associates (1999) conducted surveys in 1999 and concluded that the species had essentially disappeared from the farmed and urbanized lowland areas in Santa Clara County, as well as many of the perennial streams below major reservoirs. H.T. Harvey and Associates (1999) determined that the species is declining throughout Santa Clara County, but was still present in the Santa Cruz Mountains and fairly abundant in the foothill and mountain ranges of eastern Santa Clara County. Small populations were documented in the Llagas Creek tributary of the Pajaro River through 2007, and in headwater tributaries of the Alameda Creek drainage around Mt. Hamilton, such as Arroyo Hondo through 2011 and the Isabel Creek drainage through 2003; small to moderate populations remain throughout the Coyote Creek drainage, particularly upper Coyote Creek and Middle Fork Coyote Creek.

Alameda County

Alameda Creek

There are historical collection records from 1921-1969 in the Alameda Creek watershed, including mainstem Alameda Creek in Niles Canyon, upper Alameda Creek, and the Pleasanton and Livermore tributaries Arroyo Valle and Arroyo Mocho (CAS 2001; CMNH 2001; UMMZ 2001; UCMVZ 2015).⁴⁴

The largest remaining *R. boylei* population in Alameda County, and likely in the entire Bay Area, has been in upper Alameda Creek within Sunol Regional Park, from the Little Yosemite area upstream through Camp Ohlone. The CNDDDB has sightings from upper Alameda Creek from 1990, 1994, 1997, 1999, 2000 and 2001 in the reach from the Sunol Regional park staging area bridge upstream to the Calaveras Creek confluence, in and above Little Yosemite, and in the vicinity of the Alameda Diversion tunnel (CNDDDB

⁴³ Collection records include: 1 frog from Palo Alto before 1900; single frogs from the vicinity of Stanford University in 1897 and 1907; 3 frogs from Woodside Road at Stone Circle (likely the Bear Creek tributary) in 1935; 6 frogs from Woodside Road on two occasions in 1938; and 1 frog from Corte Madera Creek (a tributary above Searsville Lake) in 1940 (Slevin 1928; CAS 2001; FMNH 2001; USNM 2001).

⁴⁴ Historical collection records of *Rana boylei* from Alameda Creek include between Sunol and Mission San Jose (Niles Canyon) in 1921, Niles Canyon in 1939, and Sunol Regional Park in 1967; numerous specimens were collected from the Arroyo Mocho tributary in 1937 (22 frogs), 1939 (7 frogs), 1942, 1944, 1952, 1953, 1966, 1971 (2 frogs - MVZ 95204 & 95205), 1972 (2 frogs - MVZ 125364-125365), 1973 (MVZ 136289) and 1975 (7 larvae - MVZ 136418-136424), as well as 2 frogs taken from Livermore in 1969; and there is a collection record from Arroyo del Valle in 1960 (UMMZ 2001; CAS 2001; CMNH 2001; UCMVZ 2015).

2016). During surveys from July through October 1996, 295 *R. boylei* were found at 4 locations along upper Alameda Creek within Sunol and Ohlone Regional Parks (EBRPD 1998). Extensive surveys from 1997 to 1999 by Bobzien and DiDonato (2007) located the species in the Alameda Creek watershed in upper Alameda Creek (in Ohlone Regional Wilderness and Sunol Regional Wilderness); but did not locate the species in numerous Alameda Creek tributaries, including: Indian Joe Creek, La Costa Creek, Indian Creek, Shafer Creek, San Antonio Creek, Welch Creek and Whitlock Creek (Ohlone and Sunol Regional Wilderness); Sinbad Creek (Pleasanton Ridge Regional Park); Arroyo Del Valle Creek (Del Valle Regional Park); Tassajara Creek (Morgan Territory Regional Preserve and Tassajara Creek Regional Park); Altamont Creek (Brushy Peak Regional Preserve); and Brushy Creek (Vasco Caves Regional Preserve). Bobzien and DiDonato (2007) noted a small population in a 2-mile reach of Alameda Creek in the vicinity of the Sunol Visitor Center. Bobzien and DiDonato (2007) documented a “robust” population from 2000-2006 in a 1 mile stream reach of upper Alameda Creek in Camp Ohlone, and foothill yellow-legged frogs were noted to be “abundant” in 2006 during fish surveys of the Upper Ohlone reach of Alameda Creek, upstream of the Alameda Diversion Dam (B. Sak, pers. comm., 2006). However, this Camp Ohlone population has crashed since 2008 as a result of the drought; in 2015 there were only 4 clutches of eggs and no survival to metamorphosis, with all the breeding sites drying up as of early August (S. Kupferberg, pers. comm., 2015).

Foothill yellow-legged frogs in upper Alameda Creek have been particularly hard hit by the recent drought, disease and invasive species, with the “few frogs present above Little Yosemite within the Regional Park boundary and at Camp Ohlone” as of 2015, subject to predation by invasive bullfrogs and signal crayfish (Kupferberg 2015). Kupferberg and the East Bay Regional Park District documented an unprecedented chytrid fungus infection (*Batrachochytrium dendrobatidis*) in the *R. boylei* population in Little Yosemite, with confirmed mortality of frogs and Bd prevalence and loads on infected frogs an order of magnitude higher than levels leading to mortality and population decline in other well-studied amphibians (Kupferberg 2015). In the fall of 2013, foothill yellow-legged frogs in the Little Yosemite reach of Alameda Creek experienced an outbreak of Bd in which dead and dying juveniles were observed (Adams et al. in press). The SFPUC had planned to translocate frogs from this infected population to other areas of the watershed, presenting significant risk of spreading this infection to other currently uninfected *R. boylei* populations in the watershed (SFPD 2014; Kupferberg 2015). Those plans are now on hold and the project is being re-evaluated.

Small populations were documented along the Arroyo Mocho tributary SE of Livermore from 1997-2003 (CNDDDB 2016).⁴⁵ “Numerous” *R. boylei* were observed in 2000 during stream surveys in the Arroyo Mocho tributary along Mines Road SE of Livermore (A. Gunther, pers. comm., 2000). The species was present in the 1990s in the Arroyo Valle watershed above Del Valle Reservoir (EBRPD 1998), but this stream has not been surveyed for more than 15 years.

⁴⁵ On Arroyo Mocho from the Hetch-Hetchy Pumping Station bridge access road to about 0.75 upstream (~7 miles SE of Livermore), 6 adult frogs and 2 juveniles were observed on 4/15/97, 2 male and 4 female frogs on 4/6/99, and 16 egg masses and 20 adults on 3/26/03 (CNDDDB 2016).

Corral Hollow Creek

A foothill yellow-legged frog population was known historically from Corral Hollow Ecological Reserve (Schoenherr 1992) and prior to 1997 the species was “frequently” observed in Corral Hollow Creek within the Carnegie State Vehicle Recreation Area and downstream for several miles (Jones & Stokes 2000). A single juvenile *R. boylei* was found in April 1998 in upper Corral Hollow Creek (Jones & Stokes 2000; CNDDDB 2016); and a single adult frog was observed on Corral Hollow Creek about one half mile WSW of Tesla, within Carnegie SVRA, on April 28, 2014 (CNDDDB 2016). As discussed in the section above on San Joaquin County, foothill yellow-legged frogs were restricted to the upper half mile of Corral Hollow Creek in Alameda County by the late 1990s and near extirpation (Jones & Stokes 2000). This remnant population is jeopardized by plans to expand off-road vehicle activity and a vehicle stream crossing through the last known sighting area in the Carnegie SVRA (Jones & Stokes 2000; CNDDDB 2016).

Western Alameda County

The species was collected from an unknown location in Oakland in 1891 and from Berkeley in Telegraph (Claremont) Canyon in 1912 (Slevin 1928; CAS 2001; UCMVZ 2015). There are no recent observations from western Alameda County.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 4 of 13 historical locations (31%) in Alameda County. Bobzien and DiDonato (2007) conducted extensive surveys from 1997 to 1999 during peak breeding season for foothill yellow-legged frogs, at 100 stream stations on 42 streams within East Bay Regional Park District lands throughout Alameda and Contra Costa Counties. Except for populations in several drainages within the Alameda Creek watershed, the species was extirpated or absent from all Alameda County streams surveyed. The largest remaining populations in Alameda County (and likely in the S.F. Bay Area) were in upper Alameda Creek. These populations have now crashed and are jeopardized by multiple factors including drought, Bd infection, recreation impacts, ill-timed dam flow releases that cause direct mortality of early life stages, invasive predators that flourish under dam-altered flow conditions, and future hypolimnetic dam releases that shift water temperatures outside the thermal niche of tadpoles.

Contra Costa County

San Leandro Creek

There are historical records from upper San Leandro Creek and Moraga Creek (USACE 2001; CAS 2001; UCMVZ 2015).⁴⁶

There is a CNDDDB report from February 1997 of 2 adult yellow-legged frogs in an intermittent tributary to Moraga Creek near the Gateway Valley; subsequent surveys of this area failed to detect any *R. boylei* (CNDDDB 2016; J. Miller, pers. comm., 2015). The species was extirpated as early as the 1950s from all East Bay Municipal Utility District

⁴⁶ Collection records from upper San Leandro Creek include Redwood Peak in 1909 and the town of Canyon in 1947 (USACE 2001; CAS 2001; UCMVZ 2015). Yellow-legged frogs occurred historically in the Moraga Creek drainage near the Gateway Valley (USACE 2001).

watershed lands in the East Bay, which includes large portions of the upper San Leandro Creek watershed (EBMUD 1994).

San Pablo Creek

There are historical records from San Pablo Creek from 1917 to the 1950s (UCMVZ 2015).⁴⁷ *Rana boylei* was apparently once abundant in San Pablo Creek near Orinda, with 13 frogs collected on a single day in August 1922 (UCMVZ 2001); the creek was reportedly “full of yellow-legs” in the 1950s (G. Beeman, pers. comm., 2002).

The species was extirpated as early as the 1950s from all East Bay Municipal Utility District watershed lands in the East Bay, which includes large portions of the San Pablo Creek watershed (EBMUD 1994). Extensive surveys from 1997 to 1999 by Bobzien and DiDonato (2007) failed to locate the species in the San Pablo Creek watershed (Arroyo Del Hambre Creek and Bear Creek in Briones Regional Park).

Pinole Creek

The species was collected in Pinole Creek in 1939 (UCMVZ 2015).

The species was extirpated as early as the 1950s from all East Bay Municipal Utility District watershed lands in the East Bay, which includes large portions of the Pinole Creek watershed (EBMUD 1994). Extensive surveys from 1997 to 1999 by Bobzien and DiDonato (2007) failed to locate the species in the Pinole Creek watershed (Castro Creek in Sbrante Ridge Regional Preserve).

Walnut Creek

There are historical records from 1891-1953 in Walnut Creek and tributaries Mitchell Creek and Pine Creek (CAS 2001; UCMVZ 2015).⁴⁸

Rana boylei was found in unknown numbers in 1996 during aquatic surveys in the Bolinger Creek drainage, within Las Trampas Regional Park (EBRPD 1998). Extensive surveys from 1997 to 1999 by Bobzien and DiDonato (2007) failed to locate the species in other streams in the Walnut Creek watershed (Bolinger Creek in Las Trampas Wilderness, Sycamore Creek in Sycamore Valley Regional Park and Pine Creek in Diablo Foothills Regional Park). Foothill yellow-legged frogs reportedly persisted in small numbers in the early 2000s in headwaters tributaries draining Mt. Diablo, such as Pine Creek and possibly Mitchell Creek (G. Beeman, pers. comm., 2002).

Marsh Creek

There are historical records from 1939-1976 in the Marsh Creek drainage (UCMVZ 2015; CNDDDB 2016).⁴⁹ Harvey et al. (1992) noted that *R. boylei* was possibly found historically in small creeks near Pittsburgh and Brentwood.

⁴⁷ Historical collection records include: San Pablo Canyon in 1917; and 13 frogs from near Orinda in 1922 (UCMVZ 2015). The species was collected from Orinda in the 1950s (G. Beeman, pers. comm., 2002).

⁴⁸ Historical collection localities from the include: Mt. Diablo in 1891 and 1931; 9 frogs from the W side of Mt. Diablo and Pine Canyon in July 1912; Mitchell Creek in 1912; a creek at Lafayette in 1922; and Pine Creek E of Walnut Creek in 1953 (CAS 2001; UCMVZ 2015).

Extensive surveys from 1997 to 1999 by Bobzien and DiDonato (2007) failed to locate the species in the Marsh Creek watershed (Marsh Creek and Round Valley Creek in Round Valley Regional Preserve, and Marsh Creek in Morgan Territory Regional Preserve and Clayton Ranch Regional Preserve), or in streams in Black Diamond Mines Regional Preserve draining into the Delta in eastern Contra Costa County near Pittsburgh and Antioch (Kirker Creek, Sand Creek, Markley Creek, Somersville Creek, West Antioch Creek and Homestead Creek). There are no recent documented occurrences of foothill yellow-legged frogs in the eastern Contra Costa Habitat Conservation Plan inventory area, covering major portions of eastern Contra Costa County (CCC 2006).

Recent status: Jennings and Hayes (1994) were able to relocate the species during resurvey efforts from 1988-1991 at only 3 of 9 historical locations (33%) in Contra Costa County. Jennings and Hayes (1994) believed that 8 of 11 known historical populations in Contra Costa County were extinct, with the 3 remaining records concentrated in the Mount Diablo region. Bobzien and DiDonato (2007) conducted extensive surveys for foothill yellow-legged frogs from 1997 to 1999 during peak breeding season, at 100 stream stations on 42 streams within East Bay Regional Park District lands throughout Alameda and Contra Costa Counties. *Rana boylei* was extirpated or absent from all streams surveyed in Contra Costa County (Bobzien and DiDonato 2007). The species is likely very nearly extirpated from Contra Costa County, with the possible exception of a few locations in headwater streams around Mount Diablo.

Solano County

Putah Creek drains to the Yolo Bypass in the Sacramento Valley. Alamo Creek and Ulati Creek flow to Cache Slough in the San Francisco Bay Delta. Ledgewood Creek is tributary to Suisun Slough and Suisun Bay.

Putah Creek

Small to moderate populations of *R. boylei* were documented in tributaries to Lake Berryessa and Putah Creek in extreme northwestern Solano County: Cold Canyon Creek in 1999 and 2004, and in Wild Horse Creek in 2006 (Solano County Water Agency 2002; CNDDDB 2016).⁵⁰

Alamo Creek, Ulati Creek

There are historical collection records from Vacaville; and 7 frogs from 3 miles W of Vacaville in July 1912 (Slevin 1928; UCMVZ 2015).

⁴⁹ Historical collection localities in the Marsh Creek drainage include: 7 frogs from 3 miles E of the Livermore turnoff in May 1939; 10 frogs from 7 miles SE of Clayton in 1939 and 1940; 7 frogs from 4 miles E of Mt. Diablo in August 1950; 7 frogs from 6.3 miles SSE of Clayton in 1951 and 6 frogs from this location in March 1953; and 2 frogs (MVZ 55556 & 55557) collected by Zweifel from Marsh Creek 7.4 miles SE of Clayton on March 25, 1976 (UCMVZ 2015; CNDDDB 2016).

⁵⁰ In Cold Canyon Creek, more than 100 juvenile frogs were observed on 6/25/99 and 2 adults on 6/5/04; in Wild Horse Creek, about 1.2 miles upstream of the confluence with Cold Canyon Creek (in Stebbins Cold Canyon Reserve), more than 25 adult and subadult frogs were observed on 6/11/06 (CNDDDB 2016).

The species was reported to occur in the Vaca Mountain/Pleasants Valley/English Hills Conservation Area (Solano County Water Agency 2002). The Solano County Water Agency (2002) reported no recent observations of foothill yellow-legged frogs in drainages northwest of Vacaville, but there were subsequent observations of small numbers of frogs in Alamo Creek in 2003 and 2004, and in Ulatis Creek in 2004 (CNDDDB 2016).⁵¹

Sulphur Springs Creek

Foothill yellow-legged frogs are reported to have historically occurred in Sulphur Springs Creek, but were not found there during surveys in the mid-1990s (Solano County Water Agency 2002).

Ledgewood Creek

There is a 2002 report of 2 adult frogs from an unnamed tributary to Ledgewood Creek in the Rancho Solano area in northeastern Fairfield (Solano County Water Agency 2002; CNDDDB 2016).

Recent status: The Solano County Water Agency contends the scarcity of recent *R. boylei* records in the county is due to lack of survey effort for this species, as there is suitable habitat on private land in the upper reaches of streams in western Solano County such as Green Valley Creek, Suisun Creek, Wild Horse Creek, Cook Canyon Creek, Laguna Creek, Alamo Creek and its perennial tributaries, and Ulatis Creek (Solano County Water Agency 2002). There were observations in the 2000s of small to moderate populations in tributaries of Putah Creek, and small numbers of frogs in Alamo Creek, Ulatis Creek and Ledgewood Creek.

Napa County

Putah Creek drains to the Yolo Bypass in the Sacramento Valley. The Napa River drains to the Carquinez Straits on San Pablo Bay.

Historical collections of *R. boylei* were made from 1906-1974 from throughout the Napa River drainage (Napa River and tributaries Conn Creek, Dry Creek, North Slough, Rector Creek, Redwood Creek and Sulphur Creek), and above Lake Berryessa Reservoir (Putah Creek and tributaries Butts Creek, Capell Creek, Elicura Creek, Pope Creek, St. Helena Creek and Swartz Creek) (USNM 2001; FMNH 2001; CAS 2001; CNDDDB 2016; UCMVZ 2015). Large numbers of frogs were collected historically at some locations in Napa, for example, 38 frogs collected from Calistoga in 1915; 15 frogs from Conn Creek in March 1941; and 14 frogs from Sulphur Creek (where numerous collections were made) in May 1969 (UCMVZ 2015). E. Gerstung reported foothill yellow-legged frogs were “numerous” in April 1956 in Spanish Valley, SW of Lake Berryessa (CNDDDB 2016).

Foothill yellow-legged frogs were documented in very small numbers in the 1990s in only

⁵¹ In Alamo Creek, 6 larvae and 2 metamorphs in Gates Canyon (3 miles NW of Vacaville) on 8/7/03; and 2 adults 1.1 miles W of the junction of Lagoon Valley Road and Pleasants Valley Road on 6/5/04 (CNDDDB 2016). In Ulatis Creek, 13 adults and 10 tadpoles were observed along Mix Canyon Road (1.5 miles W of Lagoon Valley Road) on 5/30/04 (CNDDDB 2016).

a few locations in four Putah Creek tributaries (Butts Creek, Elicura Creek, Toll Canyon Creek and Zim Zim Creek) above Lake Berryessa (CNDDDB 2016). Foothill yellow-legged frogs were documented in small numbers in the 2000s in a few locations in the Napa River drainage (Bear Creek, Heath Creek, Sage Creek and at the Old La Joya Quicksilver Mine) and Putah Creek tributaries (Capell Creek, Elicura Creek and James Creek) above Lake Berryessa (CNDDDB 2016).

Surveys were done in 2007 for the proposed Walt Ranch development and small numbers of foothill yellow-legged frogs were found in Capell Creek (5 adults and 3 juveniles) and Milliken Creek (6 adults), but these surveys do not appear to have been recorded in the CNDDDB (Napa County 2016). Foothill yellow-legged frog populations in Milliken Creek in the Milliken Reservoir watershed and the upper Capell Creek drainage are threatened by the recently approved development of vineyards and associated infrastructure at Walt Ranch, including roads with numerous planned stream crossings (Napa County 2016).

Recent status: The species has clearly declined in Napa County and is less widely distributed in the Napa River and Putah Creek drainages, with no known significant populations remaining.

Sonoma County

Historical collections of *R. boylei* were made from 1911-1974 from tributaries throughout the Russian River drainage (Ash Creek, Austin Creek, Dry Creek, Dutch Bill Creek, Pole Mountain Creek and Warm Springs Creek), in the Gualala River drainage (Gualala River, North Fork Gualala River, Pepperwood Creek, South Fork Gualala River and Wolf Creek), and in Salmon Creek (Freestone) and Adobe Creek in Petaluma (CAS 2001, CMNH 2001, FMNH 2001; LSUMNS 2001; UMMZ 2001; UCMVZ 2015). Large numbers of frogs were collected historically at some locations in Sonoma, for example 24 frogs collected from Austin Creek on October 12 and 13, 1940 (UCMVZ 2015).

In the 1990s, Harvey et al. (1992) reported the species as still “common” in the Sonoma Mountains east of Petaluma (Petaluma River and Sonoma Creek drainages) with sightings of “large numbers” indicating that populations were stable. Foothill yellow-legged frogs were documented in the 1990s throughout the Russian River drainage (Russian River and tributaries Burns Creek, Crocker Creek, Franz Creek, Maacama Creek, Sausal Creek and Squaw Creek), throughout the Austin Creek sub-drainage (Austin Creek and tributaries Big Austin Creek, Black Rock Creek, Blue Jay Creek, Conshea Creek, Devil Creek, East Austin Creek, Gilliam Creek, Gray Creek, Lawhead Creek, Little Sulphur Creek, Pole Mountain Creek, Sulphur Creek and Ward Creek), and in the Mark West Creek sub-drainage (Mark West Creek and tributaries Porter Creek and Weeks Creek) (CNDDDB 2016). All observations were of small populations, with notable locales being Big Austin Creek, Crocker Creek, Little Sulphur Creek and Squaw Creek.⁵² Foothill yellow-legged frogs were also documented in small numbers in the 1990s in: tributaries to the Gualala River (Wheatfield Fork and unnamed tributary); Sonoma Creek; Santa Rosa Creek in April and May of 1991 (MacTague and Northern

⁵² In Big Austin Creek (1 adult and 79 juveniles captured during creek restoration construction on 10/19/98 and 10/21/98); Crocker Creek (5 adults and thousands of tadpoles on 6/9/97); Little Sulphur Creek (8 adults and thousands of tadpoles on 6/9/97); and Squaw Creek (20 adults on 8/25/98) (CNDDDB 2016).

1993); Copeland Creek, a Laguna de Santa Rosa tributary (13 adults on 8/29/96); and Adobe Creek (“lots of frogs” on 6/10/97) (CNDDDB 2016).

In the 2000s, small populations were documented throughout the Russian River drainage (Russian River and tributaries Cherry Creek, Crocker Creek, Gird Creek, Green Valley Creek, Ingalls Creek, Maacama Creek, McDonnell Creek, Miller Creek, Porter Creek, Sausal Creek and Skunk Creek), throughout the Austin Creek sub-drainage (Austin Creek and tributaries Big Austin Creek, Big Sulphur Creek, Blue Jay Creek, Kidd Creek, Schoolhouse Creek, Trestle Creek and Ward Creek), and in the Mark West Creek sub-drainage (Mark West Creek and tributary Humbug Creek) (CNDDDB 2016). All observations were of small numbers of frogs, with the exceptions of a moderate population in Cherry Creek (20+ individuals on 11/5/02) and a moderately large population in Gird Creek (about 50 adults and sub-adults on 10/17/00); other notable locales were Miller Creek, Porter Creek and Ward Creek) (CNDDDB 2016).⁵³ Foothill yellow-legged frogs were documented in small numbers in the 2000s in the South Fork Gualala River drainage, in tributaries Blue Slide Creek, Buckeye Creek (only 8 frogs documented on 5/7/08, but noted that frogs still existed in “quite large numbers”), Fuller Creek, House Creek and Patchett Creek (CNDDDB 2016). Foothill yellow-legged frogs were also documented in small numbers in the 2000s in Adobe Creek, Sonoma Creek tributaries Carriger Creek and Stuart Creek, tributaries of the Laguna de Santa Rosa (Crane Creek and Copeland Creek), and in a few small coastal drainages (Fort Ross Creek, Kolmer Gulch, East Branch Russian Gulch and Russian Gulch); with a moderate population in 2008 in Sonoma Creek (21 adults and more than 100 tadpoles on 8/7/08) (CNDDDB 2016).

The Sonoma County Water Agency (2008) compiled documented occurrences and completed habitat assessments for foothill yellow-legged frogs at 189 sites (102 unique streams) within Sonoma County, noting occurrence data for the Laguna de Santa Rosa, Petaluma River, Sonoma Creek and Russian River watersheds, as well as other scattered occurrences. SCWA (2008) concluded the species is likely distributed throughout the county in natural foothill and mountain streams with moderate gradient and permanent or semi-permanent water. SCWA (2015) reported 71 documented occurrences of foothill yellow-legged frog throughout Sonoma County.

Recent status: *R. boylei* is still widely distributed throughout Sonoma County in many Russian River tributaries (particularly Austin Creek and Mark West Creek), the South Fork Gualala River drainage, watersheds of the Laguna de Santa Rosa, Petaluma River, Sonoma Creek, Adobe Creek, and a few coastal streams, mostly in small populations. The most significant remaining Sonoma locales are a moderately large population documented in Gird Creek in 2000 and moderate populations in Sonoma Creek in 2008 and Russian River tributary Cherry Creek in 2002.

Marin County

Rana boylei was found historically throughout Marin County, including the Lagunitas Creek drainage (Lake Lagunitas, Alpine Lake, Lagunitas Creek, San Geronimo Creek, Devil’s Gulch Creek, Nicasio Creek, Arroyo Nicasio, Arroyo Sausal, Halleck Creek, Olema Creek, Big Carson Creek, Little Carson Creek), tributaries on Mount Tamalpais

⁵³ In Miller Creek and an unnamed tributary (about 15 adults, about 300 subadults and “many” tadpoles on 8/4/00); Porter Creek (100+ juveniles on 7/30/03); and Ward Creek (10 adults on 4/27/05).

(Cataract Creek, Rock Spring, Phoenix Gulch), Redwood Creek, tributaries to Bolinas Lagoon (Pine Gulch Creek, Pike County Gulch) and Tomales Bay (Walker Creek, Salmon Creek, Millerton Gulch), and several creeks in eastern Marin County draining to San Francisco Bay (San Anselmo Creek, Arroyo Corte Madera, Arroyo San Jose, Fairfax Creek, Big Rock Creek, Dairy Creek, Mill Creek) (USNM 2001; CMNH 2001; TMM 2001; UMMZ; LSUMNS 2001; CAS 2001; Garcia and Associates 2010b; UCMVZ 2015). There are 84 historic collection specimens from Marin County in the U.C. Museum of Vertebrate Zoology (UCMVZ 2015). Garcia and Associates (2010b) identified a total of 60 foothill yellow-legged frog localities within Marin County, based on queries to the CNDDDB, museum specimen records from HerpNet, and personal communications with biologists. Fifty-one of these localities were based upon museum specimen records collected from 1891-1972, 7 localities were based upon CNDDDB records of foothill yellow-legged frog observations from 1956-2008, and two localities were obtained via personal communications from biologists.

Rana boylei was historically quite abundant in Marin County, as evidenced by large specimen collections of frogs: 15 frogs from Lagunitas Creek in May 1904, 19 frogs in April 1911, 57 frogs in April 1928 and 15 frogs in 1931; 38 frogs from Mt. Tamalpais in August 1928; 8 frogs from Muir Woods (Redwood Creek) in September 1913 and 10 frogs in May 1922; 18 frogs from Pike County Gulch in March 1963; and 12 frogs from Millerton Gulch in May 1944 (UKMNH 2001; USNM 2001; CMNH 2001; CAS 2001; UCMVZ 2015).

Focused survey efforts in the 1990s failed to find any foothill yellow-legged frogs in Redwood Creek, or around Muir Woods (Ely 1993; Fong 1997). A foothill yellow-legged frog population in Cataract Creek that was considered to be abundant in the early 2000s (CDFG 2003) appears now to be extirpated (Garcia and Associates 2010b). *Rana boylei* has now been extirpated from Lagunitas Creek below Peters Dam, East Fork Lagunitas Creek, Cataract Creek and the entire Mount Tamalpais watershed above Lake Alpine and Lake Lagunitas and from west slope drainages of Mount Tamalpais.

Small populations of foothill yellow-legged frog may persist in the Tomales Bay tributaries Walker Creek and Salmon Creek; other populations may exist in Marin County, however the current population status of many historically occupied sites remains poorly understood and repeat surveys of these sites are needed (Garcia and Associates 2010b).

Only two known foothill yellow-legged frog populations remain of a once more widespread distribution within the Mount Tamalpais watershed, in Little Carson Creek and Big Carson Creek, both tributaries to Kent Lake (Garcia and Associates 2010b). Fellers documented 69 adults, 71 subadults, 1,828 larvae and 1,925 egg masses in Big Carson Creek, from 1996-2008 cumulatively (CNDDDB 2016). The Little Carson Creek population appears to be large and stable: in 2014 Garcia and Associates reported observing 96 adult males, 3 gravid females, 7 spent females and 15 egg masses at Little Carson Falls (MMWD 2014), consistent with numbers in recent years.

Recent status: The species has been extirpated from most former localities and watersheds in Marin County. The Little Carson Creek and Big Carson Creek tributaries to Kent Lake contain the only known significant populations in Marin County in recent years. Small populations may persist in the Tomales Bay tributaries Walker Creek and Salmon Creek.

Upper Sacramento River

There have been documented declines in the upper Sacramento River basin, but small populations were documented in the 1990s and 2000s in Shasta County in more than three dozen tributaries in the Sacramento River drainage, with significant populations in the Sacramento River (near Dog Creek and Campbell Creek) and in Willow Creek and its tributaries. Small numbers of frogs persist in eastern Tehama County in the Battle Creek, Paynes Creek, Antelope Creek, Little Antelope Creek, Mill Creek, and Deer Creek drainages.

Yolo County

Cache Creek drains to the Sacramento River. Putah Creek drains to the Yolo Bypass in the Sacramento Valley.

The paucity of recorded occurrences of foothill yellow-legged frogs at lower elevations suggests that the foothill yellow-legged frogs may never have been common throughout much of Yolo County (Yolo County 2013).

Cache Creek

Small populations were documented from 1997-2000 at a handful of locations in the Cache Creek drainage in extreme northwestern Yolo County, including Cache Creek and tributaries Bear Creek, Davis Creek and Fiske Creek (CAS 2001; Yolo County 2013; CNDDDB 2016).⁵⁴

Putah Creek

Slevin (1928) and Harvey et al. (1992) noted a historical collection record from Putah Creek, 4 miles W of Winters.

Recent status: Unknown. The species may never have been common throughout much of Yolo County. Small populations were documented in Yolo County in the Cache Creek drainage through 2000.

Colusa County

Sand Creek drains to the lower Sacramento Valley. Stony Creek is a tributary of the Sacramento River. Cache Creek drains to the Sacramento River.

⁵⁴ Three adults in Davis Creek upstream from Rayhouse Road on 4/21/97; 3 adults 0.5 mile downstream from Davis Creek Reservoir on 4/21/97; 1 frog above Davis Creek Reservoir in May 1997; 2 adults and 1 juvenile in Bear Creek on 6/10/97; 1 juvenile from the oxbow adjacent to Cache Creek and Bear Creek confluence on 6/20/97; 2 adults near the confluence of Cache Creek and Bear Creek (WNW of Rumsey) on 6/24/97 and 7/8/97; 2 adults and 1 juvenile in a stock pond on Blue Ridge (1.5 miles E of Fern Spring, between Lake Berryessa and Capay Valley) during surveys from May-June 1999; and 4 adults, 13 subadults and 250 larvae along Fiske Creek (Fellers site ID# P-466; about 0.35 mile S of Cache Creek confluence), on 8/5/00 (CAS 2001; Yolo County 2013; CNDDDB 2016).

Sand Creek

There is a historical collection record of 1 frog from Sand Creek (5 miles W of Arbuckle), in the southeastern portion of the county, on March 8, 1942 (UCMVZ 2015).

Stony Creek

There are historical collections from 1933-1974 in the Stony Creek drainage, including Colusa Creek, Little Stony Creek, Mill Creek, South Fork Stony Creek and Stony Creek (CAS 2001; CNDDDB 2016; UCMVZ 2015).⁵⁵ Foothill yellow-legged frogs were apparently relatively common historically in the Stony Creek drainage, with 9 frogs collected from Stony Creek over two days in 1933, and 22 frogs collected from Mill Creek on a single day in 1973 (UCMVZ 2015).

Small populations were documented throughout the Stony Creek drainage from 1989-2000, including Little Stony Creek, Little Sullivan Creek, Mill Creek, North Fork Stony Creek, South Fork Stony Creek, Stony Creek and Sullivan Creek (CAS 2001; CNDDDB 2016; UCMVZ 2015).⁵⁶ Fellers (1996) observed *R. boylei* throughout the Stony Creek drainage within the Mendocino National Forest, including the tributaries Little Stony Creek, Mill Creek, and Sullivan Creek. Fellers (1996) found significant numbers of adult frogs at locations in Little Stony Creek (13 frogs) and Mill Creek (10 frogs), with notably good reproduction in Mill Creek (2,566 tadpoles). Repeat follow-up surveys by Fellers through 2008 revealed continuing significant populations in Mill Creek and Little Stony Creek (CNDDDB 2016).⁵⁷

⁵⁵ Collections in the Little Stony Creek tributary drainage: 1 frog from 6 miles S of Stonyford on October 10, 1963; 1 frog from Colusa Creek in 1973; and Fellers collected 22 frogs from Mill Creek (at Fouts Springs Campground) on 5/5/73, 1 frog on 5/6/73, and 2 frogs on 8/3/74 (CAS 2001; CNDDDB 2016; UCMVZ 2015). Collections in the South Fork Stony Creek drainage: 2 frogs from 1 mile S of Redbridge in November 1960 (CAS 2001). Collections in Stony Creek: 9 frogs from 3 miles W of Stonyford on October 1-2, 1933; and 1 frog from 5 miles W of Stonyford on February 10, 1968 (CAS 2001; UCMVZ 2015).

⁵⁶ Two frogs from a South Fork Stony Creek tributary at Davis Flat in April 1989; 1 adult frog from an unnamed tributary to South Fork Stony Creek (in the vicinity of Davis Flat) on 3/9/91; 1 frog from North Fork Campground on 2/24/97; 3 adults from Mill Creek (below Brim Road bridge) on 4/11/97; single frogs from 3 locations in South Fork Stony Creek on 9/16/99; 1 frog from Stony Creek (NE of Candy Bucket Spring) on 9/17/99; 1 frog from Mill Creek (upstream of Mill Creek Campground) in August 2000; 1 frog from an unnamed creek between Wolf Glade and Diversion Dam Campground, in August 2000; 1 frog from Little Stony Creek (upstream of Trout Creek) on 8/15/00; 1 frog from Little Stony Creek (between Sullivan Creek and Trout Creek) on 8/15/00; 1 frog from Sullivan Creek (upstream of Little Stony Creek confluence) on 8/16/00; 1 frog from Little Stony Creek on 8/17/00; and 1 frog from Little Sullivan Creek on 8/17/00 (CAS 2001; CNDDDB 2016; UCMVZ 2015).

⁵⁷ At Mill Creek at Fouts Springs Campground (Fellers sites Y-809A and Y-809B) a cumulative total of 99 adults, 160 subadults, 4,858 larvae and 6,532 egg masses during surveys from 1995-97, 1999-2004, and 2006-08; at South Fork Stony Creek confluence, (Fellers site Y-809B), a cumulative total of 55 adults, 96 subadults, 10,544 larvae and 36,251 egg masses during surveys from 1995-1996, 1999-2004 and 2006-2007; and in Little Stony Creek (at Digger Pine Campground, Fellers site Y-828), a cumulative total of 102 adults, 264 subadults, 8,451 larvae and 5,775 egg masses during surveys from 1995-2004 and 2006-2008 (CNDDDB 2016).

Cache Creek

Small populations were documented in the Cache Creek tributaries Bear Creek and Sulphur Creek from 1997-1998 (CNDDDB 2016).⁵⁸ Fellers observed small numbers of frogs in Bear Creek from 1998-2004 (CNDDDB 2016).⁵⁹

Recent status: Significant populations remained in Colusa County in the Stony Creek drainage in Little Stony Creek and Mill Creek through 2008, with older observations of small numbers of frogs in Stony Creek and tributaries South Fork Stony Creek, North Fork Stony Creek and Sullivan Creek through 2000. The species was present in the Cache Creek tributaries Bear Creek, Sulphur Creek and Letts Creek in the 1990s and early 2000s, but there are no reported observations since 2004.

Glenn County

Stony Creek is a tributary of the Sacramento River. Black Butte River is a tributary of the Middle Fork Eel River.

Stony Creek

There are historical collections from 1912-1971 in Glenn County in Stony Creek and tributaries Grindstone Creek and North Fork Stony Creek (Slevin 1928; CAS 2001; CNDDDB 2016; UCMVZ 2015).⁶⁰

There were a handful of observations from 1995-2000 of small numbers of frogs in the Stony Creek drainage, including Black Diamond Creek, North Fork Stony Creek, Salt Creek and Stony Creek (CAS 2001; CNDDDB 2016).⁶¹

Black Butte River

A single juvenile (CAS #209128) was collected from the Black Butte River drainage, in eastern Glenn County (downstream of "The Basin" and W of Bear Wallow Ridge), on 6/24/99 (CAS 2001; CNDDDB 2016).

⁵⁸ Three frogs from Bear Creek downstream of the confluence with Sulphur Creek 4/11/97-4/25/97; 1 frog from Bear Creek (near Wilbur Springs Road bridge) on 4/25/97; 2 adults and 1 juvenile from Sulphur Creek (above Wilbur Springs) on 4/25/97 and 6/2/97; 1 frog from Bear Creek at Thompson Canyon on 6/10/97, with 2 additional adults observed; 5 adults from East Fork Sulphur Creek on 3/26/98; 5 adults from West Fork Sulphur Creek on 3/26/98 and 4/6/98; and 3 adults from Bear Creek on 5/5/98, with 3 additional adults observed (CNDDDB 2016).

⁵⁹ Three adults in Thompson Canyon (at Bear Creek confluence) on 3/20/98; 3 larvae in Bear Creek (Fellers site P-465) on 8/5/00; 2 adults in Letts Creek (Fellers site P-471) on 8/7/00; and 1 adult at confluence of Bear Creek and Sulphur Creek (vicinity of Wilbur Springs) on 3/20/04 (CNDDDB 2016).

⁶⁰ Five frogs from Winslow (5 miles W of Fruto) on June 18 and June 20, 1912; from Fruto before 1928; 1 frog in Grindstone Creek (4.5 miles S of Millsap) on March 21, 1954; 1 frog in North Fork Stony Creek (1 mile N of Redbridge), in November 1960; and 1 frog (MVZ 125357) from a Grindstone Creek tributary (along Hull Road N of Elk Creek) on March 29, 1971 (Slevin 1928; CAS 2001; CNDDDB 2016; UCMVZ 2015).

⁶¹ One adult, 4 subadults and 2 egg masses along Salt Creek at Rattlesnake Creek (Fellers site ID#Y-813) on 5/10/95; 1 adult (CAS #202599) collected on 2/24/97 and 1 adult (CAS #202583) collected on 3/29/97 in an unnamed tributary to Stony Creek, on the Brittan Ranch; 2 small adults captured and 200-300 tadpoles observed in an unnamed tributary to Salt Creek, just E (downstream) from Sanhedrin Road, 4 miles NW of the town of Elk Creek, on 4/10/97; 3+ tadpoles observed in Black Diamond Creek, just W of the Dry Creek Confluence, on 6/5/99; and 1 collection specimen from North Fork Stony Creek on 7/24/00 (CAS 2001; CNDDDB 2016).

Recent status: Unknown. Small populations were documented in Glenn County in the Stony Creek drainage through 2000, and an observation in Black Butte River in 1999.

Tehama County

Battle Creek

A single foothill yellow-legged frog was collected from Battle Creek on March 31, 1932 (UCMVZ 2015).

In the 2000s, small numbers of frogs were documented in the Battle Creek drainage, in tributaries South Fork Battle Creek, Soap Creek and Ripley Creek (CNDDDB 2016).⁶²

Paynes Creek

Grinnell et al. (1930) collected small numbers of foothill yellow-legged frogs along Paynes Creek in 1924 (UCMVZ 2015).⁶³ In April 1928, Grinnell et al. (1930) observed “several” foothill yellow-legged frogs in the Paynes Creek drainage (9 miles NE of Red Bluff), and several more adults and several egg masses later in the month at another nearby location. A single frog was collected from Meadow Ranch (3 miles W of Paines Creek Post Office) on February 15, 1931 (UCMVZ 2015).

In the 1990s, small numbers of frogs were documented in the Paynes Creek drainage (CNDDDB 2016).⁶⁴

Red Bank Creek

Bourque (2008) was able to find enough foothill yellow-legged frogs in the Red Bank Creek watershed during telemetry surveys from 2004-2005 to opportunistically capture 79 adult frogs.

Antelope Creek

Rana boylei was known to occur historically in the Antelope Creek watershed, below the elevation of approximately 3,200 feet (LNF and PNF 1999).

Small numbers of foothill yellow-legged frogs were found in Antelope Creek and in the Indian Creek tributary from 2001-2003 (Hayes et al. 2013). Fellers documented small populations from 1997-2003 along Antelope Creek at North Fork Antelope Creek, and at

⁶² One adult, 2 juveniles and 40-50 tadpoles in South Fork Battle Creek, just downstream from the PG&E South Powerhouse, S of Manton, on 6/15/00; 1 adult and 1 juvenile in Soap Creek, downstream of the diversion dam, on 6/16/00; 1 juvenile upstream of the dam on South Fork Battle Creek, on 6/26/00; 1 adult, 2 juveniles and 3 egg masses in South Fork Battle Creek, at the Manton Road crossing, on 4/25/05; and 2 adults and 16 juveniles in Ripley Creek and an associated tributary, on 4/25/05 (CNDDDB 2016).

⁶³ Four frogs from Dale’s Ranch in May 1924; 2 frogs from “Paines Creek” on May 12, 1924; 4 frogs from Elliott’s Ranch, (5 miles W of Payne’s Creek Post Office), on June 5, 1924; and 3 frogs from Lyman’s, in the Plum Creek tributary, on June 8 and June 14, 1924 (UCMVZ 2015).

⁶⁴ Three adults in Paynes Creek, approximately 200 feet N of Highway 36, on 4/1/93; and 1 adult observed by Fellers (site ID# L-177) at Plum Creek and an unnamed tributary confluence (about 2 miles N of Finley Butte), on 6/22/95 (CNDDDB 2016).

the Antelope Creek and Indian Creek confluence (CNDDDB 2016).⁶⁵ At least 10 adults and 1 egg mass were observed in Antelope Creek near Facht Place Campground on 5/25/16 (CNDDDB 2016).

Little Antelope Creek

Fellers documented small populations in 1995 in Little Antelope Creek and tributary Cottonwood Creek (CNDDDB 2016).⁶⁶

Dye Creek

Fourteen *R. boylei* were collected from Dye Creek in the Gray Davis Dye Creek Preserve (6.0 miles E and 3.5 miles N of Gerber) on April 1, 1970 (CNDDDB 2016; UCMVZ 2015).

A single frog was collected from the North Fork Dye Creek in September 1996 (CAS 2001).

Mill Creek

Slevin (1928) noted that *R. boylei* had been collected from Mill Creek, near Tehama. Grinnell et al. (1930) took specimens from Mill Creek at 260 feet (2 miles NE of Tehama), from June 8-13, 1912 (UCMVZ 2015). The species was known to occur historically in the Mill Creek watershed below the elevation of approximately 3,200 feet (LNF and PNF 1999).

Fellers found 3 larvae along Mill Creek (site L-050F; just W of Black Rock, about 3 miles NW of Flatiron Mountain), on 8/15/94 (CNDDDB 2016). Small numbers of frogs were found in Mill Creek from 2001-2003 (Hayes et al. 2013).

Deer Creek

The species was known to occur historically in the Deer Creek watershed below the elevation of approximately 3,200 feet (LNF and PNF 1999).

Fellers documented small populations from 1994-2006 in the Deer Creek drainage (CNDDDB 2016).⁶⁷

⁶⁵ One adult on 6/12/97 and 8 subadults on 6/18/97 at Antelope Creek at North Fork Antelope Creek (Fellers sites L-539 and L-543); and at the Antelope Creek and Indian Creek confluence (Fellers sites L-515A, L-515D, L-535), 2 adults and 38 subadults on 9/16/96; 1 adult and 51 subadults on 9/19/96; 3 adults, 19 subadults and 40 larvae on 6/10/97; 6 subadults on 6/11/97; 1 adult (CAS 226962) collected on 5/28/03; and 1 adult and 1 larva collected on 5/29/03 (CNDDDB 2016).

⁶⁶ Thirty larvae along Little Antelope Creek (Fellers site L-180; about 1.75 miles NE of Dewitt Peak, 1.8 miles SE of Dead Cow Flat), on 6/26/95; and 5 subadults along Cottonwood Creek (Fellers site L-179; about 1.3 air miles E of Shaw Creek Confluence, 1.4 miles NE of Dead Cow Flat), on 6/26/95 (CNDDDB 2016).

⁶⁷ One adult along Deer Creek near Highway 32 (site L-052B) on 7/7/94; 1 adult along Deer Creek, about 0.9 air miles ESE of Dead Horse Creek confluence (site L-052C), on 7/10/94; 2 adults along an unnamed tributary of Deer Creek (site L-058: 0.7 mile E of Little Pine Creek, 0.6 mile N of Pinnacle Peak), on 7/11/94; 1 adult along Deer Creek at Little Pine Creek (site L-061) on 7/11/94; 3 adults, 2 subadults and 90 larvae along Deer Creek at Little Pine Creek (site L-052D) on 7/12/94; 1 subadult and 5 larvae about 0.3 mile ENE of Deer Creek and a tributary confluence on 8/12/94; 6 adults in Deer Creek (site L-052B) on 8/12/94; 2 adults observed on 8/12/94 and 1 larva and 1 adult collected on 5/29/03 along Beaver Creek, at the crossing

Thomes Creek

Fellers (1996) observed *R. boylei* in the Thomes Creek drainage within the Mendocino National Forest, including two locations in Thomes Creek and in the Bennett Creek tributary. Bennett Creek was “notably good” area for reproduction, with 8 adults, 2,200 tadpoles, and 15 egg masses observed. There are collection records of single frogs from Thomes Creek and from the Willow Creek tributary, in August 2000 (CAS 2001).

Sacramento River

There are historical collections from 1924-1926 at the Sacramento River near Red Bluff and Tehama (UMMZ 2001; UCMVZ 2015).⁶⁸

Foothill yellow-legged frogs are now extirpated from the Sacramento River area in Tehama County (Hayes et al. 2013).

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 3 of 7 historical locations (43%) in eastern Tehama County. No foothill yellow-legged frogs were found during amphibian surveys from 1990-1998 of potentially suitable habitat on Lassen National Forest lands in eastern Tehama County (LNF and PNF 1999). Hayes et al. (2013) noted scattered collection records and sightings since 1980 in Lassen National Forest under surveys through the Federal Energy Regulatory Commission relicensing process. Small populations were documented in the 1990s and 2000s in Tehama County in the Battle Creek, Paynes Creek, Antelope Creek, Little Antelope Creek, Mill Creek, and Deer Creek drainages. The potentially large population in Red Bank Creek warrants further investigation.

Shasta County

Historical specimen collections were made from 1898-1981 throughout Shasta County in the upper Sacramento River drainage and in tributaries Ash Creek, Bars Creek, Cottonwood Creek, Dinner Gulch, Little Cow Creek, Low Pass Creek, McCloud River, Nosoni Creek, Pit River, Redding Creek, Salt Creek, Soda Creek, Squaw Creek and Stillwater Creek (CAS 2001; FMNH 2001; UMMZ 2001; USNM 2001; UCMVZ 2015).⁶⁹

of USFS Road 28N29 (site L-063); 1 gravid female collected from Deer Creek, 0.1 mile downstream from the Beaver Creek confluence, on 5/30/03; 1 adult collected from an unnamed tributary to Deer Creek (2.8 miles S of the USFS Road 28N29/Deer Creek junction), on 5/31/03; and 1 adult in a tributary 800' downstream from USFS Road 28N29 bridge crossing on 6/17/06 (CNDDDB 2016).

⁶⁸ Three frogs from 5 miles N of Tehama in the Sacramento River, on May 14, 1924 (UCMVZ 2015); 2 frogs from 8 miles N of Red Bluff and 2 frogs from 8 miles NE of Red Bluff, on April 5, 1928 (UCMVZ 2015); 1 frog from Bloody Island, along the Sacramento River, on May 28, 1926 (UCMVZ 2015); and 3 frogs from the Sacramento River below Red Bluff in August 1926 (UMMZ 2001).

⁶⁹ One frog from Baird (lower McCloud River, now flooded by Shasta Lake) in January 1884; 1 frog from the Sacramento River at Sims in July 1898; 3 frogs from McCloud River in June 1904; 2 frogs from Sweetbriar Camp in August 1907; 3 frogs from Redding in October 1911; 5 frogs from Middle Fork Cottonwood Creek (Divide 12 miles N of North Yolla Bolly Mt.) on May 12, 1926; 4 frogs from a tributary to the Sacramento River in August 1926; 1 frog from Redding Creek on August 16, 1941; 1 frog from a Sacramento River tributary 8 miles NW of Redding on November 13, 1945; 1 frog from Squaw Creek on October 14, 1950; 3 frogs from Bars Creek on October 15, 1950; 6 frogs from Low Pass Creek in 1950 and 1951; 3 frogs from Dinner Gulch in 1950 and 1951; 2 frogs from Ash Creek and Squaw Creek on August 11, 1951; 2 frogs from Salt Creek on June 20, 1952; 1 frog from the Pit River drainage, 8 miles NNW of Round Mountain, in April 1953; 4 frogs from Little Cow Creek (1 mile NE of Ingot) on May 3, 1953; 1 frog from Stillwater Creek (4

In the 1990s, small populations were reported in a dozen tributaries in the Sacramento River drainage, including Brandy Creek, Castle Creek, Cold Spring Gulch, Crystal Creek, Middle Fork Cottonwood Creek, Nosoni Creek, Prospect Creek, Sacramento River (near the delta of Dog Creek and Campbell Creek), Salt Creek, Squaw Creek and Sunday Gulch (CNDDDB 2016). Small populations were observed during May-September 1994 aquatic amphibian surveys of upper Sacramento River tributaries after the Cantera Bridge chemical spill, in Boulder Creek, Campbell Creek, Castle Creek, Little Slate Creek, Mears Creek, Middle Fork Castle Creek, Mosquito Creek, North Fork Salt Creek, North Fork Slate Creek, Sacramento River (S of McCardle Gulch at Sims), Shotgun Creek, Slate Creek and Whitlow Creek (Miller et al. 1994; CNDDDB 2016).

In the 2000s, small populations were reported in three dozen tributaries in the Sacramento River drainage: Backbone Creek, Baldwin Creek, Barney Gulch, Bear Gulch, Beegum Creek, Boulder Creek, Brandy Creek, Chain Gang Gulch, Cline Gulch, Clover Creek, Cornish Creek, Dry Creek, Duncan Creek, East Fork Clear Creek, East Fork Duncan Creek, Flat Creek, Flume Creek, Grizzly Creek, Hooten Gulch, Horse Creek, Madison Canyon, McCloud River, Mears Creek, Motion Creek, Nawtawaket Creek, Old Cow Creek, Ripgut Creek, Sacramento River (at Soda Creek), Salt Creek, Sawpit Gulch, Shotgun Creek, Soda Creek, South Cow Creek, Squaw Creek, Sugarpine Canyon, Susanville Canyon, Whiskey Creek and Willow Creek (CNDDDB 2016).⁷⁰ There were known *R. boylei* populations in the Pit River and its tributary Deep Creek (in the PG&E Pit 4 reach) and the species was expected to occur in the Pit 3 reach, between Lake Britton and Pit 4 Dam (FERC 2001).

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 3 of 14 historical locations (21%) in Shasta County. However, small populations were documented in more than three dozen tributaries to the Sacramento River in the 2000s, particularly in the vicinity of Dog Creek and Campbell Creek, and in Willow Creek and its tributaries Crystal Creek, Clear Creek and Mill Creek.

Northern Coastal California

The largest foothill yellow-legged frog populations in California are in the north coast range, with healthy populations scattered throughout the region. The strongholds for the

miles WSW of Bella Vista) on May 3, 1953; 1 frog from Low Pass Creek (15.3 miles E of Redding) in June 1953; 1 frog from Squaw Creek on August 3, 1963; 1 frog from Squaw Creek on September 2, 1966; 1 frog from Squaw Creek on May 18, 1968; 1 frog from Soda Creek (2.4 miles NE of Castle Crag Siding) on November 14, 1969; 7 frogs from Nosoni Creek at Gilman Road (tributary to McCloud arm of Shasta Lake) in September 1970; 1 frog from a tributary to McCloud arm of Shasta Lake (Gilman Road) in March 1977; and 1 frog from Pit River at Deep Creek on June 20, 1981 (CAS 2001; FMNH 2001; UMMZ 2001; USNM 2001; UCMVZ 2015).

⁷⁰ Significant (but small) populations were in Boulder Creek (5 adults and 25 metamorphs on 9/27/04 and 11 adults, 27 juveniles and 1 larva during 3 visits on 6/26, 8/31 and 10/5/05); Dead Horse Creek (15 adults, 5 juveniles and 15 tadpoles on 8/1/02); McCloud River near Tuna Creek crossing (frogs on 6/4/07; 5 frogs on 7/31/07; and 11 frogs on 9/11/07); and South Cow Creek (14+ frogs on 9/2/03) (CNDDDB 2016). somewhat larger populations were observed in the Sacramento River in the vicinity of the delta of Dog Creek and Campbell Creek (~50 subadults on 10/13/02 and 20-30 juveniles on 9/26/06); and in Willow Creek and tributaries Crystal Creek, Clear Creek and Mill Creek (10 adults and ~50 larvae on 5/1 and 5/5/03; 23 adults and 207 metamorphs in 9/04; 19 adults, 107 juveniles and 156 tadpoles on 6/2, 6/15, 6/28, 8/30, 8/31 and 10/4/05) (CNDDDB 2016).

species are in the Smith River; Red Cap Creek tributary of the Klamath; South Fork Trinity River; North, Middle and South Forks of the Eel River; Redwood Creek; coastal tributaries in Mendocino County; and Russian River tributaries. However, only 6 sites in northern California have large populations (estimated populations exceeding 100 adult frogs), with an additional 9 sites having moderately large (> 50 adult frogs) (Lannoo 2005). There have been documented declines in the northern coastal California region. Jennings and Hayes (1994) found that the species had been lost from 39 of 165 historical sites (24%) in the north coast of California. For the population with the longest term monitoring record on the University of California's Angelo Reserve, the population has been in decline over the last ten years (Peek and Kupferberg 2016).

Lake County

There are historical collection records and recent documentation of *R. boylei* populations within Lake County from the Eel River, Cache Creek and Putah Creek drainages, as well as tributaries to Clear Lake. The single *R. boylei* specimen from Lake County analyzed by Lind et al. (2011) was from northern Lake County in a tributary of the Eel River, in the North Coast hydrologic region. Other drainages in Lake County (Cache Creek, Putah Creek) in the Sacramento hydrologic region may have affinity with the Upper Sacramento River population, but are discussed here.

Eel River

Small numbers of frogs were collected in the 1970s and 1980s from the Bear Creek, Rice Creek and Welch Creek tributaries of the Eel River (CNDDDB 2016; UCMVZ 2015).⁷¹

Small numbers of frogs were observed and collected in the 1990s in Eel River tributaries above and below Lake Pillsbury, including Bear Creek, Corbin Creek, Parramore Creek, Rice Fork and Soda Creek (CNDDDB 2016; UCMVZ 2015).⁷² Fellers (1996) observed "relatively high numbers" of *R. boylei* in the Rice Fork of the Eel River (8-10 adults, 15 subadults and over 500 tadpoles).

Small populations of frogs were observed and collected in the 2000s in many Eel River tributaries above and below Lake Pillsbury, including Alder Creek, Asbill Creek, Benmore Creek, Copper Butte Creek, Corbin Creek, Dashiell Creek, Eel River, Horse Creek, Rice Fork, Skeleton Creek and Soda Creek; moderate populations (25-49 adults) were documented in 2004 in Berry Creek, Hummingbird Creek and Thistle Glade Creek; and a moderately large population (50-99 adults) in 2004 in Rattlesnake Creek (CNDDDB

⁷¹ Two frogs from Bear Creek Station on May 13, 1972; 4 frogs from Bear Creek Station on October 15, 1972; 2 frogs from Erickson Ridge, on Towhead Flat, on October 15, 1972; 1 frog from Upper Lake/Lake Pillsbury Road on April 21, 1973; and 9 frogs from the junction of Rice Creek and Bear Creek on August 19, 1986 (CNDDDB 2016; UCMVZ 2015).

⁷² One subadult and 3 egg masses observed in Corbin Creek (Fellers site Y-829) on 6/6/95; 11 adults, 1 subadult, 512 larvae and 1 egg mass observed in the Rice Fork (Fellers site Y-836) on 6/20/95; 3 adults collected from the Rice Fork on 7/6/95; 1 subadult collected from "The Slides" on 5/11/96; 1 subadult collected from Soda Creek on 5/11/96; 4 adults, 20 subadults and 55 larvae observed in Rice Fork (Fellers site Y-836) on 6/9/96; 1 juvenile collected from Parramore Creek on 9/14/99; and 1 adult collected from Bear Creek on 9/14/99 (CNDDDB 2016; UCMVZ 2015).

2016).⁷³ In Soda Creek (Fellers site Y-806), Fellers noted a total of 72 adults, 117 subadults, 21,109 larvae and 18,603 egg masses observed over 13 years from 1995-2008 (surveyed every year except 2007) (CNDDDB 2016).

Clear Lake tributaries

There are historical collection records from 1939-1967 in the Kelsey Creek and Adobe Creek drainages tributary to Clear Lake (UCMVZ 2015).⁷⁴ “Moderately abundant” numbers of frogs were observed in the McDowell Creek tributary in April 1956 (CNDDDB 2016).

Small numbers of frogs were observed and collected from 1997-2000 in the Clear Lake tributaries Adobe Creek, East Fork Middle Creek, Highland Creek, Kelsey Creek, Middle Creek and Panther Creek (a tributary of Scotts Creek) (CNDDDB 2016).⁷⁵ There are no known more recent observations from any tributaries of Clear Lake.

Cache Creek

There are historical collection records from 1937-1972 in the Cache Creek drainage, including North Fork Cache Creek, Seigler Canyon Creek, tributaries to Indian Valley Reservoir and Kilpepper Creek (CNDDDB 2016; UCMVZ 2015).⁷⁶ “Moderately abundant” numbers of frogs were observed in the Seigler Canyon Creek tributary in Lower Lake in April 1956 (CNDDDB 2016).

Observations and collections of small numbers of frogs were made in the 1990s in the North Fork Cache Creek drainage, including tributaries David Creek, Grizzly Creek, Harley Gulch, Spanish Creek and Wolf Creek (CNDDDB 2016).⁷⁷

⁷³ Two adults and 17 egg masses in Rice Fork (Fellers site Y-836) on 4/9/00; 1 adult in Asbill Creek on 4/7/01; 7 frogs in Skeleton Creek on 6/15/04 and 19 frogs on 6/16/04; 1 frog in Copper Butte Creek on 6/17/04; 25 frogs in Berry Creek at Eel River on 6/17/04; 33 frogs in Hummingbird Creek on 6/18/04; 27 frogs in Thistle Glade Creek on 6/19/04; 5 frogs in the Eel River (Trout Creek to Corbin Creek) on 7/29/04; 1 frog on 7/30/04 and 14 frogs on 7/31/04; 18 frogs in Corbin Creek on 7/31/04; 29 frogs in the Eel River and Horse Creek on 7/30/04; 54 frogs in Rattlesnake Creek on 8/1/04; 7 frogs in Alder Creek at Eel River on 6/22/05; 12 frogs in Dashiell Creek at Eel River on 6/26/05; 13 frogs in Benmore Creek on 6/27/05; and 6 frogs in Soda Creek (Fellers site Y-806) on 6/27/05 (CNDDDB 2016).

⁷⁴ One frog from Hwy. 29 (2.6 miles S of Kelseyville) on March 22, 1939; 1 frog from Forest Lake Resort on July 29, 1945; 3 frogs from 3 miles N and 1 mile W of Cobb Mt. on April 16, 1955; 1 frog from 3 miles N and 1 mile W of Cobb Mt. on April 14, 1956; and 4 frogs from the Highland Creek tributary on May 6, 1967 (UCMVZ 2015).

⁷⁵ One larva collected from Middle Creek (just below Rancheria Road) on 7/8/97; 5 adults, sub-adults and larvae collected from Middle Creek (Fellers site Y-805) on 5/14/97; 4 adults collected from Highland Creek on 7/11/97; 1 juvenile collected from East Fork Middle Creek on 5/25/99; 1 adult collected from Middle Creek (Fellers site Y-805) on 5/26/99; 1 frog observed in Kelsey Creek during May-June 1999; 1 frog observed in the Mayacamas Mountains in the upper Adobe Creek drainage during May-June 1999; 1 juvenile collected from East Fork Middle Creek on 9/13/99; and 20 adults, 14 subadults and 125 larvae observed along Panther Creek (Fellers site P-460) on 8/2/00 (CNDDDB 2016).

⁷⁶ One frog from 3.5 miles E of Bartlett Springs on August 13, 1937; 2 frogs from State Hwy. 20 (1.5 miles W of Bridge 14-12) on September 22, 1962; 2 frogs from Complexion Spring (a tributary to Indian Valley Reservoir) on October 14, 1972; 7 frogs from Kilpepper Creek (5.2 miles W of Barkersville) on October 14, 1972; and 2 frogs from North Fork Cache Creek near the confluence with Bartlett Creek on October 14, 1972 (CNDDDB 2016; UCMVZ 2015).

⁷⁷ One egg mass was observed along Spanish Creek (Fellers site Y-801) on 4/25/95; 5 adults and 92 larvae were observed along North Fork Cache Creek near Bartlett Creek (Fellers site Y-850) on 7/10/95; 1 adult was collected from Harley Gulch on 3/27/97; 3 adults were collected from Harley Gulch on 4/25/97; 7 adults

Fellers documented small populations in the early 2000s in Wolf Creek and Grizzly Creek (CNDDDB 2016).⁷⁸ “Good” *R. boylei* populations were observed in the early 2000s in the remote, upper sections of the North Fork Cache Creek (J. Olmstead, pers. comm., 2002). Hothem (2007) and Hothem et al. (2009) documented that *R. boylei* was present in Harley Gulch.

Putah Creek

There are historical collection records from 1919-1974 in the Putah Creek, including tributaries Butts Creek, Dry Creek and Hunting Creek (CAS 2001; CNDDDB 2016; UCMVZ 2015).⁷⁹ Foothill yellow-legged frogs were reported in “sparse numbers” in Coyote Creek along state route 53 in April 1956 (CNDDDB 2016).

Observations were made in the 1990s and 2000s of small populations in the Putah Creek tributaries Big Canyon Creek, Coyote Creek, Harbin Creek and Hunting Creek (CNDDDB 2016).⁸⁰ The species was reported to be present in April 2001 in “low numbers” along Putah Creek near the confluence with Coyote Creek, and to occur in Coyote Creek (Lake County 2008). Foothill yellow-legged frogs were reported to be “common” within the University of California McLaughlin Reserve, in the Hunting Creek and Knoxville Creek tributaries to Putah Creek above Lake Berryessa (UC 2009).

Recent status: In the North Coast hydrologic region, small populations were documented in the 2000s in many Eel River tributaries, with significant populations in Berry Creek, Hummingbird Creek, Soda Creek and Thistle Glade Creek, and a moderately large population in Rattlesnake Creek. Small populations were found from 1997-2000 in a half dozen Clear Lake tributaries. In the Sacramento River hydrologic region, small populations were documented in the 1990s and early 2000s in the North Fork Cache Creek drainage and in Putah Creek tributaries.

and several larvae were observed in Grizzly Creek (Fellers site Y-808) on 5/1/97; 3 adults were collected from Spanish Creek on 5/12/97; 15 adults and several tadpoles were observed in Grizzly Creek on 5/13/97; 1 juvenile, 1 subadult and 1 larva were collected from Quartz Canyon and Wolf Creek to Salt Lick Canyon (Fellers site P-461) on 5/14/97; 1 adult was collected from Harley Gulch on 3/11/98; 1 adult was collected from Harley Gulch on 3/16/98; 2 adults were observed in Grizzly Creek (Fellers site P-463) on 4/1/98; 3 adults were observed in David Creek on 8/4/98; 1 adult was observed in David Creek on 8/12/98; and 1 subadult and 1 juvenile were collected from Wolf Creek (Fellers site P-461) on 5/24/99 and 5/25/99 (CNDDDB 2016).

⁷⁸ Seven adults, 53 subadults and 350 larvae in Wolf Creek (Fellers site P-461) on 8/3/00; and 2 adults, 9 subadults and 3 larvae observed in Grizzly Creek upstream from the Cache Creek confluence (Fellers site P-463) on 8/4/00 (CNDDDB 2016).

⁷⁹ One frog from Castle Rock Springs in April 1919; 4 frogs from Hunting Creek (Hildebrand Ranch, Morgan Valley) on January 15, 1941; 1 frog from Dry Creek near Lower Lake on July 30, 1943; 1 frog from Hunting Creek on September 23, 1962; and 12 frogs from Butts Creek on September 29, 1974; and 1 frog from Middletown on an unknown date (CAS 2001; CNDDDB 2016; UCMVZ 2015).

⁸⁰ One adult collected from Big Canyon Creek on 6/21/94; 3 adults and hundreds of tadpoles observed in Harbin Creek and an unnamed tributary on 5/31/99; 1 adult and 27 metamorphs in Harbin Creek and an unnamed tributary on 9/12/99; 4 adults, 5 subadults and 6 larvae along Hunting Creek (Fellers site P-470) on 8/6/00; 1 adult in Harbin Creek on 8/20/06; and 2 tadpoles in Coyote Creek NE of Middletown on 8/24/07 (CNDDDB 2016).

Mendocino County

Eel River

There are historical collection records from 1911-1985 throughout the Eel River drainage in Mendocino County, including the Eel River, Elkhorn Creek, Fox Creek, Garcia Creek, Kenny Creek, Long Valley Creek, McKinley Creek, Middle Fork Eel River, North Fork Eel River, Outlet Creek, South Fork Eel River and Tenmile Creek (CAS 2001; CMNH 2001; LSUMNS 2001; SDNHM 2001; UMMZ 2001; USNM 2001; CNDDDB 2016; UCMVZ 2015). Relative abundance was indicated by collections of large numbers of frogs; for example 20 frogs taken from the Middle Fork Eel River (3 miles S of Covelo) in 1913 (UCMVZ 2015); and 38 frogs (33 collected in a single day) from the South Fork Eel River (just NW of Leggett) in 1978 (CMNH 2001).

Small populations were documented in the 1990s throughout the Eel River drainage in Mendocino County, including Bar Creek, Beaver Creek, Black Butte River, Buck Rock Creek, Burns Creek, Middle Fork Eel River, North Fork Eel River, Poor Man's Creek, Pothole Creek, Salmon Creek, Soda Creek, South Fork Bear Creek, South Fork Eel River, Trout Creek, Walters Creek, Whitney Creek and Williams Creek (Feller 1996; USDA and USDI 1996; CAS 2001; USNM 2001; CNDDDB 2016). Significant populations were documented along the South Fork Eel River (and Fox and McKinley Creeks) from 1992-1994 (1,292 egg masses cumulatively); the Middle Fork Eel River (near the Middle Fork-North Fork confluence) in 1993 (100 adults and 858 tadpoles observed during a 5-day survey); and in Burns Creek in 1998 (63 adults and hundreds of tadpoles during observations from May-Sept) (CNDDDB 2016). The species was reported to be regularly observed in the 1990s by State Department of Forestry personnel in the South Fork Eel River, East Branch North Fork Eel River, Standley Creek and Wildcat Creek (D. Matson, pers. comm., 2001).

Small numbers of frogs were documented in the 2000s at a few locations along the South Fork Eel River and Trout Creek (CNDDDB 2016; UCMVZ 2015), with a very large population documented from 2002-2008 in the South Fork Eel River at Fox and McKinley Creeks (CNDDDB 2016)⁸¹ - this is part of the population whose long term trends (1992-2011) are reported in Kupferberg et al. (2012).

Usal Creek

There was a historical collection specimen from Usal Creek in June 1897 (USNM 2001).

Tenmile River

There are historical collection records from the Tenmile River drainage in 1899 and 1909 (USNM 2001; UCMVZ 2015).⁸²

⁸¹ At the South Fork Eel River and Fox and McKinley Creeks, there were 29 known breeding sites along S. Kupferberg's study reach; 125 larvae were collected on 7/28/02-7/29/02; the population was approximately 600 frogs (ages not given) in 2006; 332 juveniles and 38 tadpoles were sampled in 2006; and 159 juveniles and 252 tadpoles were sampled in 2008 (CNDDDB 2016). At several sites along Trout Creek, at the Eel River, 10 frogs were observed on 6/21/05; a single adult was observed on 6/25/16. (CNDDDB 2016).

⁸² A single frog was collected from Cahto (just W of Laytonville) in May 1889 (USNM 2001); and 13 frogs from Sherwoods in August 1909 (UCMVZ 2015).

The species was reported to be regularly observed in the Ten Mile River in the 1990s by State Department of Forestry personnel (D. Matson, pers. comm., 2001).

Noyo River

There are historical collection records from the Noyo River in 1927 and tributary Sixteen Gulch from 1984-1985 (CNDDDB 2016; UCMVZ 2015).⁸³

The species was reported to be regularly observed in the Noyo River drainage in the 1990s by State Department of Forestry personnel (D. Matson, pers. comm., 2001). Foothill yellow-legged frogs were observed during the mid-1990s in the South Fork of the Noyo River, including the tributaries North Fork of the South Fork Noyo River, Parlin Creek and Brandon Gulch (DFFP 2001). A small population was documented in the 1990s in the tributary Willits Creek (CNDDDB 2016).

Big River

There are historical collection records from 1902-1982 in the Big River drainage, including Big River, James Creek and North Fork Big River (CAS 2001; UMMZ 2001; UCMVZ 2015).⁸⁴

The species was reported to be regularly observed in the Big River drainage in the 1990s by State Department of Forestry personnel (D. Matson, pers. comm., 2001). The species was observed during the mid-1990s in the North Fork of the Big River, including the tributaries Two Log Creek, Chamberlain Creek, and James Creek (DFFP 2001; CNDDDB 2016). Foothill yellow-legged frogs were observed in the North Fork Big River in 1996 and 1997 and in Big River near Mendocino Woodlands and near James Creek in 1999 (DFFP 2001). A small population was documented in the South Fork Big River in the 2000s (CNDDDB 2016).

Navarro River

There are historical collection records from 1906-1975 throughout the Navarro River drainage, including Christine Creek, India Creek and Navarro River (CAS 2001; FMNH 2001; UCMVZ 2015).⁸⁵

⁸³ Six frogs were collected from Eagle's Nest, Noyo River, on July 20, 1927 (UCMVZ 2015); and 1 frog from Sixteen Gulch 1984-85 (CNDDDB 2016).

⁸⁴ Three frogs from Big River in 1902 (UCMVZ 2015); 1 frog from 7 miles SW of Willits in November 1940 (CAS 2001); 1 frog from the North Fork Big River (at junction with Big River) on April 23, 1950 (UCMVZ 2015); frogs from James Creek on March 7, 1965 (UCMVZ 2015); 5 frogs from James Creek (14 miles W of Willits) in June 1967 (UMMZ 2001); and 2 frogs from the North Fork Big River on August 28, 1982 (UCMVZ 2015).

⁸⁵ One frog from India Creek W of Ukiah in August 1906 (CAS 2001); 1 frog from Navarro River near Dimmick Pond in May 1931 (CAS 2001); 1 frog from Christine Creek (6 miles NW of Philo) in May 1936 (FMNH 2001); 2 frogs from Boonville-Albion Road (0.5 mile W of Navarro) in March 1939 (CAS 2001); 1 frog from Mailliard Ranch (4 miles W of Yorkville) in December 1940 (CAS 2001); 1 frog from Boonville in August 1946 (CAS 2001); 2 frogs from Dimmick Grove State Park in 1950 (UCMVZ 2015); 3 frogs from 7 miles SE of Boonville in 1950 (UCMVZ 2015); 1 frog from 0.6 mile W of Navarro in 1955 (UCMVZ 2015); 1 frog from Navarro Creek W of Navarro in 1959 (UCMVZ 2015); 3 frogs along Hwy. 128 in Dimmick State Park on March 6, 1965 (UCMVZ 2015); larvae from Fishrock Bridge (7 miles SE of Boonville) in July 1975 (USNM 2001); and 2 frogs from the Navarro River in September 1975 (UCMVZ 2015).

The species was reported to be regularly observed in the 1990s in the Indian Creek tributary of the Navarro River by State Department of Forestry personnel (D. Matson, pers. comm., 2001). Small populations were also documented in the 1990s in Flynn Creek and the North Branch of the North Fork Navarro River (CNDDDB 2016). Small populations were documented in the 2000s in Anderson Creek (CNDDDB 2016).

Garcia River

There are historical collection records from 1931-1971 in the Garcia River drainage (CAS 2001; UCMVZ 2015).⁸⁶

The species was reported to be regularly observed in the Garcia River drainage in the 1990s by State Department of Forestry personnel (D. Matson, pers. comm., 2001). Small populations were documented in the 1990s in Garcia River and tributary Mill Creek (CNDDDB 2016).

Gualala River

There are historical collection records from the Gualala River in Mendocino County in 1913 and 1974 (CNDDDB 2016; UCMVZ 2015).⁸⁷

The species was reported to be regularly observed in the 1990s in the Gualala River drainage by State Department of Forestry personnel (D. Matson, pers. comm., 2001). Small populations were documented in the 2000s in the North Fork Gualala River and a tributary (CNDDDB 2016).

Small coastal rivers

Historical collection records in small coastal rivers include: 4 frogs from 2 miles S of Westport on May 9, 1959 (UCMVZ 2015); 1 frog from 5.1 miles S of Fort Bragg on Hwy. 1 in May 1971 (UCMVZ 2015); 1 frog from Jug Handle Creek on May 29, 1971 (CNDDDB 2016); 7 frogs from the Albion River in 1897 (CAS 2001); and 3 frogs from Elk in July 1946 (LSUMNS 2001).

The species was reported to be regularly observed in the 1990s in Wages Creek, DeHaven Creek, Usal Creek and the Albion River by State Department of Forestry personnel (D. Matson, pers. comm., 2001).

Russian River

There are historical collection records from 1893-1981 in the Russian River drainage in Mendocino County, including Forsythe Creek, Orr Creek, Pieta Creek, Reeves Canyon, Robinson Creek and Russian River (CAS 2001; FMNH 2001; USNM 2001; UCMVZ

⁸⁶ A single frog from the middle part of the Garcia River in October 1931 (CAS 2001); 1 frog from Ornbaun Springs near Yorkville in March 1939 (CAS 2001); 3 frogs from the Garcia River (10 miles SW of Point Arena) in December 1941 (CAS 2001); and 1 frog from Manchester Beach State Park in 1971 (UCMVZ 2015).

⁸⁷ Seven frogs from Gualala in 1913 (UCMVZ 2015); and 1 frog from the Gualala River in 1974 (CNDDDB 2016).

2015).⁸⁸ The species was reported to be “moderately abundant” in Pieta Creek in March-April 1956 (CNDDDB 2016).

Small populations were documented in the 1990s in Dry Creek, Edwards Creek, Jakes Creek, Pieta Creek and Salt Springs Creek (CNDDDB 2016). Small populations were documented in the 2000s in Dry Creek, Howell Creek, Parsons Creek and the Russian River; and a moderate population was documented in Hensley Creek in 2006 (CNDDDB 2016).

Recent status: Fellers (1996) found excellent habitat and foothill yellow-legged frogs present at 21 of 36 sites (58%) surveyed within the Mendocino National Forest in 1995.⁸⁹ The species was widespread in the 1990s and early 2000s throughout Mendocino County, in the Eel River, Tenmile River, Noyo River, Big River, Navarro River, Garcia River, Gualala River and Russian River drainages. Significant populations were found in the South Fork Eel River through 2008 and the Hensley Creek tributary of the Russian River in 2006.

Humboldt County

Klamath River

Historical collection records from the Klamath River in Humboldt County include Tectah Creek in 1947 (CAS 2001), and the Klamath River at Aikens Creek in 1976 (UCMVZ 2015).

Small populations were documented in 1990 along the Klamath River in Hoopa Valley and in tributary Tectah Creek (CNDDDB 2016). Unknown numbers of foothill yellow-legged frogs were documented in 1994 in timber harvest areas along the Klamath River and tributary Middle Fork Roach Creek (CNDDDB 2016). There were more than 2,000 observations of foothill yellow-legged frogs during 1994-1995 fisheries surveys within the Red Cap Creek watershed, a tributary to the Klamath River south of Orleans; foothill yellow-legged frog was the most frequently seen vertebrate from Schnable Diggings to the confluence of Red Cap Creek with the Klamath River, with the majority of observations near the mouth of Red Cap Creek (Mollier and Norman 1994; Cyr and Norman 1995; USDA 1995a, 1999b). The species was also encountered at 3 of 30 non-mainstem associated sites in 30-minute time-constrained searches in the Red Cap Creek watershed.

⁸⁸ Nineteen specimens (6 adults, 13 juveniles) were collected from Ukiah on a single day in October 1893 (USNM 2001); 2 frogs from Reeves Canyon (near Lake Leonard, 10 miles NW of Ukiah) in 1922 (UCMVZ 2015); 5 frogs from Forsythe Creek on June 4, 1939 (UCMVZ 2015); 1 frog from along Low Gap Road (about 3 miles W of Ukiah) in March 1951 (CAS 2001); 1 frog from Robinson Creek (5.5 miles SSW of Ukiah) on May 20, 1952 (UCMVZ 2015); 1 frog from 9.4 miles SSE of Willits in 1952 (UCMVZ 2015); 1 frog from Orr Creek (5 miles W of Ukiah) in March 1953 (FMNH 2001); 1 frog plus larvae from Forsythe Creek (Reeves Canyon) in August 1956 (CAS 2001); 2 frogs from near Hwy. 128 (3 miles WNW of McDonald) on March 19, 1965 (UCMVZ 2015); 1 juvenile from Robinson Creek in April 1979; and 1 juvenile from Robinson Creek in April 1981 (CAS 2001).

⁸⁹ This was a high proportion of sites occupied, given that most of the sites where frogs were not found were outside the geographical or elevational range of the species, and surveys were conducted during high water runoff, making surveying difficult and increasing the likelihood of frogs being overlooked.

Trinity River

There are historical collection records from 1941-1977 in the Trinity River drainage (tributary to the Klamath River), including Boise Creek, Brannan Creek, Trinity River and Willow Creek (CAS 2001; UCMVZ 2015).⁹⁰ Small numbers of frogs were observed during 1984-1985 sampling by Welsh and Lind in the tributaries Ammon Creek, Coon Creek and Fourmile Creek (CNDDDB 2016).

In the 1990s there were “numerous” sightings of foothill yellow legged frog within the tributary Horse Linto Creek and its drainages, but no sightings in the tributaries Mill Creek or Tish Tang Creek (USDA 2000). A foothill yellow-legged frog population was studied in the lower South Fork Trinity River from 1991-1993 by Redwood Sciences Lab (USDA 1999a). Fellers documented small populations from 1994-1995 at two sites in Horse Linto Creek (CNDDDB 2016). The Six Rivers National Forest detected *R. boylei* during a 1994 stream inventory in a 1,000 m section of Grouse Creek, tributary to South Fork Trinity River (USDA 1995d). A large population was documented in the South Fork Trinity River from 1992-2007 (CNDDDB 2016).⁹¹

Small populations were documented in 2000 in the South Fork Trinity River tributaries Grapevine Creek, Grouse Creek, Madden Creek and Sims Creek (CNDDDB 2016). Welsh et al. (2010) documented relatively high numbers of foothill yellow-legged frogs from 2000–2003 in western and northern headwater tributaries of the South Fork Trinity River. A significant population was documented in Madden Creek through 2007 (CNDDDB 2016).⁹²

Redwood Creek

There are historical collection records from Redwood Creek in 1942 and 1955 (UMMZ 2001; UCMVZ 2015). The species was apparently abundant, as evidenced by collection of 28 specimens from Redwood Creek over three days in September 1942 (UCMVZ 2015). Adult frogs were observed in 1974 in the tributaries Cloquet Creek and Lostman Creek, within Redwood National and State Parks (RNSP 2001). Anderson (1988) found frogs believed to be *R. boylei* during intensive sampling of twelve streams in the Redwood Creek basin in 1981; frogs were found in 14 of 112 tributaries surveyed, including Captain Creek, Copper Creek, Fern Prairie Creek, Joplin Creek, Lacks Creek, Lake Prairie Creek, Miller Creek, Panther Creek, Roaring Gulch Creek, Santa Fe Creek, Simon Creek, Sweathouse Creek and Tossup Creek.

⁹⁰ One frog from a small stream along Hwy. 96 in the Hoopa Reservation in July 1941 (CAS 2001); 1 frog near Boise Public Camp (1.76 miles W of Willow Creek) in 1945 (UCMVZ 2015); 1 frog from 3 miles S of Willow Creek in March 1947 (CAS 2001); single frogs from Boise Creek (about 2 miles W of Willow Creek) in February 1947 and in April 1948 (UCMVZ 2015); 2 frogs from 3 miles N of Willow Creek August 1949 (UCMVZ 2015); 1 frog from Willow Creek in August 1949 (UCMVZ 2015); 1 frog from Brannan Creek (2 miles W of Willow Creek) in September 1949 (UCMVZ 2015); 1 frog from Hwy. 96 (4.9 road miles S of Weitchpec) in March 1971 (UCMVZ 2015); and 1 juvenile from Boise Creek Camp off Hwy. 299 in March 1977 (CAS 2001).

⁹¹ In the South Fork Trinity River (between Surprise Creek and Madden Creek; Fellers site R-118); 175 frogs of unknown gender found 7/92; 102 frogs of unknown gender found 9/92; unknown number of frogs found in 1993 by Welsh and Lind; 6 adults and 1 juvenile observed by Fellers on 5/30/00; during Fellers surveys over 13 survey days from 1994-2000 and 2002-2007, 106 adults, 944 subadults, 28,764 larvae and 11,329 egg masses observed cumulatively (CNDDDB 2016).

⁹² Fellers (site R-121) observed a cumulative total of 146 adults and 80 subadults in Madden Creek over 12 years: 1994-2000, 2002-2004 and 2006-2007 (CNDDDB 2105).

The amphibian populations within Redwood National and State Parks were thought to be in relatively good shape in the 1990s (G. Fellers, pers. comm., as cited in RNSP 2001), with adult *R. boylei* observed in numerous tributaries throughout the Redwood Creek drainage. Fellers documented small to moderate populations from 1993-1997 at several dozen sites along Redwood Creek and tributaries Boyes Creek, Bridge Creek, Copper Creek, Forty-four Creek and Tom McDonald Creek (CNDDDB 2016). In 1992, “numerous” frogs were documented in Redwood Creek between Slide Creek and Bridge Creek (RNSP 2001). The species was observed at various locations along Redwood Creek on at least 41 occasions from 1993 to 1997 (RNSP 1997, 2001). From 1993 to 2000, small numbers of frogs were also observed in many of the tributaries to Redwood Creek, including Bridge, Brown, Cloquet, Cole, Copper, Coyote, Devils, Elam, Emerald, Forty-four, Hayes, McArthur, Rodgers, Tom McDonald, and Tossup Creeks (RNSP 2001). On a single day in November 1998, Parks biologists located 64 yellow-legged frogs along Redwood Creek, from the Bridge Creek confluence to the Tom McDonald Creek confluence (RNSP 2001).

A Parks survey on a single day in September 2000 along Redwood Creek from Forty-four Creek confluence to Bond Creek confluence located 89 yellow-legged frogs (2001). Small populations were documented from 2004-2005 in the tributaries Bridge Creek, Garrett Creek, May Creek and Pilchuck Creek (CNDDDB 2016).

The California Department of Fish and Wildlife conducted single-pass egg mass surveys in lower Redwood Creek during breeding season for foothill yellow-legged frog in 2011 and 2012. In 2011 CDFW surveyed 5.4 km of Redwood Creek above tidal influence, and detected 19 egg masses/km; in 2012 CDFW surveyed 14.2 km of Redwood Creek above tidal influence, and detected 13 egg masses/km (M. Van Hattem, pers. comm., 2016).

Mad River

There are historical collection records from 1897-1985 within the Mad River drainage, including the Mad River, Maple Creek and North Fork Mad River (CAS 2001; UCMVZ 2015).⁹³ The species was apparently abundant, as evidenced by collection of 17 frogs from Maple Creek in 1942 (UCMVZ 2015).

Foothill yellow-legged frogs were known to occur in the 1990s along the Mad River and the lower portions of major tributaries to the Mad River (USDA 1999b). Small populations were documented from 1990-1991 along the Mad River and tributaries Black Dog Creek and Maple Creek; with “many” juveniles and adults reported in 1990 in the Mad River (5 miles SE of Korb) and an unnamed tributary (6 miles SE of Korb) (CAS 2001; CNDDDB 2016). Most of these populations were threatened by active and planned timber sales.

⁹³ Two frogs from the Mad River in July 1897 (CAS 2001); 6 frogs from Cobbs in July 1936 (UCMVZ 2015); 17 frogs from Maple Creek (1 mile W of junction with Mad River) in September 1942 (UCMVZ 2015); 1 frog from 12 miles S of Korb in August 1949 (UCMVZ 2015); 7 frogs from the North Fork Mad River (7.5 miles ENE of Blue Lake) in July 1955 (UMMZ 2001); and 1 frog from the junction of Butler Valley Road and Maple Creek Road in January 1985 (UCMVZ 2015).

A small population was documented in the Mad River (near Blue Slide Creek) in 2004 (CNDDDB 2016).

The California Department of Fish and Wildlife conducted single-pass egg mass surveys in the lower Mad River during breeding season for foothill yellow-legged frog in 2011, 2012, 2015 and 2016. In 2011 CDFW surveyed 13.5 km of the Mad River below the Mad River Hatchery, and detected 59 egg masses/km; in 2012 CDFW surveyed 14.7 km in the same reach and detected 13 egg masses/km (M. Van Hattem, pers. comm., 2016). The 2015 and 2016 survey results were comparable (M. Van Hattem, pers. comm., 2016).

Eel River

There are numerous historical collection records from 1910-1989 throughout the Eel River drainage in Humboldt County, including Ascaphus Creek, Bear Creek, Cuddeback Creek, Devil's Elbow Creek, Eel River, Fish Creek, Fort Steward Creek, Redwood Creek and South Fork Eel River (Green 1986; CAS 2001; CMNH 2001; FMNH 2001; UMMZ 2001; USNM 2001; UCMVZ 2015).⁹⁴

Small numbers of frogs were observed from 1994-1995 in the Eel River (at Bear Creek, Bull Creek, and W of Shively) and in the tributaries Bull Creek, Canoe Creek and Twin Creek (CNDDDB 2016), many within proposed timber harvest plan areas. The Six Rivers National Forest conducted riparian amphibian surveys in 1995 on many tributaries within the Eel River watershed; the foothill yellow-legged frog appeared to be widely-distributed in the North Fork Eel River watershed, and "present" in the Main Eel (USDA and USDI 1996). The species was not found in the Eel River Delta, the South Fork Eel River or the Middle Fork Eel River during the 1995 surveys (USDA and USDI 1996). A significant population (50-80 metamorphosed juveniles) was documented in the South Fork Eel River (between Gould Grove and Gould Bar) on 10/7/99 (CNDDDB 2016).

Small populations were documented in from 2000-2007 in Albee Creek, Bull Creek, Carson Creek, Chadd Creek, Cuneo Creek, Mill Creek, North Creek and the South Fork Eel River (at Gould Bar, and at Bull Creek) (CNDDDB 2016).

⁹⁴ One frog from Cuddeback Creek in September 1910 (UCMVZ 2015); 11 frogs from Carlotta in May 1911 (CAS 2001); 7 frogs from Alton in May 1911 (CAS 2001); 2 frogs from Elinor in May 1911 (CAS 2001); 1 frog from Carlotta in July 1923 (UCMVZ 2015); 1 frog from Scotia before 1933 (USNM 2001); 6 frogs from the South Fork Eel River (3 miles S of Garberville) in October 1933 (FMNH 2001); 18 frogs from South Fork Eel River (3 miles S of Garberville) in October 1933 (18) (UCMVZ 2015); 18 frogs from 1 mile NW of Pepperwood in July 1935 (CAS 2001); 3 frogs from Fish Creek (2 miles S of Miranda) in 1936 (FMNH 2001); 1 frog from 1 mile E of Alton in March 1938 (UCMVZ 2015); 1 frog from Fort Steward Creek in June 1938 (CAS 2001); 1 frog from Charles B. Alexander Grove (Fish Creek) in March 1939 (CAS 2001); 1 frog from 10 miles S of Hartsook in November 1940 (CAS 2001); 3 frogs from Ascaphus Creek (0.5 mile N of Holmes) in November 1941 (CAS 2001); 1 frog from 4 miles N of Garberville in November 1941 (CAS 2001); 4 frogs from 2 miles S of Miranda in August 1950 (UMMZ 2001); 1 frog from 2.6 miles N of the south entrance to Richardson's Grove in March 1951 (UCMVZ 2015); single frogs from 6 miles NW of Dyerville in February 1952 and in March 1952 (UCMVZ 2015); 2 frogs from 11.1 miles SSE of Dyerville in March 1952 (UCMVZ 2015); 1 frog from 2 miles S of Miranda in 1955 (CMNH 2001); 4 frogs from 1 mile S of Pepperwood in June 1955 (UMMZ 2001); 3 frogs from Devil's Elbow Creek (4.8 miles E of Weott) in July 1961 (UCMVZ 2015); a larval specimen from the Bear Creek drainage (1.5 miles S of Pepperwood) in August 1975 (FMNH 2001); 7 frogs from the Eel River at Myer's Flat in October 1982 (UCMVZ 2015); 6 postmetamorphs from Myer's Flat (Green 1986); and larvae specimens collected from Redwood Creek (2 miles W of Garberville) in June 1989 (USNM 2001).

The California Department of Fish and Wildlife conducted single-pass egg mass surveys in 2012 in the Eel River mainstem, the South Fork Eel River and in tributary Bull Creek, during breeding season for foothill yellow-legged frog. In the Eel River mainstem CDFW surveyed 11 km and found only 1 egg mass (0.09 egg masses/km); in the South Fork Eel River CDFW surveyed 4.65 km and detected 191 egg masses/km; in the tributary Bull Creek CDFW surveyed 3.5 km and detected 101 egg masses/km (M. Van Hattem, pers. comm., 2016).

Van Duzen River

There are historical collection records from 1930-1967 in the Van Duzen River tributary of the Eel River (CAS 2001; SDNHM 2001; UCMVZ 2015).⁹⁵

An unknown number of frogs were found in 1992 and 1993 in a timber harvest area near the mouth of the Van Duzen River (CNDDDB 2016). Fellers documented a small population in the Van Duzen River near Grizzly Creek in 1995 (CNDDDB 2016). Foothill yellow-legged frogs were present during 1995 surveys in the Van Duzen River (USDA and USDI 1996). The species was documented in the Yager Creek tributary during surveys for timber harvest plans on private lands (USDA and USDI 1998).

A small population was documented in 2007 in the tributary Flannigan Creek (CNDDDB 2016).

The California Department of Fish and Wildlife conducted a single-pass egg mass survey in the Van Duzen River during breeding season for foothill yellow-legged frog in 2011. In 2011 CDFW surveyed 2.7 km of the Van Duzen River above the Highway 101 bridge, and detected only 9 egg masses/km (M. Van Hattem, pers. comm., 2016).

Jacoby Creek

A single foothill yellow-legged frog was found in Jacoby Creek in 2007 (CNDDDB 2016).

Mattole River

There are a handful of historical collection records from 1894-1959 in the Mattole River drainage (UCMVZ 2015).⁹⁶

Foothill yellow-legged frogs were reported to be regularly observed in the 1990s in the Mattole River by State Department of Forestry personnel (D. Matson, pers. comm., 2001). Significant populations were documented in 1992 within timber harvest areas in the tributaries Mill Creek and Conklin Creek (CNDDDB 2016).⁹⁷ Welsh and Hodgson (1997) sampled 15 tributaries of the Mattole River watershed in 1995 and 1996 and

⁹⁵ Three frogs from the Van Duzen River (SE of Strong's Station) in November 1930 (UCMVZ 2015); 4 frogs from 1 mile E of Carlotta in September 1942 (UCMVZ 2015); 1 frog from the Van Duzen River (30 miles E of Alton) in September 1960 (CAS 2001); 1 frog from the Van Duzen River in May 1961 (CAS 2001); and 4 frogs from the Van Duzen River (3 miles east of Carlotta) in December 1967 (SDNHM 2001).

⁹⁶ Four frogs from the Mattole River in June 1894 and June 1898 (UCMVZ 2015); single frogs from 10 road miles S and 13 road miles S of Honeydew in May 1956 (UCMVZ 2015); and 1 frog from Petrolia in August 1959 (UCMVZ 2015).

⁹⁷ In Mill Creek (2 miles SSW of Petrolia), 36 frogs found from April-May 1992; in Conklin Creek (N of Burgess Ridge), 15 frogs trapped from April-May 1992 (CNDDDB 2016).

observed 119 adults and 347 larvae. The species was common throughout the Mattole River watershed in 1998 (Welsh and Hodgson 2011).

Recent status: The species was well-distributed in the 1990s throughout Humboldt County watersheds, including the Klamath River, Trinity River, Redwood Creek, Mad River, Eel River, Van Duzen River and Mattole River drainages; with notable populations in the Red Cap Creek tributary of the Klamath, the Horse Linto Creek tributary of the Trinity, Redwood Creek and numerous tributaries, both the North Fork and South Fork of the Eel River, and the Mattole watershed. There is a paucity of surveys or records from the 2000s in many of these watersheds.

Trinity County

Trinity River

Large numbers of *R. boylei* were collected from 1932 to 1973 from the Trinity River and many of its tributaries within Trinity County, including Bell Creek, Brown's Creek, Canyon Creek, Coffee Creek, Eagle Creek, East Fork of the North Fork Trinity River, East Fork Trinity River, East Fork Weaver Creek, Little Bidden Creek, Little Browns Creek, Mumbo Creek, New River, Panther Creek, Reddings Creek, Rush Creek, Stuart Fork Trinity River and Stetson Creek; as well as the South Fork Trinity River and its tributaries Carr Creek, Corral Creek, East Fork of the South Fork Trinity River, Hayfork Creek, Kerlin Creek, Monroe Creek, Philpot Creek, Rattlesnake Creek, Salt Creek and Wilson Creek (Slevin 1928; Bury 1969; USDA 1999b; CAS 2001; CMNH 2001; LSUMNS 2001; UMMZ 2001; CNDDDB 2016; UCMVZ 2015). The large numbers of frogs collected indicated high densities at many locations; collections of 5 to 10 adult and juvenile frogs at single locations were common (Bury 1969); and very large numbers of frogs were collected from the South Fork Trinity River in 1932; and in 1973 and 1974 from the South Fork tributaries Hayfork Creek, Philpot Creek and Salt Creek (UCMVZ 2015).

In the 1990s, Wilson et al. (1991) found foothill yellow-legged frogs in the lower reaches of the mainstem Trinity River, above the confluence with the North Fork of the Trinity River. The foothill yellow-legged frog was documented to occur in the 1990s in small clumped populations along the mainstem Trinity River (between Lewiston Dam and the North Fork Trinity River) and tributaries Bell Creek, Davis Creek and Ripple Creek (USDA 1999c; CNDDDB 2016); a significant population was found on the mainstem east of Hawkins Bar in 1994 (CNDDDB 2016).⁹⁸ Small populations were documented in the 1990s in the South Fork Trinity River and tributaries Big Creek, Bridge Gulch, East Fork of the South Fork, Hayfork Creek and Prospect Creek (USDA 1999c; CNDDDB 2016). Small populations were documented in the 1990s in the North Fork Trinity tributary Rattlesnake Creek (CNDDDB 2016). Foothill yellow-legged frogs were rare in Trinity River mainstem in the 12 river miles below Lewiston Dam because suitable breeding areas had been reduced 95% compared to pre-dam conditions, with most frogs clustered in the limited areas of suitable habitat (Ashton et al. 1998; USFWS et al. 1999).

In the 2000s, small populations were documented at several locales along the mainstem Trinity River and tributaries Little Bidden Creek and West Weaver Creek (CNDDDB 2016). Small populations were documented in the 2000s in the North Fork Trinity River

⁹⁸ Along the mainstem Trinity River, E of Hawkins Bar (Fellers site R-120B), 13 adults and 976 subadults were observed on 9/15/94; just 5 egg masses were observed here on 6/5/95 (CNDDDB 2016).

tributaries East Twin Creek, Rattlesnake Creek, Underwood Creek and West Twin Creek (CNDDDB 2016). Small populations were documented in the 2000s in the South Fork Trinity River and numerous tributaries, including Barker Creek, Big Creek, Bridge Gulch, Butter Creek, Carrier Gulch, Chancelulla Creek, Cold Springs Creek, Ditch Gulch, Eltapom Creek, Grapevine Creek, Hayfork Creek, Hells Half Acre Creek, Jim's Creek, Monroe Creek, Olsen Creek, Orchard Gulch, Philpot Creek, Potato Creek, Salt Creek, Shiell Gulch Twenty-Two Creek and Walker Creek (CNDDDB 2016). A significant population was documented in the South Fork Trinity River (between Surprise Creek and Madden Creek) through 2007 (CNDDDB 2016).⁹⁹

Salmon River

There is a historical collection record from 1971 in Swift Creek, in the South Fork Salmon River drainage, in Trinity County (UCMVZ 2015).

Mad River

There are historical collection records from 1932-1942 in the Mad River and tributary Olson's Creek, in Trinity County (UCMVZ 2015). Very large numbers of frogs were collected in 1932 in the Mad River (UCMVZ 2015).

Eel River

There is a historical collection record from 1913 in the North Fork of the Middle Eel River, in Trinity County (UCMVZ 2015).

Wicktor and Craven (1996) surveyed 13 streams in the North Fork of the Eel River watershed in 1995, sampling for herpetofauna in 118 reaches, 55 of which were surveyed to protocol (USDA 1995b) and 63 of which had incidental sightings. Incidental sightings were also made in the main stem of the North Fork Eel River and in Tub Creek. *Rana boylei* were found in 9 of these streams, including the North Fork Eel River, West Fork, Tub Creek, Bluff Creek, Bradburn Creek, Cottonwood Creek, Cox Creek, Kettenpom Creek and Salt Creek; 64 adult frogs were located, with the mainstem (10 adults, 262 sub-adults and 1 tadpole), West Fork (14 adults and 76 sub-adults), and Salt Creek (15 adults, 1 sub-adult, 23 tadpoles and 1 larvae) reaches containing significant populations. Fellers (1996) observed *R. boylei* at six locations in the Middle Fork Eel River drainage, including the North Fork of the Middle Fork Eel, Rattlesnake Creek, and Balm of Gilead Creek tributaries, within the Mendocino National Forest; the North Fork of the Middle Fork Eel was a notably good area for reproduction, with 286 tadpoles observed (Fellers 1996; CNDDDB 2016).

Cottonwood Creek

There is a historical collection record from 1946 in Harrison Gulch, an upper tributary to the Middle Fork of Cottonwood Creek, which drains east to the Sacramento Valley (UCMVZ 2015).

⁹⁹ In the South Fork Trinity River (between Surprise Creek and Madden Creek; Fellers site R-118), Fellers observed a cumulative total of 106 adults, 944 subadults, 28,764 larvae and 11,329 egg masses over 13 survey days from 1994-2000 and 2002-2007 (CNDDDB 2016).

Recent status: The species was common in the 1990s in the North Fork Eel River and Middle Fork Eel River drainages; the status in the 2000s is unknown. Despite declines in the Trinity River mainstem below Lewiston Dam, foothill yellow-legged frogs continued to be widespread throughout the Trinity River basin through the 2000s, particularly in the South Fork Trinity River and numerous of its tributaries.

Siskiyou County

Klamath River

There are historical collection records from 1935-1972 in the Klamath River drainage in Siskiyou County, including Beaver Creek, Clear Creek, Dillon Creek, Ditch Creek, Grant Creek, Grider Creek, Klamath River, Little Bogus Creek, Little Shasta River, O'Farrell Gulch, Salmon River, Seiad Creek and Swillup Creek (CAS 2001; LSUMNS 2001; UMMZ 2001; CNDDDB 2016; UCMVZ 2015).¹⁰⁰

Foothill yellow-legged frogs were reported to be "fairly common" in the 1990s along the banks of the Klamath River, including the vicinity of the mouth of Rogers Creek, north of Somes Bar (KNF 1999).

Sacramento River

There are historical collection records from 1953 in the Sacramento River near the Mt. Shasta Fish Hatchery in Siskiyou County (UMMZ 2001; UCMVZ 2015).¹⁰¹

A small number of frogs were reported from 1994-2003 in tributaries to the Sacramento River in Siskiyou County, including Big Spring Creek, Little Castle Creek, Ney Springs Creek, North Fork Sacramento River and South Fork Sacramento River (CNDDDB 2016).¹⁰²

¹⁰⁰ Two frogs from Little Shasta River (10 miles E of Montague) on May 21-22, 1935 (UCMVZ 2015); 5 frogs from Clear Creek (3 miles W of Klamath River) on June 23-25, 1935 (UCMVZ 2015); 1 frog from Klamath River (1.5 miles S of Clear Creek) on June 27, 1935 (UCMVZ 2015); 2 frogs from Little Bogus Creek (4 miles NE of Ager) on November 4, 1951 (UCMVZ 2015); 1 frog from the Salmon River (4.5 miles NW of Forks of Salmon) in July 1955 (UMMZ 2001); 1 frog from Swillup Campground in October 1959 (CAS 2001); 4 frogs from the Salmon River (at junction near Klamath River) on September 1, 1961 (UCMVZ 2015); 1 frog from Ditch Creek at Cottonwood Creek (1 mile north of Hornbrook) in June 1963 (LSUMNS 2001); 1 frog from Dillon Creek (on Highway 98 between Willow Creek and Happy Camp) in August 1968 (UMMZ 2001); 2 frogs from Beaver Creek (about 0.25 mile E of Klamath River) on November 15, 1969 (UCMVZ 2015); 1 frog from Klamath River (2.5 miles N of Ti Bar, Hwy. 96) on November 16, 1969 (UCMVZ 2015); 1 egg mass from Beaver Creek (USFS Camp, N of Hwy. 96) on April 26, 1970 (UCMVZ 2015); 2 larvae from Grider Road (2.9 miles E of Grider Creek) on April 15, 1970 (UCMVZ 2015); 1 frog from Seiad Creek Road (5.7 road miles N of Seiad Valley) on March 23, 1971 (UCMVZ 2015); 1 frog from Hwy. 5 at Hilt on March 23, 1971 (UCMVZ 2015); 3 frogs from Salmon River (Hwy. 93, 3.9 miles SE of Somes Bar Bridge) on March 24, 1971 (UCMVZ 2015); 1 frog from Hwy. 93 and Grant Creek (about 8.1 miles NW of Forks of Salmon) on March 24, 1971 (UCMVZ 2015); 1 frog from junction of O'Farrell Gulch and Hwy. 93 (4.5 road miles SE of Forks of Salmon) on March 24, 1971 (UCMVZ 2015); and 1 frog from 1 mile SE of Hilt (off Hwy. 5) on March 19, 1972 (UCMVZ 2015).

¹⁰¹ Thirteen frogs collected from 2 miles S of the Mt. Shasta Fish Hatchery in May 1953 (UMMZ 2001); and 1 frog from 2 miles S of Mt. Shasta Fish Hatchery on June 7, 1953 (UCMVZ 2015).

¹⁰² During a 1994 stream survey for the Cantara Spill Recovery and Restoration Program, 2 adults observed in Ney Springs Creek (SE of Lake Siskiyou) and 1 adult observed in Little Castle Creek (SW of Dunsmuir). One adult observed at the confluence of the North and South Forks of the Sacramento River on 6/12/96; 3 adults along the South Fork Sacramento River (0.9 mile SW of confluence of South and North Forks of Sacramento River) on 7/29/96; 1 adult just S of Little Castle Creek (at Railroad Park pond, 2 miles SSW of

Recent status: Unknown. Other than reports of small numbers of frogs from 1994-2003 in tributaries to the Sacramento River, there are no known recent data for Siskiyou County.

Del Norte County

Rogue River

There is a historical collection record from the East Fork Illinois River, in Del Norte County just south of the Oregon State Line, in 1935 (UCMVZ 2015).¹⁰³

Welsh and Lind found a single frog in Bybee Gulch, a tributary of the East Fork Illinois River in Del Norte County just south of the Oregon State Line, during sampling in 1984-1985 (CNDDDB 2016).¹⁰⁴

Smith River

There are historical collection records from 1940-1986 in the Smith River drainage, including Mill Creek, Patrick's Creek, Smith River and South Fork Smith River (CAS 2001; CMNH 2001; FMNH 2001; RNSP 2001; SDMNH 2001; UMMZ 2001; UTA 2001; UCMVZ 2015).¹⁰⁵

The foothill yellow-legged frog was considered common in the 1990s across the Six Rivers National Forest (which encompasses portions of Del Norte, Humboldt, Trinity, and Siskiyou counties); the species was found in most tributaries to the Smith River and was "very abundant" in the Middle Fork Smith River (USDA 1999b, 1995c). However, the Six Rivers National Forest sampled 10 creeks in the Smith River basin from 1990-1992 and *R. boylei* was located in only 3 tributaries; Muzzleloader Creek and Hurdygurdy

Dunsmuir) on 5/23/01; 2 adults along South Fork Sacramento River (1.5 miles W of Lake Siskiyou) on 6/9/01 and 1 adult in August 2001; 1 adult along the South Fork Sacramento River (about 1.7 miles W of Lake Siskiyou) in August 2001; 1 frog in Big Spring Creek (near Mount Shasta State Fish Hatchery) on 9/4/01; and 1 adult along the South Fork Sacramento River (about 1.7 miles W of Lake Siskiyou) on 5/30/03 (CNDDDB 2016).

¹⁰³ Three frogs were collected from the East Fork Illinois River, 0.25 mile south of the Oregon State Line, on July 15, 1935 (UCMVZ 2015).

¹⁰⁴ One frog found by Welsh and Lind in Bybee Gulch (4 Miles N of Chicago Peak) during time-constrained sampling of a 1 mile reach during in 1984-85 (CNDDDB 2016).

¹⁰⁵ One frog from 8 miles NE of Crescent City (Smith River) in November 1940 (CAS 2001); 3 frogs from Mill Creek Park (Mill Creek) in February 1942 (CAS 2001); 1 frog from a small stream near Siskiyou Mountain Camp along Hwy. 199 (Smith River drainage) in September 1949 (FMNH 2001); 18 frogs from the Smith River (8 miles NE of Crescent City) in August 1950 (UMMZ 2001); 2 frogs from 2.25 miles N and 6.5 miles E of Crescent City on February 5, 1952 (UCMVZ 2015); 2 frogs from Mill Creek, 5 miles ENE of Crescent City, on July 22, 1952 (UCMVZ 2015); 3 frogs from Jedediah Smith State Park along the Smith River, in September 1953 (FMNH 2001); 2 frogs from the Smith River on an unknown date before 1955 (CAS 2001); 5 frogs from Patrick's Creek along the Smith River in 1955 (CMNH 2001); 2 juveniles and 2 juvenile/adult frogs from 12 miles NE of Crescent City, 2 miles W of Gasquet school, in June 1956 (CAS 2001); 1 frog from Jedediah Smith State Park in November 1961 (UCMVZ 2015); 3 frogs from the Smith River (2 miles W of Gasquet) in August 1975 (FMNH 2001); 2 frogs from the South Fork Smith River (South Fork Road milepost 3.05 and milepost 9.35, 3.6 road miles NW of Steven Memorial Bridge) on January 9, 1985 (UCMVZ 2015); 2 frogs from Panther Flat Campground in the Six Rivers National Forest (Smith River) in July 1985 (UTA 2001); and 2 frogs from Hiouchi and 2 frogs from Patrick in August 1986 (SDNHM 2001). A single frog was observed in Mill Creek at the confluence with the Smith River, within Redwood National and State Parks, in 1973 (RNSP 2001).

Creek West in the South Fork tributary, and Patrick Creek in the Middle Fork tributary (USDA 1995c, 1999b).

Small populations were documented from 1991-1995 in the Smith River drainage, including Hurdygurdy Creek, Hutsinpillar Creek, Mill Creek, Muzzleloader Creek and Patrick Creek, as well as along the Smith River at Cedar Creek, Clarks Creek and near Hiouchi Bridge (CAS 2001; RNSP 2001; CNDDDB 2016).¹⁰⁶ Fellers had continued observations of unspecified numbers of *R. boylei* in Hurdygurdy Creek during surveys from 1986-1993 and 2002-2007 (CNDDDB 2016).

Wheeler et al. (2006) documented a heavily used *R. boylei* breeding site in the lower 2 km of Hurdygurdy Creek from 2002-2005; with 129 male and 34 female frogs observed from 2002-2005 and 36 oviposition sites in 2003 and nine oviposition sites in 2004.

Klamath River

There were observations in 1990 of three frogs at two localities near Omegaar Creek, a tributary of the lower Klamath River in Del Norte County (CNDDDB 2016).¹⁰⁷

Recent status: Unknown. The species was abundant in the Middle Fork Smith River and there were observations of small populations throughout the Smith River drainage in the 1990s; other than the apparently significant population documented in Hurdygurdy Creek from 2002-2005, there are no known recent data.

Southern Sierra Nevada

The southern Sierra Nevada includes drainages of the Sierra foothills in Mariposa, Madera, Fresno, Tulare and Kern counties. There are no known *R. boylei* records from Kings County or eastern Merced County, which are considered to be outside the historic range for this species (Jennings 1996).

There has been confusion over identification of a few yellow-legged frog specimens collected from the southern Sierra Nevada. While the foothill yellow-legged frog (*R. boylei*) inhabits low to moderate elevation streams, from sea level to 1,830 m (6,000 feet)

¹⁰⁶ In Hurdygurdy Creek and Hurdygurdy Creek tributaries near Horse Flat, 1 adult female was collected on 4/24/91 (CAS 2001), and 1 frog was observed in May or June 1991 near Horse Flat; in Muzzleloader Creek (W of Ship Mountain), 1 frog was found during surveys from 7/29/91-8/7/91, and 1 male and 3 females were found during surveys from 7/14/92-7/23/92; in Patrick Creek, 2 frogs were found between 8/19/91 and 9/9/91; in the Smith River and Hutsinpillar Creek (North Bank Road and Highway 101) "several" frogs were found in 1994; in Mill Creek at the Smith River (Fellers site R-102B; Stout Grove, Jedediah Smith Redwoods State Park), Fellers observed 13 adults, 2 subadults and 35 larvae on 8/3/94, and observed 1 adult and 1 subadult on 5/31/95; along the Smith River at Cedar Creek (Fellers site R-109A; between Stout Grove and Hiouchi), Fellers observed 1 adult, 1 subadult and 100 larvae on 8/4/94; at three sites in Clarks Creek and along the Smith River (about 1.8 miles NE of the NE edge of Crescent City), Fellers observed 1 adult and 7 subadults on 7/5/94 at site R-110, 1 adult and 3 subadults on 7/5/94 at site R-111, 7 adults, 15 subadults and 4 larvae on 7/5/94, and 7 subadults on 5/31/95 at site R-109B; and along the Smith River (Fellers site R-109C; about 0.25 mile SW of Hiouchi Bridge and 0.5 mile N of Jedediah Smith Campground), 9 adults, 3 subadults and 2 larvae were observed on 8/26/94 (CNDDDB 2016). Two *Rana boylei* were observed along a creek tributary to the Smith River, within Redwood National and State Parks (RNSP 2001).

¹⁰⁷ One frog observed about 1.5 air miles S of Omegaar Creek at Klamath River (about 3.3 miles WNW of Blue Creek Campground); and 2 frogs observed in Hoopa Valley (about 1.5 miles SSE of Klamath River at Omegaar Creek, about 2.7 miles NW of Blue Creek Campground), on an unspecified date in 1990 (CNDDDB 2016).

in California (Stebbins 1985; Stebbins and McGinnis 2012), the mountain yellow-legged frog (*Rana muscosa*) occurs in the southern Sierra at elevations mostly above 1,820 meters (6,000 feet), with the lowest documented occurrence of this species in the Sierra foothills at 1,044 m (3,425 feet) (Jennings and Hayes 1994; USFS 2000). Historically, the ranges of *R. muscosa* and *R. boylei* abutted each other at mid-elevations in the southern Sierra Nevada, and the two species were sometimes found in close proximity to each other in the same drainages (Zweifel 1955). Though the two species can appear morphologically similar, genetic analysis can differentiate between the species (e.g. Lind et al. 2011; Poorten et al. 2013).

Between 1911 and 1920, Grinnell and Storer (1924) surveyed terrestrial vertebrates at 41 sites along a transect that stretched from the western foothills of the Sierra Nevada through Yosemite National Park. Grinnell and Storer (1924) found foothill yellow-legged frogs at 7 sites throughout the western foothill portion of their transect. Grinnell and Storer (1924) described the foothill yellow-legged frog as widespread and “fairly common” along Smith Creek, occurring in “moderate numbers” along Blacks Creek, and observed “several” along Piney Creek (Storer field notes, 1915, 1916, as cited in Drost and Fellers 1994).

Moyle (1973) found *R. boylei* at only 30 of 95 sites (32%) sampled in the southern and central Sierra Nevada foothills (from the Yosemite area south) in 1970, and believed the species was declining at that time. Intensive surveys by David Graber in the 1980s of 15 different stream reaches throughout the southern Sierra Nevada foothills did not locate any surviving populations of *R. boylei* (D. Graber, Sequoia National Park, pers. comm., as cited in Drost and Fellers 1994, 1996).

Surveys of the southern Sierra foothills in 1993 by Fellers (1994) found only one subadult foothill yellow-legged frog south of Calaveras County, in spite of surveying 310 sites within the frog’s known range. Drost and Fellers (1994) re-surveyed 38 of 40 Grinnell and Storer collecting sites in the Yosemite section, and were unable to locate foothill yellow-legged frogs at any of the historical sites, despite careful searches. Drost and Fellers (1994, 1996) also searched 10 additional sites at other streams that offered suitable habitat within the elevational range where Grinnell and Storer found the species, without finding foothill yellow-legged frogs, even though these sites included a number of areas where *R. boylei* had been recorded in the past, including sites within Yosemite National Park and in tributaries of the Merced River below El Portal. Drost and Fellers (1996) concluded that *R. boylei* had been essentially extirpated from the southern Sierra Nevada. Additional surveys in the 1990s found no foothill yellow-legged frogs in Yosemite National Park, while searching suitable habitat in Yosemite Valley, Foresta, Wawona, and El Portal (Fellers and Freel 1995; Fellers 1997).

Moritz (2007) conducted a resurvey from 2003 through 2005 of amphibians at 7 sites within Yosemite National Park that were originally surveyed from 1911-1920 by Grinnell; the Moritz surveys also detected no foothill yellow-legged frogs.

Hayes et al. (2013) summarized the evidence that *R. boylei* is extirpated from Yosemite, Sequoia and Kings Canyon National Parks, and near extirpation in Sequoia and Sierra National Forests, with few remaining populations and limited distribution. The species is now nearly extirpated from the southern portion of its Sierra range. The few known remaining localities in the southern Sierra Nevada are small populations in Mariposa County (Merced River through 2009 and tributaries above Lake McClure: Bull Creek

through 2007 and Sherlock Creek through 2009) and eastern Fresno County (a tributary of the San Joaquin River, Jose Creek, through 2007); and two moderate populations in Tulare County (in tributaries of the North Fork Kern River and upper Kern River).

Southern Sierra National Forests and National Parks

Hayes et al. (2013) comprehensively reviewed the foothill yellow-legged frog's historical (prior to 1980) and recent (1980 to 2001) status across the two National Forests (Sierra and Sequoia) and three National Parks (Yosemite, Sequoia and Kings Canyon) that overlap with the southern Sierra Nevada *R. boylei* population. Although historical records imply that *R. boylei* populations were robust in this region until the 1960s and 1970s, Hayes et al. (2013) found fewer than 10 recent records from the Cosumnes River south, indicating the species is near extirpation in the southern portion of its Sierra range. A summary of the findings of Hayes et al. (2013), evaluated by National Forest and National Park:

Sequoia National Forest

Hayes et al. (2013) noted historical records from 1891-1970 in Sequoia National Forest and adjacent private lands, primarily in the Kern, Kings and Tule river systems, including from Angel Creek, Caliente Creek, Canebrake Creek, Cedar Creek, Clear Creek, Cottonwood Creek, Cowflat Creek, Deer Creek, Fay Creek, Kern River, Kings River, Middle Fork Tule River, North Fork Kern River, North Fork of the Middle Fork Tule River, Pechacho Creek, Salmon Creek, South Fork Kern River, Tehachapi Creek, Tejon Creek and White River. By the end of the 1970s, 17 foothill yellow-legged frog localities had been documented from the Sequoia National Forest, and 23 additional foothill yellow-legged frog localities were documented from drainages downstream or outside of Sequoia National Forest lands. An indication of declines was evident by the late 1970s.

Hayes et al. (2013) noted that no collections and very few sightings of foothill yellow-legged frogs exist for the Sequoia National Forest and vicinity from 1980 to the present. Lind et al. (2003b) indicated that none of 6 historical localities resurveyed on the Sequoia National Forest had foothill yellow-legged frogs. The two most recently occupied localities in the Sequoia National Forest consisted of tributaries of the North Fork Kern River, Ash and Jywood Creeks, which were surveyed multiple times from 1998-2003. Foothill yellow-legged frogs now have likely been extirpated from Ash Creek. Hayes et al. (2013) noted that foothill yellow-legged frogs in the Sequoia National Forest appear to be very few and limited in distribution, and may be near extirpation in the region.

Sierra National Forest

Hayes et al. (2013) noted historical records from 1916-1970 in Sierra National Forest and adjacent private lands in the Merced, San Joaquin and Kings river systems, including from Big Creek, Feliciana Creek, Kings River, Merced River, Middle Fork Kings River, Mill Creek and North Fork Kings River. By the end of the 1970s, 6 foothill yellow-legged frog localities had been documented from the Sierra National Forest, and 14 additional foothill yellow-legged frog localities were documented from drainages downstream or outside of Sierra National Forest lands. Indication of declines was evident by the late 1970s.

Hayes et al. (2013) noted that no collections and very few sightings of foothill yellow-legged frogs exist for the Sierra National Forest and vicinity from 1980 to the present. Lind et al. (2003b) indicated that none of the 6 historical localities on the Sierra National Forest had foothill yellow-legged frogs. The only drainage recently confirmed to have foothill yellow-legged frogs in the Sierra National Forest is Jose Creek, a tributary of the San Joaquin River that is isolated by the presence of upper Redinger Lake at its mouth. Hayes et al. (2013) noted that foothill yellow-legged frogs in the Sierra National Forest appear to be rare and limited in distribution, and may be near extirpation in the region.

Sequoia and Kings Canyon National Park

Hayes et al. (2013) noted 3 historical records of foothill yellow-legged frogs from Sequoia National Park from the 1930s, from the North Fork of the Kaweah River and Alder Creek. There were no existing historical records from Kings Canyon National Park.

Hayes et al. (2013) noted that there have been no collections or sightings of foothill yellow-legged frogs from Sequoia National Park from 1980 to present. The foothill yellow-legged frog has not been recorded in or near Sequoia or Kings Canyon National Parks for more than 30 years despite substantial searching, and is considered locally extinct (SKCNP 2001).

Yosemite National Park

Hayes et al. (2013) noted few historical records exist for foothill yellow-legged frogs in Yosemite National Park, and that Grinnell and Storer's (1924) transect was outside National Park boundaries and did not document the species within Yosemite National Park. Hayes et al. (2013) noted 1 historical record within Yosemite National Park, in 1948 along the Merced River (see the Mariposa County section below for additional historical records).

Hayes et al. (2013) noted that systematic surveys of the amphibian fauna of Yosemite National Park (Drost and Fellers 1994, 1996) and surveys conducted by National Park personnel through 2006 have also failed to find the species within Yosemite National Park. Hayes et al. (2013) concluded that foothill yellow-legged frogs are likely to be extirpated from Yosemite National Park.

Kern County

Kern River

There are historical collection records from 1891-1954 in the Kern River drainage, including the Kern River, South Fork Kern River and tributaries Canebrake Creek, Cowflat Creek and Fay Creek (UMMZ 2001; LSUMNS 2001; CAS 2001; USNM 2001; CNDDB 2016; UCMVZ 2015).¹⁰⁸

¹⁰⁸ Historical collection records include: 2 frogs (USNM #18951 and 18952) from Kern River at Old Kernville in June 1891 (now inundated by Isabella Lake); 1 adult (USNM #18950) collected from Canebrake Creek at Walker Pass Campground (elevation 5,000 feet) on July 4, 1891; 5 frogs from the Kern River 12 miles below Bodfish in June 1911; 30 frogs from the Kern River near Bodfish in June 1911; Fay Creek, 6 miles N of Weldon, in July 1911; 1 frog from Miracle Hot Springs on the Kern River in November 1938; 1 frog from Kern River at Kernville in April 1940; 1 frog from Kern Canyon E of Bakersfield in August 1940; 4 frogs and 32 tadpoles from 4-5 miles S of Glennville in August 1940; 2 frogs from 5 miles E of Onyx along South Fork

There are no known observations from the Kern River drainage in Kern County since 1954.

Tehachapi Creek

There are historical collection records from 1947-1963 in Tehachapi Creek (UCMVZ 2015).¹⁰⁹

There are no known observations from Tehachapi Creek since 1963.

Caliente Creek

There are historical collection records from 1952-1967 in Caliente Creek (UCMVZ 2015).¹¹⁰

There are no known observations from Caliente Creek since 1967.

Tejon Creek

There is a historical collection record of 8 frogs from the Tejon Creek drainage in 1875.

There are no known observations from Tejon Creek.

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at any of 15 historical locations in Kern County. There are no recent records from Kern County. The foothill yellow-legged frog is extirpated from Kern County.

Tulare County

Kern River

There are historical records from 1891 and 1953 in the Kern River (USNM 2001; UCMVZ 2015).¹¹¹

Kern River in August 1940; 2 frogs from 2 miles E of Onyx, in August 1949; 5 frogs from Cowflat Creek, Kern River Canyon, in April 1952; 1 frog from Miracle Hot Springs in April 1952; and 8 frogs from Canebrake Creek, and 1 frog from 9 miles ENE of Onyx in February 1954 (UMMZ 2001; LSUMNS 2001; CAS 2001; USNM 2001; CNDDDB 2016; UCMVZ 2015).

¹⁰⁹ Historical collection records include: 2 frogs from Tehachapi Creek in May 1947; 2 frogs from Tehachapi Creek, 6 miles NW of Tehachapi, in May 1952; 1 frog from Keene, along Tehachapi Creek, in June 1963; and 1 frog from the Keene Fire sub-station along Hwy. 58 in March 1970 (UCMVZ 2015).

¹¹⁰ Historical collection records include: 7 frogs from Caliente Creek 6 miles ESE of Caliente in May 1952; 5 frogs from Caliente Creek 8 miles E of Caliente in July 1952; 2 frogs from Caliente Creek 3 miles W of Paris-Lorraine in June 1967; and 4 frogs from 4.5 miles W of Paris-Lorraine in June 1967 (UCMVZ 2015).

¹¹¹ The U.S. National Museum of Natural History has 8 *Rana boylei* specimens collected from the upper Kern River, 25 miles above Kernville in July 1891 (USNM 2001). The U.C. Museum of Vertebrate Zoology has 7 *Rana boylei* specimens collected from Salmon Creek, in the Kern River Canyon, 2.5 miles SE of Fairview (elevation ~3200' feet) in April 1953 (UCMVZ 2015).

Fellers documented one moderate and one small population from 1998 through 2008 along the Rincon Trail in the Sequoia National Forest, in unnamed tributaries of the upper Kern River (CNDDDB 2016).¹¹²

Kaweah River

There are historical records from 1907-1970 in the Kaweah River drainage, including Alder Creek Reservoir, Cottonwood Creek, East Fork Kaweah River, Kaweah River, Little Deer Creek, North Fork Kaweah River and South Fork Kaweah River (Moyle 1972, 1973; CAS 2001; HMCZ 2001; CNDDDB 2016; UCMVZ 2015).¹¹³

There are no known observations in the Kaweah River drainage since 1970. Sequoia and Kings Canyon National Park noted that the species had not been recorded in or near Sequoia National Park for more than 30 years despite substantial searching, and was considered locally extinct (SKCNP 2001).

Deer Creek/White River

There are historical collection records from 1940 and 1970 in Deer Creek and tributary Tyler Creek, in Moore Creek, in Yokohl Creek and in the White River (Moyle 1972, 1973; UMMZ 2001; CNDDDB 2016).¹¹⁴

¹¹² The first population (Fellers site ID #s S-849 & S-849B) is about 3.6 miles W of Sherman Peak and about 4.5 miles ENE of Ida Lake, where Fellers observed 3 adults on 9/12/98; 11 adults on 6/4/99; 3 adults on 6/6/00; 1 adult on 5/31/03; 1 adult and 1 subadult on 6/28/05; 1 adult and 1 subadult on 6/29/05; 3 adults and 5 subadults on 5/15/06; 35 adults and 2 subadults on 5/25/07; and 6 adults and 12 subadults on 5/31/08 (CNDDDB 2016). The second population (Fellers site ID #s S-840) is about 2.7 miles SSE of Durwood Camp and about 4.7 miles ENE of Ida Lake, where Fellers observed 2 adults and 3 subadults on 8/20/98; 4 adults on 6/6/99; 1 adult, 1 subadult and 5,000 larvae on 6/6/00; 4 adults, 1 subadult and 30 larvae on 7/5/01; 5 adults and 2 subadults on 6/1/02; 24 adults and 22 subadults on 5/31/03; 6 adults and 2 subadults on 6/28/05; 3 adults and 1 larvae on 5/25/07; and 1 subadult on 5/31/08 (CNDDDB 2016).

¹¹³ Cornell University has a specimen collected from Giant Forest from 1907 (CU 2002). The California Academy of Sciences has specimens collected from the Kaweah River, Potwisha Camp, and Sequoia National Park in August 1941 (CAS 2001). The Harvard Museum of Comparative Zoology has 8 specimens collected from Giant Forest in Sequoia National Park (Little Deer Creek tributary to the Marble Fork Kaweah River) in August 1960 (HMCZ 2001). The U.C. Museum of Vertebrate Zoology has historical collection specimens from the Kaweah River drainage: 6 frogs from the North Fork Kaweah River (elevation ~2,000 feet) in July and August 1935; 1 frog from Alder Creek Reservoir (elevation ~1,700 feet) in August 1935; 1 frog from Cottonwood Creek 0.5 mile SE of Aukland (elevation ~1,300 feet) in June 1938; 1 frog from 6 miles NE of Three Rivers on March 29, 1952; and 2 frogs from 8.5 miles NW of Woodlake (elevation below 2,000 feet) in April 1952 (UCMVZ 2015). The U.C. Museum of Vertebrate Zoology also has frog specimens labeled as *Rana boylei* that were collected from elevations over 7,500 feet: 31 frogs from Quaking Aspen Meadow in July and August 1934; and 3 frogs from Long Meadow 3 miles NNE of Giant Forest in June 1955 (UCMVZ 2015); however, these specimens are likely mislabeled, since *Rana boylei* is not known to occur above 6,000 feet (Stebbins 1985; Stebbins and McGinnis 2012). Moyle documented foothill yellow-legged frogs from the Kaweah drainage during surveys from July 27 to September 4, 1970, including: South Fork Kaweah River; East Fork Kaweah River, approximately 8 miles ENE of Lake Kaweah; and North Fork Kaweah River, 2 miles W of Sequoia National Park (Moyle 1972, 1973; CNDDDB 2016).

¹¹⁴ The University of Michigan Museum of Zoology has *Rana boylei* specimens collected in August 1940: 4 adults and 7 tadpoles from Deer Creek 4 miles below the highway to Cal Hot Springs; and 7 adults and 44 tadpoles from the White River 20 miles SE of Porterville (UMMZ 2001). Moyle documented foothill yellow-legged frogs from the Deer Creek/White Creek drainages during surveys from July 27 to September 4, 1970, including: Deer Creek; Tyler Creek (tributary to Deer Creek); and Deer Creek approximately 1.5 miles W of California Hot Springs (Moyle 1972, 1973; CNDDDB 2016). Moyle also documented *Rana boylei* in several small streams in the low foothills of Tulare County, including: Moore Creek (W of Aukland); Yokohl Creek (E of Exeter); and White River in southern Tulare County (Moyle 1972, 1973; CNDDDB 2016).

There are no known observations in any of these drainage since 1970.

Tule River

There are historical records from 1952-1970 throughout the Tule River drainage including Middle Fork Tule River; North Fork Tule River, North Fork of Middle Fork Tule River, Rancheria Creek and Tule River (Moyle 1972, 1973; CNDDDB 2016; UCMVZ 2015).¹¹⁵

There are no known observations in the Tule River drainage since 1970, other than a report of a single frog along the North Fork Tule River in 2004 (CNDDDB 2016).¹¹⁶

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at any of 17 historical locations in Tulare County. The only documentation of foothill yellow-legged frogs in Tulare County since 1970 are two small and moderate populations in tributaries of the upper Kern River, and a report of a single frog in 2004 on the North Fork Tule River. The foothill yellow-legged frog is nearly extirpated from Tulare County.

Fresno County

Kings River

There are historical collection records from 1910-1970 in the Kings River drainage, including Big Creek, Kings River, Middle Fork Kings River, North Fork Kings River, Rush Creek, Sycamore Creek, Watts Creek, Watts Lake and White Deer Creek (Wright and Wright 1949; Moyle 1972, 1973; CAS 2001; CNDDDB 2016; UCMVZ 2015).¹¹⁷

¹¹⁵ The U.C. Museum of Vertebrate Zoology has *Rana boylei* specimens collected from the Tule River drainage in April 1952: 5 frogs from 5 miles ENE of Springville on the road to Camp Wishon (elevation ~4,000 feet); 7 frogs from 8.7 road miles ENE of Springville on the road to Camp Wishon; 1 frog from Camp Wishon; and 1 frog from 3.8 miles E of Springville (elevation ~3,500 feet) (UCMVZ 2015). The UCMVZ also has 1 frog collected from along Hwy. 190, 5.8 mi NE of Springville (elevation ~3,200 feet) in December 1970 (UCMVZ 2015). Moyle documented foothill yellow-legged frogs from throughout the Tule River drainage during surveys from July 27 to September 4, 1970, including: Rancheria Creek, tributary to Bear Creek, approximately 6 miles NNE of Springville; Middle Fork Tule River; North Fork Tule River; North Fork Tule River just N of Milo; and North Fork of Middle Fork Tule River about 0.2-1.5 miles WSW of Camp Wishon Forest Service Station (Moyle 1972, 1973; CNDDDB 2016).

¹¹⁶ G. Adest reported a single adult *Rana boylei* on August 29, 2004 along the North Fork Tule River, about 2 miles NE of Springville; this was on private land, with the population threatened by bullfrogs and unregulated water withdrawals (CNDDDB 2016).

¹¹⁷ The California Academy of Sciences has specimens collected from the Kings River Canyon in July 1910 (CAS 2001). The U.C. Museum of Vertebrate Zoology has specimens collected from the Kings River drainage: 14 frogs from the Sycamore Creek tributary at Dunlap in September and October 1916; 2 frogs from the Kings River at Minkler in October 1916; and 1 frog from the North Fork Kings River below Balch Camp in December 1970 (UCMVZ 2015). Wright and Wright (1949) recorded capturing *Rana boylei* in 1942 along Sycamore Creek, an intermittent stream of the lower foothills, in an area now submerged by Pine Flat Reservoir. Moyle (1972, 1973) reported the stream above the reservoir containing only bullfrogs (*Rana catesbeiana*), with a few *Rana boylei* found far up a small tributary. Moyle documented foothill yellow-legged frogs from tributaries to Pine Flat Reservoir during surveys from July 27 to September 4, 1970, including Big Creek, Rush Creek, White Deer Creek, Watts Lake and Watts Creek (Moyle 1972, 1973; CNDDDB 2016). The California Academy of Sciences has specimens collected from the Middle Fork Kings River in September 1953: 1 frog from 5 miles upstream of the junction with North Fork in September 1953 and 3 frogs from Mill Flat Campground (CAS 2001).

There are no known observations in the Kings River drainage since 1970. Sequoia and Kings Canyon National Park noted that the species had not been recorded in or near Kings Canyon National Park for more than 30 years despite substantial searching, and was considered locally extinct (SKCNP 2001).

San Joaquin River

There are historical collection records from 1945-1953 in the Big Creek and Dry Creek tributaries of the San Joaquin River and in Huntington Lake (TMM 2001; UCMVZ 2015).¹¹⁸

Fellers located small populations of foothill yellow-legged frogs from 1994-2007 in Jose Creek, a tributary of the San Joaquin which feeds Redinger Lake (CNDDDB 2016).¹¹⁹ Surveys by PG&E (2000) were unable to locate the species on the San Joaquin River or in Millerton Lake or Kerckhoff Lake.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 4 of 9 historical locations (44%) in eastern Fresno County. There are no recent sightings from the Kings River basin and the only known locality in eastern Fresno County with *R. boylei* is the Jose Creek tributary of the San Joaquin River.

Madera County

San Joaquin River

There are historical records from 1951-1970 in the Willow Creek and South Fork Willow Creek tributaries of the San Joaquin River and in Little Finegold Lake (Moyle 1972, 1973; PG&E 2000; CNDDDB 2016; UCMVZ 2015).¹²⁰

Tietje and Vreeland (1997) failed to find *R. boylei* from 1987-1990 at the San Joaquin Experimental Range, in the Cottonwood Creek tributary to the San Joaquin River, using timed searches and pitfall traps. PG&E (2000) noted no recent records of the species in Willow Creek below Bass Lake; abundant predatory fish and bullfrogs in Willow Creek

¹¹⁸ The U.C. Museum of Vertebrate Zoology has collection specimens from the San Joaquin drainage: 4 frogs from 1.3 miles N of Big Creek along Hwy. 168 in July 1953; and 6 frogs from 1 mile W of Tollhouse, on the Dry Creek tributary, in September 1953 (UCMVZ 2015). The Texas Memorial Museum has a *Rana boylei* specimen collected in July 1945 from Huntington Lake, in the upper San Joaquin River drainage (TMM 2001).

¹¹⁹ In the Sierra National Forest, from just downstream of Italian Bar Road upstream to Jose Basin Road: at the Mill Creek and Jose Creek confluence (Fellers site ID S-715), 9 adults, 3 juveniles and 2 egg masses were observed in 2002; and 5 adults, 37 subadults, 554 larvae and 150 egg masses were observed during visits in 1997, 1998, and from 2004-2007 (CNDDDB 2016). At Jose Creek and Million Dollar Road crossing (Fellers site ID S-464a and S-464b), 54 adults, 585 subadults and 441 egg masses observed during visits from 1994-1996, 1998, 2000, and 2003-2007 (CNDDDB 2016).

¹²⁰ The U.C. Museum of Vertebrate Zoology has 9 foothill yellow-legged frog specimens collected from the Willow Creek tributary at O'Neals from 1951-1953 (UCMVZ 2015). Moyle documented foothill yellow-legged frogs in the South Fork Willow Creek and at Little Finegold Lake during surveys from July 27 to September 4, 1970 (Moyle 1972, 1973; CNDDDB 2016). The foothill yellow-legged frog still occurred in Willow Creek below Bass Lake in the 1970s (PG&E 2000).

and reduced flows below Bass Lake were suspected to be factors in the frog's disappearance.

Fresno River

There are historical records from 1938-1970 in the Fresno River tributaries Carter Creek, Coarsegold Creek and Miami Creek (Madera County 2007; UCMVZ 2015).¹²¹

The species could not be founding Miami Creek or Carter Creek during focused surveys in 1989 and 2003 (Madera County 2007). There were reports of small numbers of *R. boylei* in China Creek in 1991 and Bass Lake in 1994; subsequent surveys and visits were unable to locate the species (PG&E 2000; CNDDDB 2016).¹²²

Chowchilla River

Moyle documented foothill yellow-legged frogs in the Chowchilla River during surveys from July 27 to September 4, 1970 (Moyle 1972, 1973; CNDDDB 2016). There are no known recent records from the Chowchilla River.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 3 of 6 historical locations (50%) in Madera County. There are no records of *R. boylei* from Madera County since 1994. The species may be extirpated from Madera County.

Mariposa County

Merced River

There are numerous historical collection records from 1899-1980 throughout the Merced River drainage, within and to the west of Yosemite National Park, including Bear Creek, Big Creek, Blacks Creek, Corbet Creek, Feliciano Creek, Mariposa Creek, Merced River, Piney Creek, Pleasant Valley, Rancheria Creek, Sherlock Creek, Smith Creek, South Fork Merced River and Sweetwater Creek (Grinnell and Storer 1924; Martin 1940; Richards 1958; Moyle 1972, 1973; Drost and Fellers 1994; FMNH 2001; HMCZ 2001; UMMZ 2001; USNM 2001; CNDDDB 2016; UCMVZ 2015).¹²³

¹²¹ The U.C. Museum of Vertebrate Zoology has 1 foothill yellow-legged frog specimen collected from Coarsegold Creek at Coarsegold in September 1938 (UCMVZ 2015). The foothill yellow-legged frog was observed in Miami and Carter creeks, tributaries of the upper Fresno River NW of Oakhurst, in the late 1960s and early 1970 (Madera County 2007).

¹²² The CNDDDB has a record of 5 adult *Rana boylei* observed on August 25, 1994 on the N shore of Bass Lake, 0.25 miles NW of "The Pines" Village. These frogs were noted to be threatened by predation from non-native bullfrogs (CNDDDB 2016). There is also a CNDDDB record of a single adult frog observed in May 1991 in China Creek, 3 miles SW of Bass Lake; subsequent visits revealed only bullfrogs (CNDDDB 2016). Pacific Gas and Electric surveys (PG&E 2000) were unable to locate the foothill yellow-legged frog in Bass Lake, noting that the species was last recorded there in 1994. PG&E (2000) noted that a high level of recreation in the lake was suspected to be a factor in the frog's disappearance.

¹²³ The U.S. National Museum of Natural History has 7 *Rana boylei* specimens collected from Mariposa (presumably along Mariposa Creek) in October 1899 (USNM 2001). Grinnell and Storer (1924) found foothill yellow-legged frogs from 1911 to 1920 along Piney Creek and Pleasant Valley (now flooded by Lake McClure), E to Sweetwater Creek (near Feliciano Mountain) and to Smith Creek, E of Coulterville (Grinnell and Storer 1924; Drost and Fellers 1994). Grinnell and Storer (1924) observed tadpoles in Blacks Creek near Coulterville in May 1919, collected 6 adult frogs plus an adult female of breeding age (UCMVZ #5687-

Extensive resurveys of the Grinnell and Storer (1924) Yosemite transect (Drost and Fellers 1994, 1996), as well as searches of suitable *R. boylei* habitat in the Yosemite area (Yosemite Valley, Foresta, Wawona, and El Portal) (Fellers and Freel 1995; Fellers 1997; Moritz 2007) were unable to locate any foothill yellow-legged frogs. Moritz (2007) surveyed 32 localities in the Hetch-Hetchy Area (between Ranger Station and O'Shaughnessy Dam along Hetch Hetchy Road), 16 localities in the Foresta Area, (between Hodgdon Meadows Campground and Foresta Road along Big Oak Flat Road), 13 localities in Yosemite Valley (between Arch Rock entrance station and Happy Isles Nature Center), 11 localities in Lyell Canyon (between Tuolumne Meadows and Mount Lyell), 25 localities in Glen Aulin (between McGee Lake and California Falls, including Tenaya Lake, Olmsted Point, and Siesta Lake), 15 localities along Bridalveil Creek (between Glacier Point and Chinquapin), and 16 localities in the Wawona Area (between Mariposa Grove and Rail Creek along Wawona Road), without detecting the species.

Fellers located small populations in the 1990s in the Bull Creek tributary of the North Fork Merced River, with a single larva seen in Bull Creek in 2007 (CNDDDB 2016).¹²⁴ Small populations were documented from 1998-2009 in Sherlock Creek (CNDDDB 2016), but during a NPS and BLM survey of Sherlock Creek in spring of 2015, the creek was over-run with bullfrogs, and only 1 *R. boylei* was found by a crew of 6 searchers looking from the confluence with the Merced River for at least a few miles upstream (S.

5692) from Smith Creek E of Coulterville in June 1915, and an adult female near Feliciana Mountain in November 1915. The Harvard Museum of Comparative Zoology has a *Rana boylei* specimen collected from 6 miles E of the W entrance to Yosemite National Park in August 1922 (HMCZ 2001). A 1924 Lyell Canyon expedition found the species in tributaries to the Merced River below El Portal (specimens are in CAS holdings) (Drost and Fellers 1994). The Field Museum of Natural History has a specimen collected from Yosemite, between Glacier Point and the Ranger Station, in May 1936 (FMNH 2001). The University of Michigan Museum of Zoology has foothill yellow-legged frog specimens collected from throughout the Merced River drainage in the 1930s to 1950s: a single frog from Briceburg in July 1938; 3 adults and 12 tadpoles taken from 14 miles S of Mariposa in August 1940; 1 tadpole from the bridge below Coulterville in August 1940; 1 frog from the Merced River 0.5 mile below Cascade Creek, in Yosemite National Park, in July 1948; and 4 frogs from Bower Cave, 6.5 miles S of Buck Meadows, in August 1950 (UMMZ 2001). Foothill yellow-legged frogs were documented at Wawona on the South Fork Merced River (Martin 1940) and at Fern Springs in Yosemite Valley (Richards 1958). The U.C. Museum of Vertebrate Zoology historical collection specimens include: 2 frogs from Pleasant Valley on May 17 and May 24, 1915; 1 frog from 3 miles NE of Coulterville on June 3, 1915; 6 frogs from Smith Creek, 6 miles E of Coulterville on June 5, 1915; 1 frog from 1 mile W of Coulterville on May 11, 1919; 10 frogs from Rancheria Creek, 2.8 miles SW of Briceburg on May 29, 1952; 2 frogs from 4.4 miles NE of Briceburg on July 11, 1952; 1 frog from 0.7 miles NE of Briceburg on November 16, 1952; 2 frogs from Big Creek at Fish Camp on April 19, 1953; 6 frogs from Merced River, 1.9 miles E of Indian Lodge, on April 19, 1953; 2 frogs from Feliciana Creek, 2 miles NE of Briceburg, on April 19, 1953; 1 frog from Rancheria Creek, 2.8 miles SW of Briceburg, on May 29, 1953; 1 frog from Bower Cave on March 8, 1953; 1 frog from 1.8 miles S of Briceburg on February 26, 1954; 1 frog from Bear Creek, 2 miles S of Briceburg, on April 7, 1954; 1 frog from Briceburg on April 26, 1955; 1 frog from the Bear Creek drainage, 0.8 miles SE of Briceburg, on March 1, 1957; 2 frogs from Bower Cave on May 30, 1959; and 2 frogs collected from 2.5 miles N of Midpines on an unknown date (UCMVZ 2015). Moyle documented foothill yellow-legged frogs in Corbet Creek (SE of Lake McClure), Sweetwater Creek, and Bear Creek (near Briceburg) during surveys from July 27 to September 4, 1970 (Moyle 1972, 1973; CNDDDB 2016). The U.C. Museum of Vertebrate Zoology has 2 tadpole specimens (MVZ #175103) collected July 8, 1980 from Sherlock Creek, E of Bear Valley (UCMVZ 2015).

¹²⁴ In Bull Creek at Road 02S02 (1 mile N of Little Grizzly Mountain, 4 miles N of Merced River Mile 101), Fellers observed 1 tadpole and 1 adult in June 1993. Along Bull Creek about 0.7 miles N of Kinsley Guard Station and about 3.6 miles NW of Jenkins Hill (Fellers site ID #Y-347D), Fellers observed 1 subadult frog on 10/12/93 and 1 subadult frog on 9/30/94. Along Bull Creek at the confluence with an unnamed tributary, about 0.5 miles SW of Anderson Flat and about 1.5 miles N of Little Grizzly Mountain (Fellers site ID #Y-347A), Fellers observed 1 subadult on 9/30/94, 1 larva on 6/17/96, 1 larva on 6/8/97, 1 adult and 1 subadult on 6/29/98, and 1 larva on 8/6/07.

Kupferberg, pers. comm., 2016). There was a report in 2008 of “many” foothill yellow-legged frogs along the Merced River downstream from McCabe Flat (CNDDB 2016).¹²⁵

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 1 of 5 historical locations (20%) in Mariposa County. Extensive surveys during the 1990s and 2000s (Drost and Fellers 1994, 1996; Fellers and Freel 1995; Jennings 1996; Fellers 1997; Moritz 2007) failed to locate any foothill yellow-legged frogs. The remaining populations appear to be small ones in the Merced River (and tributaries Bull Creek and Sherlock Creek) above Lake McClure. The species is nearly extirpated from Mariposa County.

Central/Northern Sierra Nevada

The Northern/Central Sierra includes drainages of the Sierra foothills in Plumas, Butte, Yuba, Sierra, Nevada, Placer, El Dorado, Amador, Calaveras and Tuolumne counties. Sutter County is discussed here: a single historical record of *R. boylei* from Sutter Buttes (a small isolated range of eroded volcanic lava domes in the Central Valley), in northwestern Sutter County east of the Sacramento River, was likely a small disjunct population which is now extirpated.

There has been confusion over identification of some yellow-legged frog populations in the northern Sierra Nevada. While the foothill yellow-legged frog (*R. boylei*) inhabits low to moderate elevation streams from sea level to 1,830 m (6,000 feet) in California (Stebbins 1985; Stebbins and McGinnis 2012), the Sierra Nevada yellow-legged frog (*Rana sierrae*) occurs at elevations mostly above 1,820 meters (6,000 feet), with the lowest documented occurrence of this species in the Sierra foothills at 1,044 m (3,425 feet) (Jennings and Hayes 1994; USFS 2000). Historically, the ranges of *R. sierrae* and *R. boylei* abutted each other at mid-elevations in the western and northern Sierra Nevada, and the two species were sometimes found in close proximity to each other in the same drainages (Zweifel 1955). Though the two species can appear morphologically similar, genetic analysis can differentiate between the species (e.g. Lind et al. 2011; Poorten et al. 2013).

Hayes et al. (2013) reviewed the status of *R. boylei* on national forests in the northern and central Sierra Nevada (Lassen, Plumas, Tahoe, El Dorado and Stanislaus National Forests), and documented declines and apparent loss of many historic populations in the northern and central Sierra National Forests. Although *R. boylei* populations persist in many river basins, including the American, Clavey, Cosumnes, Feather, Merced, Mokelumne, Stanislaus, Tuolumne, and Yuba Rivers, the majority of the recent observations in these national forests are of small and scattered populations, with limited evidence of successful reproduction.

There have been severe declines in the central Sierra foothills (Moyle 1973; Drost and Fellers 1996) and populations in the northern Sierra may be in decline as well (Lannoo 2005). As discussed below by county, at least half of known historical locations have

¹²⁵ In Sherlock Creek from the confluence with the Merced River to about 1 mile upstream of Lyons Gulch (SW of Telegraph Hill): 1 adult on 3/22/98; 7 frogs on 9/20/05; dead and live frogs and tadpoles in lower Sherlock Creek and at mouth of Merced River on 8/28/08; and 3 adults and 7 juveniles on 8/28/09 (CNDDB 2016). The CNDDB has a recent report of “many” *Rana boylei* observed in September 2008 along the Merced River downstream from McCabe Flat, about 1.5 miles NE of Telegraph Hill and 1.8 miles SE of Quartz Mountain (CNDDB 2016).

been lost in every northern and central Sierra county except Plumas County, and most of the recently documented populations in the northern and central Sierra are small and scattered. Significant populations remain in El Dorado County (Rubicon River), Placer County (North Fork American River and North Fork of the Middle Fork American River), Nevada County (Middle Yuba River, South Yuba River and Bear River), and Plumas County (North Fork Feather River, Middle Fork Feather River, Oroleve Creek, South Fork Feather River, Spanish Creek, and Canyon Creek tributaries Slate Creek and Onion Creek).

Central and Northern Sierra National Forests

Hayes et al. (2013) comprehensively reviewed the foothill yellow-legged frog's historical (prior to 1980) and recent (1980 to 2001) status across the five National Forests (Stanislaus, El Dorado, Tahoe, Plumas, and Lassen) in the northern and central Sierra Nevada, which overlap extensively with the range of the northern Sierra Nevada *R. boylei* population. Hayes et al. (2013) evaluated records from museum databases and more recent visual survey efforts. A summary of the findings of Hayes et al. (2013), by National Forest:

Stanislaus National Forest

Hayes et al. (2013) noted that a number of historical foothill yellow-legged frog records exist for the Stanislaus National Forest from the Middle and South Forks of the Tuolumne River, and the North Fork of the Merced River. In the Tuolumne River basin, collections were made from 1948 to 1974 in the South Fork Tuolumne River and Middle Fork Tuolumne River. In the North Fork Merced River basin, collections were made from Smith Creek in 1915 and the North Fork Merced River in the 1950s, and one record exists from Sherlock Creek in the 1980s. Numerous historical collection records from 1899 to 1953 exist for five major hydrographic basins that extend onto Stanislaus National Forest lands, downstream of the current National Forest boundary: the Mokelumne River (Licking Creek); Calaveras River (Big Trees Creek); Stanislaus River (Angels Creek and Moran Creek), Tuolumne River (Mocassin Creek, Turnback Creek and Woods Creek); and the Merced River (Blacks Creek, Cuneo Creek, Maxell Creek, Merced River and Piney Creek).

Hayes et al. (2013) noted that scattered sightings of foothill yellow-legged frogs exist for the Stanislaus National Forest and vicinity since 1980, but the earliest systematic surveys date from 1993. Recent records exist for the Calaveras, Clavey, Merced, Stanislaus, and Tuolumne River basins. Two historical sites have recent foothill yellow-legged frog sightings, and surveys since 1993 have identified 19 previously unidentified sites. In the Calaveras River drainage, a single frog was reported sighted from San Antonio Creek (Lind et al. 2003) but resurvey efforts 2003-2005 failed to locate the species: some possibility exists that foothill yellow-legged frogs have been extirpated from the Calaveras River. In the Clavey River drainage, foothill yellow-legged frogs were recorded reproducing from 1995-2002 and 2009-2010 in Bull Meadow Creek, Clavey River mainstem, and a Hull Creek tributary, but with low numbers of post-metamorphs. Kupferberg et al. (2013) documented breeding at 4 or 5 sites on the Clavey River. In the Merced River drainage, small numbers of frogs were recorded from 1995-2003 in Bull Creek, a tributary of the North Fork of the Merced River, with low numbers of post-metamorphs. In the Stanislaus River drainage, foothill yellow-legged frogs were found at 12 localities between 1993 and 2002, with evidence of reproduction in Rose Creek and

the mainstem Middle Fork Stanislaus River, and sightings in Coyote Creek and Skull Creek; overall, numbers of adults and juveniles were low, and indication of recruitment success was limited to 3 sites. In the Tuolumne River drainage, small numbers of foothill yellow-legged frogs were reported from 4 localities between 1993 and 2012, with evidence of reproduction in the North Fork Tuolumne River mainstem and Hunter Creek, and sightings in the mainstem Tuolumne River; indication of recruitment success was limited to 1 site (Hayes et al. 2013).

El Dorado National Forest

Hayes et al. (2013) noted historical records for the foothill yellow-legged frog on the Eldorado National Forest from 1916 and 1935 along the South Fork of the American River. Additional historical records exist downstream of the current National Forest boundary in two major hydrographic basins, the American (including Middle and South Forks) and Cosumnes Rivers. Collection records on the American River from 1850 to 1961 include from Dry Creek, North Fork of the American River, South Fork of the American River, Weber Creek and a tributary of Slate Creek; from the Cosumnes River basin include Squaw Hollow Creek and Martinez Creek in 1942. Hayes et al. (2013) noted no foothill yellow frog records from the 1970s from either the Eldorado National Forest or lands lower in elevation than Eldorado National Forest lands.

Systematic frog surveys in the Eldorado National Forest and region were not conducted until the 1990s. Foothill yellow-legged frogs were detected in the early 1990s (Martin 1992) in the Eldorado National Forest in South Fork American River tributaries Camp and Snow Creeks, as well as Bark Shanty Creek (this is at elevation 6,270 feet, so possibly could represent *R. sierrae*). A single frog was seen on Sopiago Creek, a third-order tributary of the Cosumnes River, but subsequent significant survey efforts found no other foothill yellow-legged frogs on this creek. Recent amphibian surveys for FERC relicensing have documented foothill yellow-legged frog populations in four major hydrographic basins: the Middle Fork of the American River and its tributaries, including the Rubicon River and Otter Creek; the South Fork American River and its tributaries, including Silver Creek and Soldier Creek; the Cosumnes River system, especially Camp Creek and its tributaries; and the North Fork Mokelumne and its tributaries, especially Camp, Green, and East Panther Creeks.

Tahoe National Forest

Hayes et al. (2013) noted scattered historical records from 1899-1969 of foothill yellow legged frogs for the Tahoe National Forest within the Yuba River drainage, and from drainages below National Forest lands including Deer Creek, Dry Creek, Middle Fork American River, Middle Fork Yuba River, Middle Yuba River, New York Creek, North Fork American River, North Fork Yuba River, South Honcut Creek, South Poorman Creek, South Yuba River, Washington Creek, Willow Creek and Yuba River.

Systematic surveys on the Tahoe National Forest were not initiated until the late 1990s (Koo and Vindum 1999). Very small numbers of foothill yellow-legged frogs were recorded at a number of localities on the North, Middle, and South Yuba Rivers, and the North and Middle Forks of the American River, including several localities for which historical records were lacking. Very small numbers of *R. boylei* were recorded at 10 localities in the North Yuba River system (Devil's Creek, Fiddle Creek, Humbug Creek, Kanaka Creek, North Yuba River and Woodruff Creek); a single frog was recorded from

the Middle Yuba River system (Grizzly Creek); 2 frogs were recorded in the South Yuba River; 1 frog was recorded in the North Fork American River system (North Shirttail Creek); and 2 localities were recorded on the Middle Fork American River system (single frogs in Skunk Canyon Creek and in North Fork of the Middle Fork of the American River). Hayes et al. (2013) had not incorporated *R. boylei* data from recent FERC surveys during in the South and Middle Forks of the Yuba River, the Bear River, the North Fork Middle Fork and the Middle Fork of the American River, and the Rubicon River (including tributary streams within these watersheds). The Nevada Irrigation District and PG&E conducted *R. boylei* surveys from 2008-2010 on the Middle Yuba and South Yuba – these data were not incorporated by Hayes et al. (2013).

Plumas National Forest

Hayes et al. (2013) noted many historical collections of foothill yellow-legged frog from 1924-1978 in the Plumas National Forest and other lands either in in-holdings or the westslope Sierra below the Plumas National Forest boundary. Historical records were primarily in the Feather River, Butte Creek and Chico Creek drainages, including Anderson Fork, Big Chico Creek, Butte Creek, Cherokee Creek, Dooley Creek, Feather River, Last Chance Creek, Little Butte Creek, Little North Fork of the Feather River, Middle Fork Feather River, Mud Creek, North Fork Feather River, Rock Creek, South Fork Feather River, Spanish Creek, Sulphur Creek and West Branch Feather River. By the end of the 1970s, foothill yellow-legged frogs had been recorded from 12 different localities on the Plumas National Forest, and 40 additional localities at elevations below Plumas National Forest lands. Foothill yellow-legged frogs seem to have disappeared from at least one valley floor at that time, but available data imply that species was still present over its Sierra range in this region.

Systematic surveys were initiated on the Plumas National Forest in the late 1990s (Koo and Vindum 1999); foothill yellow-legged frogs were recorded at only 45% of the sites where they were historically found. The species was recorded at 6 of 6 historical sites in the Canyon Creek drainage (including Onion and Slate Creeks); at 2 of 2 historical sites in the South Fork Feather River drainage (including Oroleve Creek); at only 2 of 4 sites in the Middle Fork of the Feather River; only 1 of 7 sites in the East Branch of the North Fork Feather River (Spanish Creek); and not located at 2 historical sites in Little Butte Creek; 1 historical site in Dry Creek; 1 historical site in the North Fork Yuba River; or 1 historical site in the West Branch Feather River. The Plumas National Forest has continued to conduct extensive amphibian surveys since the late 1990s, finding that all drainages in which foothill yellow-legged frogs were detected during 1990s CAS surveys still appear to have foothill yellow-legged frogs present, and detecting several locations in addition to the 24 historical sites. Extensive surveys and monitoring since 2001 of frog populations in reaches of the North Fork Feather River have documented an increasing *R. boylei* population on the Poe reach and a significant decline of the population on the Cresta reach. PG&E data and FERC data on *R. boylei* for hydropower relicensing activities had not yet been incorporated by Hayes et al. (2013).

Lassen National Forest

Hayes et al. (2013) noted only 3 historical records from 1938-1978 in Lassen National Forest, in tributaries of the North Fork Feather River and West Branch Feather River, and in Little Butte Creek. Resurveys from 1973-1978 revealed the species had been extirpated from the former site on Little Butte Creek. Several additional pre-1980 records

(from 1911-1970) exist downstream of the current National Forest boundary, in primary tributaries of the Sacramento River including Battle, Big Chico, Butte, Cow, Dye, Mill, Paynes, Rock and Stillwater Creeks, as well as the mainstem Sacramento River (Hayes et al. 2013). Resurveys from 1973-1976 in former collection localities on the Sacramento River could not find foothill yellow-legged frogs, making it possible that the species was extirpated on the mainstem Sacramento River by the mid-1970s.

Jennings documented dozens of foothill yellow-legged frogs in 1996 in Dye Creek and its tributaries. Hayes et al. (2013) noted scattered collection records and sighting since 1980 in Lassen National Forest under surveys through the Federal Energy Regulatory Commission (FERC) relicensing process. Small numbers of frogs were found in Antelope Creek, Indian Creek and Mill Creek from 2001-2003. Several records documented since 1980 also exist for drainages that extend onto Lassen National Forest, but the records of which are from elevations below National Forest lands.

Tuolumne County

Tuolumne River

There are historical collection records from 1932-1974 from the South Fork Tuolumne River, Middlefork Tuolumne River and tributaries of Don Pedro Reservoir, including Eleanor Creek, Hatch Creek, Turnback Creek and Woods Creek (Martin 1940; Richards 1958; Moyle 1972, 1973; USNM 2001; UCMVZ 2015; CNDDDB 2016).¹²⁶

Small populations were documented in the 1990s and early 2000s in several tributaries above Don Pedro Reservoir, including Big Jackass Creek, Hunter Creek and Moccasin Creek (CNDDDB 2016).¹²⁷ Foothill yellow-legged frog habitat assessments were conducted in 2012 for FERC in the Tuolumne River upstream of areas regularly inundated by Don Pedro Reservoir, and the following tributaries: Big Creek, Blue Gulch, Deer Creek, Drainage #8, Fleming Creek, Hatch Creek, Kanaka Creek, Moccasin Creek, Poor Man's Gulch, Rogers Creek, Rough and Ready Creek, Six-Bit Gulch, Slate Creek, Smarts Gulch, Sullivan Creek, Tuolumne River, West Fork Big Creek, Willow Creek and Woods Creek (HDR 2013). Foothill yellow-legged frog surveys were performed at five of these streams from 6/18/12-6/21/12, focused on detecting frog larvae, adults and juveniles. No *R. boylei* were detected during surveys at any of the sites (HDR 2013).

¹²⁶ Six frogs from 2 miles NW of Jacksonville in August 1932 (USNM 2001); and 2 frogs from Berkeley-Tuolumne Camp on the South Fork Tuolumne River on June 12, 1948; 1 frog from Sawmill Flat on Woods Creek on May 14, 1949; 1 frog from Woods Creek on April 23, 1950; 1 frog from Turnback Creek on April 15, 1951; 1 frog from 4 mi W of Hardin Flat on the South Fork Tuolumne River on August 31, 1962; 10 frogs (UCMVZ# 136239-136248) from Monroe Middlefork Camp, Middlefork of the Tuolumne River on July 8 and 15, 1972; and 1 frog from Monroe Middlefork Camp, Middle Fork Tuolumne River on August 2, 1974 (UCMVZ 2015; CNDDDB 2016). The species was reportedly once found at the base of Lake Eleanor Dam (elevation 4,657 feet) along Eleanor Creek, a tributary to Cherry Creek (Martin 1940; Richards 1958). Moyle documented foothill yellow-legged frogs in Hatch Lake (Hatch Creek tributary above Don Pedro) and Second Lake during surveys from July 27 to September 4, 1970 (Moyle 1972, 1973; CNDDDB 2016).

¹²⁷ Three eggs and 2 tadpoles observed in Hunter Creek at Forest Service Road 01N03 crossing, 2 miles S of Murphy Ranch, in 1993; 4 adults seen near Bull Meadow at the Forest Service Road 01N69 crossing, about 0.9 miles ESE of Clavey River mile 9, in 1993; several large tadpoles undergoing metamorphosis observed S of Table Mountain about 1 mile S of Yosemite Junction on May 15, 1997; and 2 adults and 4 egg masses observed near the confluence of Big Jackass Creek and Moccasin Creek, S of Highway 49, 4 miles E of Don Pedro Reservoir, on May 4, 2001 (CNDDDB 2016).

Stanislaus River

David Martin documented 1 adult *R. boylei* on a tributary of the South Fork Stanislaus River, between river mile 24 and 25 (about 3 miles W of Cold Springs) in 1993 (CNDDDB 2016).

Small populations were documented consistently from 1993-2008 in Rose Creek, a tributary of the South Fork Stanislaus River (CNDDDB 2016).¹²⁸

There is a *R. boylei* population in the Sand Bar Dam reach of the Stanislaus River (S. Kupferberg, pers. comm., 2016).

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 2 of 6 historical locations (33%) in Tuolumne County. Other than small populations documented in the Rose Creek tributary of the South Fork Stanislaus River through 2008, and a few small populations in the Tuolumne River above Don Pedro Reservoir through 2001, there are no other known recent observations in Tuolumne County. See the discussion above regarding recent Stanislaus National Forest surveys.

Sutter County

There is a single historical record of *R. boylei* from Sutter Buttes (a small isolated range of eroded volcanic lava domes in the Central Valley), in northwestern Sutter County, east of the Sacramento River. This was likely a small disjunct population.

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at the single historical location in Sutter County. There are no CNDDDB reports of *R. boylei* from Sutter County and the species is presumed to be extirpated from Sutter County.

Calaveras County

Stanislaus River

There are historical collection records from 1953 from the Stanislaus River drainage, including tributaries Angel's Creek and Moran Creek (UCMVZ 2015).¹²⁹

There were small populations reported in the Coyote Creek tributary from 1993-2005 (CNDDDB 2016).¹³⁰

¹²⁸ At Italian Bar Road Crossing, about 2 miles NE of American Camp Station (Feller ID# Y-441C): 1 adult on 7/1/93; 1 subadult on 5/10/94; 1 subadult and 1 larva on 6/25/96; 1 adult and 105 larvae on 7/3/97; 1 adult, 221 larvae and 200 egg masses on 6/30/98; 1 adult on 6/17/99; 200 larvae and 200 egg masses on 6/1/00; 2 larvae on 8/30/02; and 1 adult on 6/2/03 (CNDDDB 2016). In Rose Creek at Forest Road 03N03 crossing (Fellers site ID# Y-441A&B): 1 adult on 7/1/93; 43 adults, 277 subadults, 117 larvae and 600 egg masses cumulatively during visits in 1994, 1996 and 2002-2008 (CNDDDB 2016). In Rose Creek at Eagle Creek (Fellers site ID# Y-441C): 1 adult and 33 subadults on 9/20/05; 1 larva on 9/1/06; and 6 adults and 204 larvae on 7/24/07 (CNDDDB 2016).

¹²⁹ Five frogs from Angel's Creek, 1.2 miles WSW of Murphys, on March 23, 1953; 2 frogs from Moran Creek, 1.5 miles E and 3 miles N of Avery, on May 10, 1953; and 2 frogs from near Hwy. 4 at the S boundary to Calaveras Big Trees State Park on March 10 and 23, 1953 (UCMVZ 2015).

Mokelumne River

There is one historical collection record of 3 frogs from Licking Creek, a tributary of the South Fork Mokelumne River, in October 1899 (USNM 2001).

Small populations were documented in Jesus Maria Creek (a Mokelumne River tributary above Pardee Reservoir) in 2002, and along the North Fork Mokelumne River during 2001 and 2009 FERC amphibian surveys (CNDDDB 2016).¹³¹

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 2 of 9 historical locations (22%) in Calaveras County. Small populations were documented in the North Fork Mokelumne River drainage as of 2009, the Jesus Maria Creek tributary to the Mokelumne River in 2002, and the Coyote Creek tributary to the Stanislaus River as of 2005. See the discussion above regarding recent Stanislaus National Forest surveys.

Amador County

There are no known historical collection records of *R. boylei* from Amador County.

Mokelumne River

Small populations were documented during 2001 and 2009 FERC amphibian surveys in Amador County in the North Fork Mokelumne River and tributary Tiger Creek (CNDDDB 2016).¹³² Garcia and Associates may have more recent survey data for this reach of the Mokelumne River.

¹³⁰ A single frog was collected from the Coyote Creek tributary to New Melones Reservoir, from Natural Bridge SW to past Krappeau Gulch, in September 1993 (CAS 2001). Fellers observed *Rana boylei* along Coyote Creek at Natural Bridge, about 0.7 mile W of Sugarloaf Mountain, and about 2.1 miles S of Vallecito (Fellers site ID# Y-425A & Y-425B): a total of 8 adults, 49 subadults, 244 larvae and 3 egg masses observed at Y425A during annual surveys from 1993-2001, 2003 and 2005; and 1 adult and 5 subadults observed on 10/19/93 at Y-425B (CNDDDB 2016).

¹³¹ Fellers observed 1 adult and 6 subadults in Jesus Maria Creek (a Mokelumne River tributary above Pardee Reservoir), about 0.64 mile W of the Spring Gulch Confluence (Fellers site ID# T-585), on August 12, 2002 (CNDDDB 2016). In the North Fork Mokelumne River, 0.9 mile W of Devils Nose (PG&E FYLF Site 16 and 30): 7 adults, 28 juveniles and 13 tadpoles from 5/15/01-9/7/01; 1 egg mass on 7/6/09; 1 adult, 17 tadpoles and 10 juveniles on 9/8/09; and 2 adults and 14 juveniles on 10/6/09 (CNDDDB 2016). North Fork Mokelumne River, from near river mile 114 to about 1.5 river miles upstream (about 5.5 miles S of Hams Station): unspecified number of tadpoles from 5/15/01-9/7/01; 2 adults on 7/8/09; 1 adult and 36 tadpoles on 9/10/09; and 2 adults and 25 juveniles on 10/8/09 (CNDDDB 2016). North Fork Mokelumne River, about 0.4 mile upstream from the mouth of Tiger Creek, 4.6 miles ENE of Pioneer: 80 tadpoles and 5 adults on 7/9/09; 1 tadpole, 5 juveniles and 4 adults on 9/9/09; and 7 juveniles and 2 adults on 10/7/09 (CNDDDB 2016). Bear River, about 20 m upstream from North Fork Mokelumne River, about 6.4 miles SE of Hams Station: 1 adult on 8/18/09 (CNDDDB 2016).

¹³² Along North Fork Mokelumne River, about 0.4 mile upstream from the mouth of Tiger Creek: 80 tadpoles and 5 adults on 7/9/09; 1 tadpole, 5 juveniles and 4 adults on 9/9/09; and 7 juveniles and 2 adults on 10/7/09. Along Tiger Creek, 1 adult seen about 0.4 mile NE of the confluence with North Fork Mokelumne River on 8/19/09; and 1 adult seen about 1.3 miles ENE of the mouth of Tiger Creek on 8/19/09 (CNDDDB 2016).

Dry Creek

Small populations were observed during surveys in the early 2000s in Else Creek, an upper tributary of Dry Creek (CNDDDB 2016).¹³³

Recent status: Jennings and Hayes (1994) were unable to locate the species during resurvey efforts from 1988-1991 at any of 3 historical locations in Amador County. Small populations were documented in the North Fork Mokelumne River and tributary Tiger Creek as of 2009, and in an upper tributary of Dry Creek in the early 2000s.

El Dorado County

South Fork American River

There are historical collection records from 1916-1953 in El Dorado County in the South Fork American River drainage, including the Indian Creek and Webber Creek tributaries (Slevin 1928; UCMVZ 2015).¹³⁴

Small populations were documented in 1992 and 1993 in the South Fork American River tributaries Bark Shanty Canyon Creek and Snow Creek, with a significant population in the Indian Creek tributary in 2003 (CNDDDB 2016).¹³⁵

Small populations were documented in 2002 and 2004 at various locations along the South Fork American River (CNDDDB 2016).¹³⁶ DTA and Stillwater Sciences (2005) conducted foothill yellow-legged frog surveys from 2002-2004 in moderate to high quality habitat within the 8-mile Slab Creek Dam Reach (from the base of Slab Creek Reservoir Dam downstream to White Rock Powerhouse). The surveys, conducted at 5 South Fork American River sites (downstream of Iowa Canyon Creek, at the confluence with Rock Creek, one mile downstream of Rock Creek, upstream of White Rock Powerhouse, and

¹³³ At Indian Grinding Rock State Historic Park (1.1 miles NE of Pine Grove), 5 frogs in July 2000; 13 frogs in July 2001; and 1 adult on July 1, 2004 (CNDDDB 2016).

¹³⁴ The U.C. Museum of Vertebrate Zoology has historical collection specimens from the South Fork American River drainage in El Dorado County: 3 frogs from Fyffe in July 1916; 4 frogs from the S Fork American River, 2.5 miles W of Kyburz, in May 1935; 9 frogs from 7 miles W and 1 mile S of Placerville (likely the Indian Creek tributary) in November 1938; 4 frogs from Webber Creek, 2.2 miles WSW of Placerville, in June 1952; and 1 frog from 4 miles NW of Coloma, in May 1953 (UCMVZ 2015).

¹³⁵ Three adults and 150 tadpoles in Bark Shanty Canyon Creek (2.5 km NNW of Fitch Rantz Bridge and 3.3 km SE of Owens Camp Field Station) on 7/6/92; and 1 adult in Snow Creek at Big Pebble Canyon Road crossing (about 1.2 miles S of Iron Mountain Road, 1.5 miles E of Matulich Spring) in 1993 (CNDDDB 2016). A significant population (100 adults and juveniles, no distinction made) was seen in the Indian Creek tributary, 2 miles N of Lotus, on 10/27/03 (CNDDDB 2016).

¹³⁶ Two frogs just downstream of Blackbird Campground on 6/26/02; 12 tadpoles and 4 adults about 1.8 air miles ESE of Silver Creek Crossing (about 1.9 miles NE of Pollock Pines) on 8/7/02; 3 adults, 4 juveniles and 5 of unknown age about 0.9 mile ESE of Silver Creek Crossing (about 1.6 miles N of Pollock Pines), on 8/8/02; 2 adults, 1 juvenile and 6 tadpoles at gaging station about 0.16 mile NE of Silver Creek Crossing (about 2.1 miles N of Pollock Pines), on 8/8/02; 4 juveniles about 1.1 miles E of El Dorado Power House (about 2.3 miles NNW of Pollock Pines), on 8/8/02; and 1 adult and 2 tadpoles about 1.7 miles NW of USFS Pacific Ranger Station (about 2.3 miles ENE of Pollock Pines), on 8/30/02 (CNDDDB 2016). At El Dorado Power House (about 4.9 miles NE of Camino): 6 frogs on 6/27/02; 8 adults and 4 juveniles on 10/28/02; and 2 adults, 1 juvenile and 1 of unknown age on 10/30/02 (CNDDDB 2016). In the vicinity of Maple Grove Campground (about 2.2 to 3.5 miles ENE of USFS Pacific Ranger Station): 1 adult on 5/29/02; 1 adult and 1 of unknown age on 8/14/02; 7 juveniles on 10/30/02; and from June-October 2004, 1 juvenile, 2 metamorphs, 2 tadpoles and 3 egg masses at 3 sites (CNDDDB 2016).

at White Rock Powerhouse), and 2 tributary sites (the lower portion of Iowa Canyon Creek and the lower portion of Rock Creek), did not detect any *R. boylei*. In 2003, 1 adult foothill yellow-legged frog was incidentally observed on the South Fork American River approximately 0.5 miles upstream of White Rock Powerhouse.

Garcia and Associates (2010a) conducted foothill yellow-legged frog surveys for the El Dorado Irrigation District at 8 sites along the South Fork American River, on July 26 and 27, 2010. The survey included 3 sites upstream of Silver Creek, 1 site at the confluence with Soldier Creek, 1 site upstream of Ogilby Creek, and 3 sites near Maple Grove Campground. No *R. boylei* were detected at any of the sites. ECORP (2011) conducted foothill yellow-legged frog surveys in 2010 for the Sacramento Municipal Utility District along a 0.25 mile reach of the South Fork American River below Slab Creek Reservoir Dam and in 1,800 feet of lower Iowa Canyon Creek. No *R. boylei* were detected.

Middle Fork American River

2007 surveys by the Placer County Water Agency (PCWA) documented foothill yellow-legged frog breeding in the lower portions of the Rubicon River (below 3,350 feet) and Middle Fork American River (below 1,800 feet), four lower elevation tributaries (American Canyon Creek, Gas Canyon Creek, Todd Creek and Otter Creek), and comparison river reaches, Shirttail Creek and North Fork of the Middle Fork American River (PCWA 2008). *Rana boylei* were reported dispersed widely throughout the study area in varying densities depending on stream size, flow regulation, and water temperatures. Abundance was highest in the downstream reaches of the Rubicon River and in comparison reaches and tributaries. Abundance was low in the Middle Fork American River bypass reach upstream of Ralston Afterbay, and frogs were observed rarely in the Middle Fork American River peaking reach. No frogs were observed above approximately 1,800 ft in elevation on the Middle Fork American River, 3,350 ft elevation on the Rubicon River, and above 1,550 ft elevation on Long Canyon Creek (near the Long Canyon Creek confluence with the Rubicon River). Breeding was observed in the lower portions of the Rubicon River and Middle Fork American River bypass reaches, in four lower elevation tributaries (American Canyon Creek, Gas Canyon Creek, Todd Creek, and Otter Creek), and in the comparison river reaches. No egg masses were observed in the mainstem of the Middle Fork American River reach. Fall surveys generally reflected this distribution with the highest number of observed tadpoles and young-of-the-year in the Rubicon River, peaking reach tributaries, and at comparison sites.

The density of egg masses at breeding locations, as an index of population size, varied by river reach. The Rubicon River bypass reach had the highest density of egg masses (19 egg masses/km in the three lower sites) and the Middle Fork American River bypass reach had one of the lowest densities of egg masses (2 egg masses/km). The tributaries along the peaking reach (Todd Creek, Gas Canyon and Otter Creek) had moderate egg mass densities (average of 9 egg masses/km). Two of the unregulated comparison sites, Shirttail Creek and the upper site on the North Fork of the Middle Fork American River, had high egg mass densities similar to the lower Rubicon River of 17 egg masses/km and 14 egg masses/km, respectively. Two of the comparison survey sites, North Fork of the Middle Fork American River near the confluence with the Middle Fork American River and the mainstem of the North Fork American River near Shirttail Creek, had low egg mass densities (3 egg masses/km and 2 egg masses/km, respectively). (PCWA 2008)

Most of the PCWA survey results appear to have been submitted to the California Natural Diversity Database, with small numbers of foothill yellow-legged frogs reported from 2003 and 2007 surveys in the Middle Fork American River and the tributaries North Fork of the Middle Fork, Gas Canyon Creek and Otter Creek; one moderate and several small populations were reported in the Rubicon River and its tributaries Long Canyon Creek and Pilot Creek in 2007 (CNDDDB 2016).¹³⁷

Cosumnes River

There are historical collection records from 1942-1961 in the North Fork Cosumnes River drainage, including tributaries Martinez Creek and Squaw Hollow Creek (UMMZ 2001; UCMVZ 2015).¹³⁸

Small populations were documented from 1992 to 1999 in the North Fork Cosumnes River and tributaries Camp Creek and Middle Fork Cosumnes River (CNDDDB 2016).¹³⁹

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 1 of 9 historical locations (11%) in El Dorado County. However, small populations were documented from 1992-2007 in El Dorado County at multiple locations in the Cosumnes River, South Fork American River and

¹³⁷ In the Middle Fork American River at Paradise Canyon, near Auburn State Recreation Area: 1 subadult on 11/2/03; 1 adult on 5/17/07; 1 adult on 6/5/07; and 5 adults on 8/23/07. In the Todd Creek tributary: 6 adults, 4 juveniles and 1 egg mass on 5/17/07; and 1 adult on 6/5/07. Along Gas Canyon Creek at Middle Fork American River (about 3.9 miles SE of Applegate): adults captured and released and 9 additional observed on 11/2/03; 9 adults, 4 juveniles and 4 egg masses on 5/17/07; 2 adults and 4 juveniles on 6/5/07; 7 adults on 8/22/07; and 1 adult on 8/23/07. At Poverty Bar along Middle Fork American River (about 2.3 miles ESE of Stony Hill, 6.7 miles ENE of Auburn): 7 adults, 2 juveniles and 22 young-of-year on 9/24/07. At Fords Bar along Otter Creek (about 2.4 miles W of Little Bald Mountain, 3.4 miles NNW of Georgetown): 2 egg masses on 5/23/07; 2 adults, 570 tadpoles and 1 egg mass on 6/6/07; and 2 adults, 86 young-of-year and 9 tadpoles on 8/31/07. At the junction of the Middle Fork American River and North Fork of Middle Fork American River (about 3.9 miles E of Foresthill): 2 egg masses on 5/23/07; 2 adults, 570 tadpoles and 1 egg mass on 6/8/07; and 2 adults, 86 young-of-year and 9 tadpoles on 8/31/07. (CNDDDB 2016). Along the Rubicon River (about 0.9 mile SE of Middle Fork American River confluence): 11 adults and 18 egg masses on 5/22/07; 15 adults, 3 juveniles, 2,910 tadpoles and 7 egg masses on 6/7/07; and 18 adults, 18 juveniles, 101 young-of-year and 49 tadpoles on 8/21/07. In the Rubicon River and Long Canyon Creek tributary (about 0.33 air miles E of Squaw Flat): 16 adults, 6 juveniles and 18 egg masses on 5/25/07; 16 adults, 11 juveniles, 4,850 tadpoles and 1 egg mass on 6/12/07; and 47 adults, 22 juveniles, 233 young-of-year and 117 tadpoles on 8/24/07. In the Rubicon River and tributary Pilot Creek (about 6.5 miles NE of USFS Georgetown Ranger Station): 16 adults, 2 juveniles, 96 young-of-year and 4 tadpoles on 8/28/07. Along the Rubicon River about 1.5 miles NE of Stumpy Meadows Lake: 200 tadpoles and 24 egg masses on 5/30/07; and 3 adults, 1 juvenile and 4,500 tadpoles on 6/14/07; 3 adults, 67 young-of-year and 25 tadpoles on 8/27/07. Along the Rubicon River, about 0.55 mile SSW of Ellicott Bridge: 2 adults and 4 egg masses on 5/29/07; 400 tadpoles on 6/13/07; and 1 young-of-year on 8/28/07. (CNDDDB 2016)

¹³⁸ An unknown number of tadpoles from the Martinez Creek tributary, 4 miles S of El Dorado, in October 1942; and 11 tadpoles from Squaw Hollow Creek (a tributary of Martinez Creek), near Placerville, in July 1942 (UMMZ 2001). The U.C. Museum of Vertebrate Zoology has a historical collection specimen of 1 frog from 2 miles S of El Dorado, on March 31, 1961 (UCMVZ 2015).

¹³⁹ One adult was observed in Camp Creek, 2.1 km S of Iron Mountain, on 6/17/92. In Camp Creek at old gaging site, 1.5 miles SSE of Jenkinson Lake: 1 frog observed and 2 other frogs heard jumping on 7/18/92; 6 tadpoles on 8/10/94; 4 adults and 2 subadults on 7/21/95; 4 adults and 2 large egg masses on 5/12/97; and 2 adults and 8 larvae on 8/25/99. In the North Fork Cosumnes River at the Sweeney Road Bridge Crossing, 2.5 miles SE of Pleasant Valley, an unknown number of tadpoles were observed on 8/11/94. In the Middle Fork Cosumnes River at the Mt. Aukum Road Bridge Crossing, 1.5 miles SSW of Somerset, 10+ tadpoles were observed on 8/23/94. (CNDDDB 2016).

Middle Fork American River drainages. Significant populations were recorded in the Rubicon River and its tributaries Long Canyon Creek and Pilot Creek, through 2007. See the discussion above regarding recent El Dorado National Forest surveys.

Placer County

North Fork American River

There is a historical collection record of 6 frogs from 3.0 mi NW of Cool, on April 11, 1952 (UCMVZ 2015).

Recent data suggest that the species still persists in a dozen or so localities in the foothills in Placer County, particularly the undammed North Fork American River (Lehr 1998; PLSWG 2002). In the 1990s and 2000s, there were scattered observations of small populations of frogs in Placer County in the North Fork American River and tributaries (including Canyon Creek, Codfish Creek, North Fork of the North Fork, Shirttail Creek and Yankee Jim Creek); one significant population of frogs was documented along the North Fork American River mainstem in 2007 (CNDDDB 2016).¹⁴⁰ Foothill yellow-legged frogs were reported to be “common” in the Shirttail Creek tributary during surveys on August 29 and 30, 2007 (CBI 2008).

Middle Fork American River

There were scattered observations of small populations in the 1990s and 2000s in the Middle Fork American River and tributaries (including North Fork of the Middle Fork,

¹⁴⁰ Along the North Fork American River, Mumford Bar Trail to 1 km E of Andrew Gray Creek, 10-20 adults and some juveniles were observed on 9/9/94 (CNDDDB 2016). The California Academy of Sciences has a juvenile frog specimen (CAS #205873) collected from the Shirttail Creek tributary, upstream from Sugar Pine Reservoir, on 7/2/98 (CAS 2001; CNDDDB 2016). A single adult was found along the Yankee Jim Creek tributary in Auburn State Recreation Area (2.75 air miles E of Weimar), on 4/20/00 (CNDDDB 2016). Along the North Fork American River, just downstream from the Ponderosa Way Bridge (9 miles NE of Auburn), 5 adults and 8 subadults were observed on 5/26/07; and more than 10 adults and over 100 young-of-year on 10/4/07 (CNDDDB 2016). Along the North Fork American River, about 0.4 mile upstream of Iowa Hill Road crossing (about 1.4 miles ENE of Colfax), 2 metamorphs were observed on 9/8/06; and 20 adults and 5 egg masses were observed nearby on 5/9/08. In an unnamed tributary to the North Fork American River, 1 mile NE of Colfax, 2 adults were observed on 8/9/07; along the North Fork American River, about 0.23 air miles N of Snakehead Point (about 3.6 miles SE of Dutch Flat), more than 10 adults and 100 young-of-year were observed incidentally on 8/31/07; at Codfish Falls, about 0.35 mile N of Codfish Creek and North Fork American River Crossing (about 1.9 miles E of Applegate), 4 adults were seen on 4/10/08; along an unnamed tributary just NE of the North Fork American River confluence (about 0.5 mile SSE of Dinner Tree and 2 miles ESE of Colfax), 2 gravid female adults and 1 unknown adult were observed on 4/26/08; on Moody Ridge, about 1.6 miles S of Alta and 2.1 miles SE of Dutch Flat, 2 adults and 13 juveniles were observed on 6/18/08; on a tributary about 0.3 miles W of the North Fork of the North Fork American River and Fulda Creek Crossing (about 1.7 miles SSE of Blue Canyon), single adult frogs were observed on 6/18/08 and 7/24/08; at Euchre Bar, where the North Fork of the North Fork American River and the North Fork American River branch (3 miles ESE of Alta), 13 adults, 12 juveniles, 310 larvae and 2 egg masses were observed on 6/17/08; along Canyon Creek and I-80 (about 0.8 mile ESE of Alta and 2.2 miles E of Dutch Flat), 1 juvenile was observed on 6/26/08, and 1 adult on 8/25/08. (CNDDDB 2016). North Fork American River, between Shirttail Canyon and Bunch Canyon (about 1.1 miles ENE of Big John Hill and 4.5 miles SE of Colfax) in 2008: Site A had 115 adults, 5,334 non-adults and 16 egg masses during three visits on 5/18/07, 6/4/07 and 8/20/07; and 11 adults on 5/17/08. Site B had 48 adults, 713 non-adults and 3 egg masses during three visits on 5/18/07, 6/4/07 and 8/20/07. Site C had 17 adults, 52 non-adults and 0 egg masses during three visits on 5/18/07, 6/4/07 and 8/20/07 (CNDDDB 2016).

Skunk Canyon Creek); moderate populations in Skunk Canyon Creek in 2002 and North Fork of the Middle Fork in 2007 (CNDDDB 2016).¹⁴¹

Dry Creek

ECORP (2006) did not detect foothill yellow-legged frogs during 2006 surveys in 3 reaches of the Clover Valley Creek tributary.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 2 of 4 historical locations (50%) in Placer County. Small populations were documented in the 1990s and 2000s in the North Fork and Middle Fork American River drainages; with moderate populations along the North Fork American River mainstem and North Fork of the Middle Fork in 2007 and in Skunk Canyon Creek in 2002. See the discussion above regarding recent Tahoe National Forest surveys.

Nevada County

Yuba River

Mainstem Yuba

There is a historical collection specimen from the Deer Creek tributary drainage, in Olympic Park in Nevada City, from June 1903 (CAS 2001).

Middle Yuba

The species was reported (age/number not given) from the Kanaka Creek tributary in 1991 and Fellers documented *R. boylei* in small numbers from 1996-1997 at several sites along the Middle Yuba River and tributary Grizzly Creek (CNDDDB 2016).¹⁴²

PG&E documented foothill yellow-legged frogs in 2008 in the Middle Yuba River below Milton Diversion Dam: relatively high numbers of egg masses and tadpoles were seen in the reach below the diversion dam downstream to Wolf Creek; and the species was

¹⁴¹ A single adult frog was collected (CAS 205859) from Skunk Canyon Creek, upstream from Mosquito Ridge Road, on 6/30/98; 26 adults were observed in this location on 7/18/02 (CAS 2001 CNDDDB 2016). The California Academy of Sciences has 1 frog larva specimen (CAS #206178) collected from the North Fork of the Middle Fork American River on 7/8/98 (CAS 2001); 3 adults were observed at this location 7/18/02; 15 adults, 3 juveniles and 13 egg masses on 5/23/07; 12 adults and 2,275 tadpoles on 6/8/07; 22 tadpoles and 1 egg mass on 6/11/07; and 23 adults, 2 juveniles, 115 young-of-year and 53 tadpoles on 8/24/07 (CNDDDB 2016). A single adult was observed on Ralston Ridge, about 1.3 miles E of the Middle Fork American River at Rubicon River (about 3.15 miles SE of Michigan Bluff), on 6/24/01 (CNDDDB 2016). Fellers made numerous observations in 3 sites along the Middle Fork American River, upstream of the confluence with the Rubicon River (Fellers site ID: T-562): 3 adults, 1 subadult and 6 larvae on 7/17/02; 1 adult, 5 subadults and 1 juvenile on 8/29/02; 1 adult on 5/22/07; 1 adult and 2 egg masses on 6/7/07; 1 adult, 28 young-of-year, 2 tadpoles and 2 egg mass on 8/21/07; 1 juvenile on 6/12/07; and 1 adult on 8/29/07 (CNDDDB 2016).

¹⁴² In the Kanaka Creek tributary (4 miles NNW of North Bloomfield) on 7/18/91 (CNDDDB 2016). Along Grizzly Creek (Fellers site ID# T-010), about 0.15 mile W of Grizzly Creek at Tyler Foote Road (1 adult and 4 subadults on 6/2/97); along Grizzly Creek (Fellers site ID# T-009), about 0.6 mile E of Grizzly Creek at Tyler Foote Road (2 subadults on 5/29/97); and in Barnhouse Ravine (Fellers site ID# P-347), about 0.9 air mile S of Barnhouse Ravine at Middle Yuba River (1 adult on 9/5/96) (CNDDDB 2016).

observed upstream from Our House Reservoir to about river mile 30, between Wolf Creek and East Fork Creek (FERC 2013).¹⁴³

South Yuba

There are historical collection records from 1967-1973 in the South Yuba River and tributaries Poor Man Creek and Washington Creek, along with a report of “many” frogs seen in Poor Man Creek in 1967 (CAS 2001; UCMVZ 2015).¹⁴⁴

Yarnell (1999) noted the species in Shady Creek, a tributary to the South Yuba River. There were small populations documented from 1991-2008 in the South Yuba River and tributaries Diamond Creek, Logan Canyon, Poor Man Creek and Washington Creek (CAS 2001; CNDDDB 2016).¹⁴⁵

PG&E documented small numbers of foothill yellow-legged frogs in the South Yuba tributary Canyon Creek from 2008-2009 (FERC 2013).¹⁴⁶ Focused surveys by PG&E in 2008 also identified numerous small to moderate populations throughout the South Yuba River drainage, including in the Canyon Creek and Shady Creek tributaries (CNDDDB 2016).¹⁴⁷

¹⁴³ Along the Middle Yuba River about 2.75 miles SE of Alleghany, including Buckeye Ravine, Wolf Creek and Mohawk Ravine (9 adults, 6 juveniles, 80 larvae and 23 egg masses on 6/19/08; 5 adults, 2 juveniles, 140 larvae on 7/8/08; 10 larvae on 8/5/08; and 10 adults, 151 juveniles and 138 larvae on 9/5/08); at the Middle Yuba River and Indian Creek Crossing, upstream of Our House Dam (1 adult and 1 juvenile on 6/9/08; 45 adults and 58 juveniles on 6/24/08; 12 adults and 15 juveniles on 7/14/08; 12 adults and 17 juveniles on 7/22/08; 20+ frogs and dead larvae and juveniles on 8/22/08; 20 adults and 114 juveniles on 8/27/08; and 27 adults and 324 juveniles on 9/3/08); at Chinese Bar and National Gulch Crossing (2 adults and 3 juveniles on 7/7/08; 2 adults and 1 juvenile on 7/16/08; and 4 adults on 9/4/08); and about 0.3 miles ENE of Middle Yuba River and National Gulch Crossing (1 adult on 7/16/08; and 2 adults and 2 larvae on 9/4/08) (CNDDDB 2016).

¹⁴⁴ 1 frog from Poor Man Creek near the confluence with the South Yuba River, in September 1967 (CAS 2001); 13 frogs (MVZ 136314-136326) from South Yuba River at Washington and in Washington Creek on July 15, 1973 (UCMVZ 2015).

¹⁴⁵ In the South Yuba at Poorman Creek, 1 adult and 1 juvenile (CAS 203444, 203450) collected on 8/28/97; 2 adults seen on 7/5/99; and 4 adults, 24 juveniles, 11 larvae and 1 unknown seen in 2008 (CNDDDB 2016). In South Yuba River at Washington and in Washington Creek, 2 adults seen on 6/13/97; 3 adults seen in Washington Creek, 1 mile S of Washington, on 5/23/91 and 1 frog in 8/97 (CNDDDB 2016). One adult in an unnamed tributary to Diamond Creek on 6/3/91; 1 adult 3 miles W of Lake Spaulding on 6/5/91; 1 adult in a Diamond Creek tributary on 6/10/91; 1 adult near Zeibright Mine on 6/19/92; and 6 frogs in South Yuba River State Park on 9/6/06 (CNDDDB 2016). Fellers documented 1 adult and 1 subadult along Logan Canyon (site ID# T-025) on 6/13/97; and 10 adults, 24 juveniles, 64 larvae, 1 egg mass and 1 unknown frog in the South Yuba River at Scotchman Creek (Site ID# T-024) in 2008 (CNDDDB 2016).

¹⁴⁶ Below Towle Canal Diversion Dam frogs were found twice in 2008 and once in 2009 in the downstream portion of this reach; and frog egg masses were found in Canyon Creek about 9.3 miles downstream of Bowman-Spaulding Diversion Dam (FERC 2013).

¹⁴⁷ At the Humbug Creek Confluence (3 juveniles on 4/3/04; 3 adults, 3 juveniles and 6 egg masses on 6/2/08; 5 adults, 2 juveniles and 22 larvae on 6/23/08; 32 adults, 159 juveniles and 2 larvae on 9/16/08; and 117 juveniles on 9/16/08); South Yuba at Holbrook Flat (6 adults, 340 larvae and 1 egg mass on 6/14/08; and 25 juveniles and 2 larvae on 8/26/08); in the Canyon Creek tributary (15 adults observed cumulatively during visits on 6/4/08, 6/12/08, 6/19/08, 7/3/08, 7/10/08 and 8/5/08); at Purdon Creek Crossing (26 adults, 11 juveniles and 1 egg mass on 5/20/08; 15 adults, 3 juveniles, 2 larvae and 2 egg masses on 6/10/08; and 11 adults, 79 juveniles and 2 larvae on 9/12/08); along Canyon Creek (12 adults, 3 juveniles and 1 egg mass on 6/11/08; 1 adult on 7/28/08; 4 adults, 5 juveniles and 73 larvae on 8/12/08; and 2 adults and 13 juveniles on 9/18/08); at Spring Creek Crossing (41 adults, 8 juveniles and 310 larvae on 6/20/08; 32 adults, 4 juveniles and 29 larvae on 7/8/08; and 57 adults, 54 juveniles and 1 larva 9/9/08); about 0.3 mile E of South Yuba River and Diamond Creek Crossing (2 adults, 3 juveniles and 4 egg masses on 6/5/08); about

Bear River

The City of Grass Valley noted declining populations of foothill yellow-legged frog in several creeks in the Grass Valley area: the lower portion of Wolf Creek, Squirrel Creek and South Fork Wolf Creek (Grass Valley 2000).

PG&E documented all life-stages of foothill yellow-legged frogs in “moderate to high numbers” from 2002-2009 in the Bear River below Dutch Flat Afterbay Dam, and a population in the Steephollow Creek tributary (FERC 2013; CNDDDB 2016).¹⁴⁸ The population at PG&E site 1A was very large (349 adults, 2,082 juveniles and 1,063 larvae in August 2008). Small populations were documented in the Greenhorn Creek tributary from 1997-2009 (CNDDDB 2016).¹⁴⁹ There were additional observations from 2007-2008 of small populations along the Bear River (CNDDDB 2016).¹⁵⁰

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 2 of 5 historical locations (40%) in Nevada County. There were small populations documented in the 1990s and 2000s in the Middle Yuba, South Yuba and the Bear River drainages, with a large population in the Bear River and several moderate populations in the South Yuba and tributaries in 2008. See the discussion above regarding recent Tahoe National Forest surveys.

Sierra County

There are no known historical museum collection specimens from Sierra County.

North Yuba River

Small populations were documented in the North Fork Yuba River and a dozen tributaries (Cherokee Creek, Downie River, Fiddle Creek, Goodyears Creek, Grizzly Creek, Humbug Creek, Oregon Creek, Ramshorn Creek, Saint Catherine Creek, Slate

1.1 miles W of South Yuba River and Fall Creek Crossing (1 adult on 6/5/08; and 7 adults, 12 juveniles and 5 larvae on 7/7/08); at Fall Creek Crossing (8 adults, 6 juveniles & 4 egg masses on 6/16/08); and along Shady Creek (39 adults and 19 larvae on 8/12/08) (CNDDDB 2016).

¹⁴⁸ At PG&E site 1A (2 adults/500 juveniles on 10/31/02; 45 adults/36 juveniles/85 larvae/66 egg masses on 6/5/02; and 49 adults/1,885 juveniles/1 larva in 10/03); at PG&E site 1B (67 adults/83 juveniles/188 larvae/2 unknown in 6/08; 22 adults/288 juveniles/187 larvae/15 egg masses/321 unknown in 6/08; 349 adults/2,082 juveniles/1,063 larvae in 8/08; 250 larvae in 9/08; and 10 larvae in 8/09); at PG&E site 2 (6 adults on 11/1/02; 9 adults/2 juveniles on 6/4/02; 7 adults/5 juveniles/1 egg mass on 6/17/02; and 9 adults/5 juveniles on 10/303); and at PG&E site 3 (11 adults/10 juveniles/7 egg masses on 6/4/02; 13 adults/10 juveniles/7 larvae/23 egg masses on 6/16/02; and 9 adults/59 juveniles/19 larvae on 10/2/03). (CNDDDB 2016)

¹⁴⁹ Three adults and 720 subadults seen along Greenhorn Creek (Fellers site ID# T-114, about 0.6 miles downstream from the South Fork Confluence) on 9/27/97, 2 adults (MRJ #1484, CAS #238587) collected on 8/13/99, and 80 larvae observed on 6/9/00; 4 adults observed in the Missouri Canyon Creek tributary (0.75 mile NW of Pleasant Peak) on 11/2/01; and 3 adults observed along Greenhorn Creek (0.5 mile SW of Arrowhead Mine and 1.5 miles E of Highway 174), on 8/11/09 (CNDDDB 2016).

¹⁵⁰ Stump Canyon (2 adults on 5/22/08; 3 adults and 1 juvenile on 9/11/08; and 2 adults on 9/15/08); about 2.7 miles N of Hayford Hill and 4.1 miles NE of Chicago Park (12 larvae on 6/25/08); about 1.8 miles W of Colfax and 2.8 miles SSW of Chicago Park (2 juveniles on 6/15/08); about 2 miles SW of Colfax and 4.4 miles SSW of Chicago Park (1 juvenile and 1 unknown age on 6/9/08; 5 adults and 3 juveniles on 6/24/08; and 6 adults on 8/28/08); and at Dog Bar Bridge (2 juveniles on 9/28/07; 1 juvenile on 6/2/08; and 1 juvenile on 6/15/08)(CNDDDB 2016).

Creek, Woodruff Creek and Willow Creek) in the late 1990s and early 2000s (CAS 2001; CNDDDB 2016).¹⁵¹

Middle Yuba River

Small populations were documented from 1997-2008 in 3 tributaries of the Middle Yuba River: Grouse Creek, Kanaka Creek and Wolf Creek (CNDDDB 2016).¹⁵²

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 1 of 4 historical locations (25%) in Sierra County. Small populations were documented in the late 1990s and 2000s in the North Fork Yuba River and a dozen of its tributaries, as well as 3 tributaries of the Middle Yuba River. See the discussion above regarding recent Tahoe National Forest surveys.

Yuba County

Yuba River

There are historical collection records from 1899-1938 in the North Fork Yuba River and tributary Moonshine Creek (USNM 2001).¹⁵³

¹⁵¹ In the North Fork Yuba River, from the mouth of Humbug Creek to Devils Canyon Creek, 1 adult (CAS #202875) was collected on 5/17/97; 3 adults and 1 juvenile (CAS #205943-205946) were collected on 7/14/98; and 1 adult (CAS #205953) was collected on 7/15/98 (CAS 2001; CNDDDB 2016). In Cherokee Creek, 1 subadult (CAS #203371) was collected on 8/21/97 (CAS 2001; CNDDDB 2016). In the Downie River along Sailor Ravine, Fellers (site ID# T-058) observed 1 adult on 7/22/97 (CNDDDB 2016). In Fiddle Creek, 1 adult (CAS #203284) was collected at Fiddle Creek Ridge Trailhead on 7/21/97; and at Fiddle Creek Campground at North Yuba River near Highway 49, Fellers (site ID# T-050) observed 3 adults and 5 subadults on 7/7/97 (CAS 2001; CNDDDB 2016). In Goodyears Creek (about 0.4 air mile NNE of North Yuba River confluence and 0.5 mile N of Goodyears Bar), Fellers (site ID# Y-821A) observed 3 subadults on 6/19/97 (CNDDDB 2016). In Grizzly Creek, 1 adult (CAS #202921) was collected at Pike City Road on 5/20/97; and along an unnamed tributary of Grizzly Creek (about 0.5 mile E of Pike and 1.1 miles ESE of Alaska Peak), Fellers (site ID# T-014) observed 1 subadult on 6/10/97 (CNDDDB 2016). In Humbug Creek, at the mouth at the North Fork Yuba River, frogs were collected in May 1997 and July 1998 (CAS 2001). Fellers (site ID# T-016C) observed 1 adult along Oregon Creek (about 0.4 mile NE of American Flat, and 1.2 miles NNW of Alleghany) on 6/8/99 (CNDDDB 2016). In Ramshorn Creek (about 0.35 air mile NNW of North Yuba River Confluence and 0.35 mile NW of Ramshorn Campground), Fellers (site ID# Y-822) observed 3 adults on 5/24/95 (CNDDDB 2016). In Saint Catherine Creek a frog was collected on 7/21/98 (CAS 2001). In Slate (Castle) Creek (0.7 km N of Slate Castle), 2 adults were observed on 9/2/92; and along a tributary to Slate Creek (about 0.8 mile W of Little Table Rock Reservoir), 1 adult and 2 metamorphs were observed in October 2000 (CNDDDB 2016). In Willow Creek (In Oak Valley, about 0.4 mile W of North Yuba River and Indian Creek Crossing), 2 adults and 6 juveniles were observed on 8/21/08 (CNDDDB 2016). In Woodruff Creek, approximately 2.0 miles S of Goodyear's Bar, 3 adults were observed on 8/25/92; in an unnamed tributary to Woodruff Creek along Mountain House Road, 1 juvenile (CAS #202880) was collected on 5/31/97 and 1 adult (CAS #202918) on 5/20/97; and 0.6 mile S of Goodyear's Bar, Fellers (site ID# T-031) observed 3 adults on 6/18/97 and 1 adult (CAS #203285) was collected on 7/21/97 (CNDDDB 2016).

¹⁵² In Grouse Creek, just N of Squirrel Creek, Fellers (site ID# T-034) observed 3 subadults on 6/30/97; and in the Grouse Creek headwaters (along Forest Road 180-8), 1 adult was observed on 6/26/08 and 2 adults on 8/4/08 (CNDDDB 2016). In Kanaka Creek, 5 miles N of North Bloomfield (Fellers site ID# T-092B), 1 adult (CAS #203363) was collected on 6/16/97; 24 adults and 1 subadult on 6/8/99; 7 adults on 6/16/00; and at a nearby site 2 adults and 5 juveniles on 5/14/08 (CNDDDB 2016). Along Kanaka Creek (about 1.7 miles SW of Alleghany, and 2.7 miles SSW of Forest), Fellers (site ID# T-092) observed 1 subadult and 6 larvae on 8/26/97 (CNDDDB 2016). Along an unnamed tributary to Kanaka Creek (at Pliocene Ridge), 3 adults were observed on 5/14/08; 2 adults on 5/21/08; 1 adult on 6/26/08; and 1 adult on 8/4/08 (CNDDDB 2016). In Wolf Creek on 6/19/08, 2 adults were observed about 0.8 mile NNE of Wolf Creek and Middle Yuba River Crossing, and 1 adult and 1 juvenile about 1.1 miles NNE of Wolf Creek and Middle Yuba River Crossing (CNDDDB 2016).

The species was known to occur in the 1990s in the lower Yuba River at lower elevations (PG&E 2000). Fellers documented small populations from 1996 to 2000 in the Oregon Creek, Willow Creek and Mosquito Creek tributaries to the North Fork Yuba River above New Bullards Bar Reservoir (CNDDDB 2016).¹⁵⁴ There were collections and observations of single frogs in the Slate Creek tributary of the North Fork Yuba River in 1998 and 2006 (CAS 2001; CNDDDB 2016).¹⁵⁵

The Yuba County Water Agency initiated frog surveys as part of a FERC project on the Yuba River, New Bullards Bar Reservoir on the North Yuba River, Middle Yuba River and Oregon Creek in Yuba County (YCWA 2011). The Yuba County Water Agency (2011) cited 7 California Academy of Sciences historic records in Yuba County outside of the project area. Vindum and Koo (1999) cited historical Yuba County records for the drainages of the North, Middle, and South Yuba rivers above the YCWA project reaches. Foothill yellow-legged frog records within the project area were noted in the vicinity of Log Cabin Diversion Dam on Oregon Creek (adults and subadults), and upstream and downstream of Our House Diversion Dam on the Middle Yuba River (YCWA 2011). YCWA (2011) cited 16 Tahoe National Forest reports of *R. boylei* in the project area, mostly in Oregon Creek, North Yuba River, Kanaka Creek, Grizzly Creek, Woodruff Creek, Grizzly Gulch, and the Middle and South Yuba rivers. During stream habitat mapping in 2009, the YCWA incidentally observed *R. boylei* in Oregon Creek and in the Middle Yuba River downstream of Our House Diversion Dam (YCWA 2011).

Dry Creek

There are historical collection records from 1943-1952 in the Dry Creek drainage (CAS 2001; UCMVZ 2015).¹⁵⁶

There are no known recent records from Dry Creek.

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at 2 of 3 historical locations (67%) in Yuba County. Small populations were documented in the 1990s and 2000s in the lower Yuba River, Middle Yuba River, South Yuba River, and the North Yuba River and its tributaries

¹⁵³ Three frogs from Bullards Bar on the North Fork Yuba River in August 1899 (USNM 2001); and 2 frogs from the Moonshine Creek tributary of the Yuba River in September 1938 (USNM 2001).

¹⁵⁴ One adult and 3 subadults along Oregon Creek (Fellers site ID# P-345), about 0.5 mile NNE of Oregon Creek Campground (about 3.7 miles SSW of Camptonville), on 9/6/96; 3 adults along Oregon Creek (Fellers site ID# T-016) at Gaging Station (about 0.4 mile S of Camptonville and 0.7 mile NE of Log Cabin Dam), on 6/10/97; 1 adult at the confluence of Willow Creek and Horse Creek (Fellers site ID# T-017) at Pendola Road (about 0.8 mile WNW of Camptonville), on 6/11/97; 1 adult and 1 subadult along Mosquito Creek at Oregon Creek (Fellers site ID# T-035), near Hwy. 49 and Celestial Valley (about 2.8 miles SSW of Camptonville), on 6/24/97; and in Oregon Creek at Middle Yuba River (Fellers site ID#), Oregon Creek Campground (about 1.5 miles SW of Celestial Valley and 4.2 miles SSW of Camptonville), 7 adults observed on 6/7/99; and 10 adults and 18 subadults observed on 6/15/00 (CNDDDB 2016).

¹⁵⁵ Collections were made from two locations above Slate Creek Reservoir in September 1998 (CAS 2001). An adult female was observed along a tributary to Slate Creek, just E of the confluence (about 0.8 mile WNW of Poverty Hill), in July 2006 (CNDDDB 2016).

¹⁵⁶ Two adults, 2 juveniles and larva from between Brownsville and Challenge (3 miles W of Challenge), in September 1943 (CAS 2001); and 9 frogs from 2.6 miles ENE of Rackerby (possibly in the Dry Creek drainage) on March 2, 1952 (UCMVZ 2015).

(notably Slate Creek and Oregon Creek). See the discussion above regarding recent Tahoe National Forest surveys.

Butte County

Mud Creek/Rock Creek

There are historical collection records from 1945-1952 in Mud Creek and Rock Creek (UCMVZ 2015).¹⁵⁷

Gallaway (1999) noted that foothill yellow-legged frogs had been observed in Mud Creek and Rock Creek. There are no more recent observations known.

Big Chico Creek

Twelve frogs were collected from Big Chico Creek, 8 miles NE of Chico, on October 21, 1945 (UCMVZ 2015).

Gallaway (1999) noted foothill yellow-legged frogs were common along Big Chico Creek within the Big Chico Creek Ecological Preserve, and had been observed along small tributaries. There are no more recent observations known.

Butte Creek

There are historical collection records from 1945-1953 in Butte Creek and Little Butte Creek (UCMVZ 2015).¹⁵⁸ Lassen and Plumas National Forests (LND and PNF 1999) noted Chico State University museum records of historical occurrences of *R. boylei* downstream of the forest boundary in Butte Creek.

Resurveys from 1973-1978 revealed the species had been extirpated from former the former site on Little Butte Creek (Hayes et al. 2013). Systematic surveys on the Plumas National Forest in the late 1990s (Koo and Vindum 1999) failed to locate the species at 2 historical sites in Little Butte Creek. *Rana boylei* was still present in Butte Creek and at Centerville in the 1990s (PG&E 2000). There are no recent observations known.

Dry Creek

There are historical collection records from 1937-1950 in Dry Creek (CAS 2001; UCMVZ 2015).¹⁵⁹

¹⁵⁷ Single frogs collected from Richardson Springs along Mud Creek, on September 14, 1945 and April 1, 1952; and 3 frogs at Cohasset Pioneer Spring, in the Anderson Fork of Rock Creek, in April 1950 (UCMVZ 2015).

¹⁵⁸ Two frogs were collected from De Sabla on October 21, 1945; 1 frog at Centerville covered bridge on January 2, 1946; 3 frogs from Little Butte Creek, and at and near Magalia Dam and Reservoir, on March 23, 1946; 4 frogs from 1.6 miles W of De Sabla on March 1, 1952; 1 frog N of Butte Creek, 7 miles E of Chico, on April 22, 1952; and 1 frog from Magalia on May 2, 1953 (UCMVZ 2015).

¹⁵⁹ Three frogs were collected near Cherokee ("over the divide in the Sacramento River drainage" presumably in the Dry Creek tributary) in April 1937 (CAS 2001); 1 frog was collected from Cherokee on May 18, 1946; and 1 frog from Cole Canyon Falls, 3 miles S of Pentz, on April 20, 1950 (UCMVZ 2015).

Systematic surveys on the Plumas National Forest in the late 1990s (Koo and Vindum 1999) failed to locate the species at one historical site in Dry Creek. A single adult frog was caught and released in the West Branch Clear Creek tributary of Dry Creek on 10/26/15 (CNDDDB 2016).

Cottonwood Creek

Slevin (1928) noted that foothill yellow-legged frogs were found historically in Chamber's Ravine N of Oroville (in the Little Cottonwood Creek drainage); this probably referred to 6 frogs collected from May 26-29, 1912 (UCMVZ 2015).

Rana boylei was reported to be present in the 1990s in Cottonwood Creek in Coal Canyon, W of Lake Oroville (PG&E 2000). There are no recent observations known.

Feather River

Main Feather River

Three frogs were collected from Bidwell Bar Park on March 9, 1941 (UCMVZ 2015).

West Branch Feather

In the West Branch Feather River drainage, 13 frogs were collected from Cherokee Creek in April 1937 (CAS 2001).

Rana boylei was reported to be present in the 1990s in the West Branch Feather River near Lime Saddle (PG&E 2000). Systematic surveys on the Plumas National Forest in the late 1990s (Koo and Vindum 1999) failed to locate the species at 1 historical site in the West Branch Feather River. There are no recent observations known.

Middle Fork Feather

There is a historical collection record of a single frog (MVZ 117615) from the Little North Fork of the Middle Fork Feather River, 2 miles upstream from the junction with the Middle Fork Feather River, on November 29, 1971 (CNDDDB 2016; UCMVZ 2015).

Fellers documented small populations from 1996-1998 in the Middle Fork Feather River, South Branch Middle Fork Feather River and Fall River (CAS 2001; CNDDDB 2016).¹⁶⁰

South Fork Feather

There are collection records from 1998-1999 in the South Fork Feather River drainage (CAS 2001; CNDDDB 2016).¹⁶¹

¹⁶⁰ Fellers observed 2 adults and 1 subadult along the Middle Fork Feather River at the Little North Fork of Middle Fork Feather River (near Milsap Bar and Gauging Station; Fellers site ID# U-033), on 6/25/96; 5 adults along the South Branch Middle Fork Feather River at Milsap Bar Campground (just NE of the confluence with the Middle Fork Feather River; Fellers site ID# U-032), on 6/25/96; and 1 adult along Fall River about 0.8 mile NNE of Feather Falls (about 1.2 air miles NE of Middle Fork Feather River confluence; Fellers site ID# U-036), on 7/2/96 (CNDDDB 2016). There are collection records of: 1 subadult (CAS #205588) from E of Milsap Bar along Forest Road 22N62, on 6/2/98; and 1 subadult (CAS #205590) from along Forest Road 22N62, 0.8 mile NE of Milsap Bar, on 6/2/98 (CAS 2001; CNDDDB 2016).

North Fork Feather

A single frog was collected from Yankee Hill on April 14, 1950 (UCMVZ 2015).

The species was reported during 1994 surveys in the Bear Ranch Creek and Flea Valley Creek tributaries to the North Fork Feather River; and a single frog was observed in a pool along an unnamed tributary to the North Fork Feather River (about 1.7 miles ESE of Jarbo Gap) on 7/27/98 (CNDDDB 2016). One adult male and 1 adult female frog were observed in the North Fork Feather River in the vicinity of Bardees Bar (about 1.15 miles W of Hungary Hunt Peak and 1.75 miles WSW of Big Bar Mountain), in May 2008 (CNDDDB 2016).

Honcut Creek

A single frog was collected from 3.1 miles NE of Bangor, in the South Honcut Creek drainage, on March 2, 1952 (UCMVZ 2015).

There are no recent observations known.

Recent status: As late as the 1980s, *R. boylii* was found in Butte County in “most drainages” east of the Central Valley floor at elevations as low as 250 feet (Hayes and Cliff 1982). Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 5 of 17 historical locations (29%) in Butte County. No foothill yellow-legged frogs were found within Butte County within during Lassen and Plumas National Forest surveys conducted from 1990-1998, despite surveys of areas of potentially suitable habitat within the elevational range of the species (LNF and PNF 1999). See the discussion above regarding recent Plumas and Lassen National Forest surveys.

Plumas County

Feather River

There are historical collection records from 1899-1952 in the Rice Creek tributary of the North Fork Feather River; the Rock Creek, Spanish Creek, Indian Creek and Last Chance Creek tributaries of the East Branch of the North Fork Feather River; and the Onion Valley Creek tributary of the Middle Fork Feather River (CAS 2001; UMMZ 2001; UCMVZ 2015).¹⁶²

¹⁶¹ One adult male (CAS #206366) from Oroleve Creek along Forbestown Dam Road on 10/8/98; and 1 adult male (CAS #209716) from the South Fork Feather River at Forest Road 22N24, on 8/11/99 (CAS 2001; CNDDDB 2016).

¹⁶² Nine frogs from Quincy (possibly the Spanish Creek tributary of North Fork Feather River) in September 1899 (CAS 2001); 1 frog from Meadow Valley (likely Meadow Valley Creek) in June 1924 (CAS 2001); 1 frog from Feather River Meadows (the Rice Creek tributary to North Fork Feather River) on 21 July 21, 1938 (UCMVZ 2015); 6 frogs from 1.5 miles WSW of McKesick Peak (likely in the Last Chance Creek tributary of North Fork Feather River, above Lake Almanor) on July 15, 1941 (UCMVZ 2015); 3 frogs from 9 miles NNW of Beckwourth in 1942 (UMMZ 2001); 1 frog from Onion Valley Creek, near Middle Fork Feather River in July 1947 (CAS 2001); 1 frog from 8 miles SW of Quincy Meadows Valley in June 1951 (UMMZ 2001); 4 frogs from 5 miles SW of Quincy Rock Creek in July 1951 (UMMZ 2001); and 28 frogs from 5.5 miles W of McKesick Peak (Last Chance Creek drainage) on May 31, 1952 (UCMVZ 2015).

The California Academy of Sciences has a single specimen (CAS #206271) collected from Spanish Creek on 9/3/98 (CAS 2001; CNDDDB 2016). This is likely the population referred to by PG&E (2000), in a drainage adjacent to the North Fork of the Upper Feather River in 1998. Plumas National Forest surveys of the best, accessible habitat from 1993-1998 located *R. boylii* in Fall River, Little North Fork Feather River, Middle Fork Feather River, and South Branch Middle Fork Feather River; with “numerous” sightings in the Middle Fork, Oroleve Creek and Spanish Creek; and an “abundant” population along the South Fork Feather River in 1995 (LNF and PNF 1999; Vindum and Koo 1999).

There was still an apparently significant foothill yellow-legged frog population in Spanish Creek in the 2000s, since a telemetry study done from June 2005 to 2008 used over 50 adult frogs from a site just N of the confluence of Spanish Creek and Bean Creek, with an unknown number of tadpoles and egg masses also observed at the site (CNDDDB 2016). Questions about the affiliation of the Spanish Creek population with the Sierra Nevada yellow-legged frog (*R. sierrae*) or the foothill yellow-legged frog (*R. boylii*) remained after 1 of 4 samples collected from the Bean Creek tributary was identified as *R. sierrae*, with 3 samples identified as *R. boylii* (Lind et al. 2011). Poorten et al. (2013) did subsequent genetic analysis of 7 frogs from Spanish Creek and demonstrated that they were all *R. boylii*. There is only one other CNDDDB record in Plumas County from the 2000s: 1 adult female observed along the South Fork Rock Creek in June 2005 (CNDDDB 2016).

North Yuba River

Plumas National Forest surveys of the best, accessible habitat from 1993-1998 located *R. boylii* in Flea Creek, Onion Creek and Slate Creek; with “numerous” sightings in Slate Creek and Onion Creek (LNF and PNF 1999; Vindum and Koo 1999). There are recent CNDDDB observations and collections of small numbers of frogs from Slate Creek and Onion Creek in the 1990s and in 2006 (CNDDDB 2016).¹⁶³

Recent status: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 4 of 11 historical locations (36%) in Plumas County. Koo and Vindum (1999) found foothill yellow-legged frogs at only 45% of historical sites on the Plumas National Forest in the late 1990s. In the Feather River drainage, significant populations remain in North Fork Feather River, Middle Fork Feather River, Oroleve Creek, South Fork Feather River and Spanish Creek. In the Yuba River drainage, significant populations remain in Canyon Creek tributaries Slate Creek and Onion Creek. The species appears to be extirpated from most historic sites in the East Branch of the North Fork Feather River, Little Butte Creek, Dry Creek, North

¹⁶³ A single adult was observed along Onion Creek, just E of Diamond Springs Hill (about 0.8 mile NNE of Diamond Ravine confluence) on 6/6/96; 4 adults were observed along Slate Creek about 0.3 air miles SSW of American House Ravine confluence (about 1 mile NNW of Poverty Hill), on 6/19/96 (CNDDDB 2016). The California Academy of Sciences has 6 specimens of adults and larva collected from 6 sites in Onion Creek, Slate Creek and the vicinity of Slate Creek Reservoir from July to October 1998 (CAS 2001; CNDDDB 2016). In Slate Creek N of Forest Road 512 (3 miles SE of Little Grass Valley Reservoir), 1 adult female (CAS #209249) was collected on 7/23/99; and 1 adult and 4 juveniles were observed on 9/3/09 (CAS 2001; CNDDDB 2016). Two subadults were observed along American House Ravine, a tributary of Slate Creek, about 0.4 mile NW of their confluence, in July 2006 (CNDDDB 2016).

Fork Yuba River, and West Branch Feather River (Hayes et al. 2013). See the discussion above regarding recent Plumas and Lassen National Forest surveys.

Population Trends

The best measures of long term (i.e. > 50 yr) population trends for foothill yellow-legged frogs are re-survey efforts at known historical and former localities (e.g. Sweet 1983; Jennings and Hayes 1994; Drost and Fellers 1996; Borisenko and Hayes 1999; Lind 2005; Davis and Olson 2008; Olson and Davis 2009). Over shorter and more recent time scales, several researchers and utility companies have been collecting time series data from egg masses in several northern California rivers (see summary by Peek and Kupferberg 2016).

Across California in the last 25 years, breeding season censuses of *R. boylei* have been conducted by researchers, government agencies, and utility companies, in both regulated and unregulated systems. Because each female *R. boylei* lays one discrete clutch of eggs which is readily visible and countable, it has become standard practice to assess their abundance by taking a census of egg masses. Peek and Kupferberg found information for 50 sites which met the following criteria: multiple visits over the course of the breeding season; experienced surveyors; search reach includes more than a single riffle-pool sequence (i.e. not isolated spot checks of breeding sites). With multiple visits, previously overlooked clutches are found upon return, and therefore detectability is high. Two-way Analysis of Variance was used to compare the natural log of mean breeding density (clutches·km⁻¹), of coastal and montane populations in rivers with three levels of modification by dams: flow regulation by an upstream dam; no flow regulation but embedded in a regulated watershed (e.g. a free-flowing tributary of a regulated river, or upstream of a reservoir); or unaffected locally by flow regulation and habitat fragmentation). The sites used were not a random sample of extant populations and many sites were censused in only one or two years (n=29 sites). Some sites (n=11) were censused more than 10 years. Peek and Kupferberg calculated weighted means using the number of years sampled as a weighting factor. Comparison of weighted and unweighted means (Fig. 5), indicates that any bias inherent in compiling information collected for disparate purposes did not greatly affect the overall pattern. The metric used to indicate population size, clutch density, was significantly lower where streams and watersheds are modified flow regulation and fragmentation of the habitat by reservoirs and dam control of flow.

For populations with four or more consecutive years in the time series, Peek and Kupferberg (2016) found that there was significantly higher inter-annual variability in the regulated river populations. They concluded that when high variability is combined with sparse density, population trends may not be detectable prior to the point when populations fall below critical thresholds.

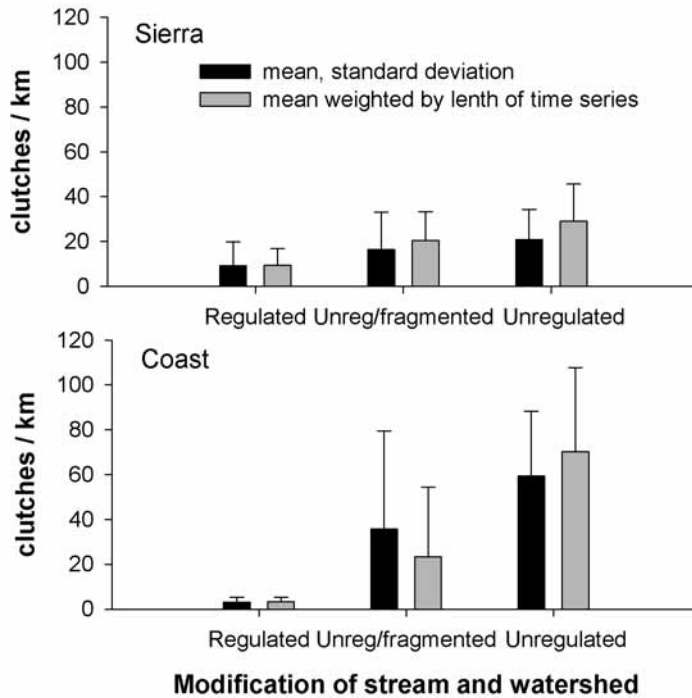


Figure 6. Significant effect of flow regime on breeding population abundance in 50 sampled populations. Two-way ANOVA: Reg vs. Unreg. vs. Unreg. in modified watershed $F_{2,44}=10.9$, $p<0.001$; Coastal vs. Sierran, $F_{1,44}=2.6$, $p=0.1$; Location x Flow regime $F_{2,44}=1.1$, $p=0.33$.

Rangewide

Lind (2005) assessed foothill yellow-legged frogs across their entire geographic range (excluding the lone record from Baja California), with a composite of data from the interval 1996-2000. Lind (2005) revealed that foothill yellow-legged frogs had disappeared from 51 percent of their historical localities throughout their range. This dramatic disappearance across the species' geographic range may even be underestimated (Hayes et al. 2013, 2016). Lind (2005) compiled unique localities in Oregon ($n = 90$) and California ($n = 1,049$), and using a stratified random selection process, chose a subset of 372 California sites and all the Oregon sites for status assessment. To evaluate persistence of frogs at historic sites, she eliminated sites from her sample that were detected after 1975, and used resurveys of sites conducted in the 1980s and 1990s to determine current status. Of the 394 historic sites remaining in her sample, she found frogs were absent from 201 (51%).

California

Jennings and Hayes (1994) comprehensively evaluated the status of *R. boylei* in California: they reviewed all available reports, surveys, and CDFG files and data, conducted field reconnaissance from 1988-1991, searched museum specimens and field notes of naturalists, and relied on their 25 years of field experience for historical locations. Jennings and Hayes (1994) found that the species had been extirpated from at least 225 of 425 known historical locations (53%) and had disappeared from 45 percent of its historic range in California by 1991. While the number of populations is important, population size is also critical (Lanoo 2005). Fellers (2005) found that only 30

of the 213 sites in California with foothill yellow-legged frogs (14%) had populations estimated to be 20 or more adult frogs.

Southern California

A large decline occurred in southern California (Sweet 1983; Jennings and Hayes 1994), with the species likely extirpated from the Tehachapi Mountains southward (Drost and Fellers 1996). Jennings and Hayes (1994) found that the species had been completely extirpated from 21 of 21 historical sites (100%) in the southern transverse ranges. Jennings and Hayes (1994) recommended endangered status for the species in southern California, but foothill yellow-legged frogs are now extirpated from all of southern California.

South Coast

The species is still present but nowhere abundant in coastal California from Monterey County southward to northwestern San Luis Obispo County. Jennings and Hayes (1994) recommended endangered status in central California south of the Salinas River, Monterey County. Jennings and Hayes (1994) found that the species had been extirpated from 81 of 118 historical sites (69%) in southern coastal California. The foothill yellow-legged frog is now nearly extirpated from the south coast region, with the exception of some recent sightings in the Salinas River drainage and some small coastal streams in San Luis Obispo and Monterey counties.

Central Coast/Bay Area

There have been dramatic declines in many parts of the greater San Francisco Bay Area, with the species still present but nowhere abundant. There appear to be significant populations of foothill yellow-legged frogs remaining in the Diablo Range through western Fresno, San Benito, western Stanislaus, Santa Clara and Alameda counties. The species appears to be extirpated from Monterey County north of the Salinas River and western San Joaquin County; and may be near extirpation in western Merced, Contra Costa, Santa Cruz and San Mateo counties.

Marin/Sonoma

The species has been extirpated from many former localities and watersheds in Marin County, with only one significant population remaining in Big Carson Creek. The species is still widely distributed throughout Sonoma County, including in many tributaries of the Russian River drainage, the South Fork Gualala River drainage, the watersheds of the Laguna de Santa Rosa, Petaluma River, Sonoma Creek and Adobe Creek, and in a few coastal streams. However, there are no reports in Sonoma County of populations with more than 50 adults.

North Coast

The largest foothill yellow-legged frog populations in California are in the north coast range, with healthy populations scattered throughout the region. The strongholds for the species are in the Smith River; Red Cap Creek tributary of the Klamath River; South Fork Trinity River; North, Middle and South Forks of the Eel River; Redwood Creek;

coastal tributaries in Mendocino County; and Russian River tributaries. However, only 6 sites in northern California have estimated populations exceeding 100 adult frogs, with an additional 9 sites having > 50 adult frogs (Lannoo 2005). There have been documented declines in this region. Jennings and Hayes (1994) found that the species had been lost from 39 of 165 historical sites (24%) in the north coast of California.

Upper Sacramento River

There have been documented declines in the upper Sacramento River basin: Jennings and Hayes (1994) were able to locate the species during resurvey efforts from 1988-1991 at only 3 of 14 historical locations (21%) in Shasta County and only 3 of 7 historical locations (43%) in eastern Tehama County. The species persists in small numbers in Shasta County in more than three dozen tributaries in the Sacramento River drainage, with larger populations in the Sacramento River (near Dog Creek and Campbell Creek) and in Willow Creek and tributaries. Small numbers of frogs persist in eastern Tehama County in the Battle Creek, Paynes Creek, Antelope Creek, Little Antelope Creek, Mill Creek, and Deer Creek drainages.

Sierra Nevada

The foothill yellow-legged frog was historically common across stream ecosystems of the lower west slope Sierra Nevada (roughly one-quarter of its historical geographic range), but the species now appears to be increasingly rare and near extirpation over at least two-thirds of its Sierra Nevada range (Hayes et al. 2013, 2016).

Historical data indicate that foothill yellow-legged frogs occurred in westside streams at low to moderate elevations all along the west slope of the Sierra Nevada (Storer 1925; Stebbins 1951, 2003; Zweifel 1955; Jennings and Hayes 1994). No quantitative abundance data exist for the Sierran slope prior to the introduction of exotic fishes and major hydrological changes. Storer (1925) suggested that the species was widespread on the Sierran slope, and Zweifel (1955) indicated that the species was at least moderately abundant at scattered locations over that region. Moyle (1973), whose data were collected after significant incursion by introduced fish fauna, indicated that the species was still moderately abundant in foothill streams in the 1970s. In the Sacramento Valley hydrographic basin, low elevation areas make up a large portion of the valley floor, where presumably suitable foothill yellow-legged frog breeding habitat once existed (Hayes et al. 2013, 2016). The species range once extended to the valley floor margin at least in the 1920s, 1930s, and 1940s (Storer 1925, Wright and Wright 1949). The scarcity of records undoubtedly underestimate the historical distribution of the species in this region, as all records are pre-1930, prior to the major hydrological changes and expansion of exotic aquatic predators that changed much of the lowland Central Valley in California to its present condition (Moyle 2002).

Surveys extending back to the 1990s indicated that foothill yellow-legged frogs have disappeared from most of the southern half of the Sierran slope, from approximately Madera County southward (Jennings and Hayes 1994; Jennings 1995, 1996). Throughout the entire Sierra Nevada, the species had been extirpated from 105 of 142 historical sites (74%) and had disappeared from at least 66% of its historical range by the early 1990s (Jennings and Hayes 1994; Jennings 1996).

These data generally agree with the more recent survey efforts of Lind (2005), indicating that foothill yellow-legged frog populations have become even more sparse over this portion of the Sierran slope. Further, evidence exists of considerable local extirpation from different drainage systems in the northern half of the Sierra Nevada, a pattern that becomes less widespread as one moves north (Lind 2005). Fellers (2005) found occupancy of foothill yellow-legged frog sites in the Sierra Nevada was about 12 percent, but historical occupancy of these sites is unknown. Lind (2005) used a randomized selection of 47 historically occupied sites from across the Sierra Nevada and found that 51% (n = 24) of the sites were currently unoccupied. The species' disappearance is more pronounced with decreasing latitude and the species is near extirpation over roughly the southern half of its Sierran range. The species appears to be moving slowly, but inexorably toward extirpation across its Sierran range in a northerly direction (M. Jennings, pers. comm., 2006, as cited in Hayes et al. 2013).

Southern Sierra Nevada

Moyle (1973) found *R. boylei* at only 30 of 95 sites (31%) sampled in the southern and central Sierra Nevada foothills (from the Yosemite area south) in 1970, and believed the species was declining at that time. The species was thought to be near extirpation from the southern Sierra Nevada by the 1990s due to the paucity of observations during focused surveys and resurveys (Fellers 1994; Jennings and Hayes 1994; Fellers and Freel 1995; Drost and Fellers 1996; Fellers 1997). Jennings and Hayes (1994) recommended threatened status in the west slope drainages of the Sierra Nevada. Jennings (as cited by Lannoo 2005) considered the situation for foothill yellow-legged frogs in the southern Sierra Nevada foothills bleak, with no populations that are likely to remain viable. *Rana boylei* is extirpated from Yosemite, Sequoia and Kings Canyon National Parks, and near extirpation in Sequoia and Sierra National Forests, with few extant populations and limited distribution (Hayes et al. 2013, 2016). The species is now nearly extirpated from the southern portion of its Sierra range. The few known populations remaining in the southern Sierra Nevada are in Mariposa County (Merced River and tributaries), eastern Fresno County (Jose Creek), and Tulare County (tributaries of the North Fork Kern River and upper Kern River).

Central/Northern Sierra Nevada

There have been documented declines and apparent loss of many historic populations in the northern and central Sierra National Forests (Hayes et al. 2013, 2016). Although *R. boylei* populations are still extant in many river basins, including the American, Clavey, Cosumnes, Feather, Merced, Mokelumne, Stanislaus, Tuolumne, and Yuba Rivers, the majority of the recent observations in these national forests are of small and scattered populations, with limited evidence of successful reproduction.

There have been severe declines in the central Sierra foothills (Moyle 1973; Drost and Fellers 1996) and populations in the northern Sierra may be in decline as well (Lannoo 2005). Jennings and Hayes (1994) recommended threatened status in the west slope drainages of the Sierra Nevada. At least half of known historical locations have been lost in every northern and central Sierra county (Tuolumne, Calaveras, Amador, El Dorado, Placer, Nevada, Sierra, Yuba, and Butte) except Plumas, and most extant populations are small and scattered. Significant populations remain in El Dorado County (Rubicon River), Placer County (North Fork American River and North Fork of the Middle Fork

American River), Nevada County (Middle Yuba River, South Yuba River and Bear River), and Plumas County (North Fork Feather River, Middle Fork Feather River, Oroleve Creek, South Fork Feather River, Spanish Creek, and Canyon Creek tributaries Slate Creek and Onion Creek).

FACTORS AFFECTING ABILITY TO SURVIVE AND REPRODUCE

The decline of foothill yellow-legged frogs across their range in California can be attributed to a combination of anthropogenic stressors. Primary threats include habitat loss, fragmentation, and degradation of in-stream conditions via water abstraction, flow diversion, and flow regulation.

Extirpation has occurred more frequently downstream of dams than in free-flowing systems and extirpation is positively correlated with the height of upstream dams (Lind 2005; Kupferberg et al. 2012). Dams and reservoir operations suppress winter peak discharges and thus allow woody riparian vegetation to encroach into the active channel. The roots stabilize the cobble and gravel bar features where frogs congregate in groups (called leks) to find mates and to lay eggs. In regulated rivers, vegetation encroachment often eliminates the suitability of these bars for breeding via shading and/or changing bar shape and bank slope. The diminution of winter flooding and conversion of ephemeral water bodies to permanent ones also promotes populations of non-native taxa such as bullfrogs, crayfish, and bass in managed river systems (Fuller et al. 2011). Bullfrogs and crayfish negatively affect amphibian populations in general (Kats and Ferrer 2003) and are implicated in declines of foothill yellow-legged frogs specifically (Moyle 1973; Hayes and Jennings 1986; Kupferberg 1997a). Ill-timed water releases through dams have the potential to create lethal velocities for early life stages and cold hypolimnetic releases shift water temperatures below the thermal tolerances for tadpoles (Catenazzi and Kupferberg 2013).

Generally, activities that disrupt the natural flow and sediment transport regime of rivers, including timing of flows, water depths, velocities, or water temperature can affect foothill yellow-legged frogs (Lind 2005; Yarnell et al. 2010; Kupferberg et al. 2012). Direct and indirect impacts associated with changes to instream flows include: desiccation or stranding of eggs or tadpoles due to rapid reductions in flow, delays in breeding and embryo or tadpole development due to cold water temperatures (Wheeler et al. 2014), declines in algal productivity and shifts in species composition of periphyton (Catenazzi and Kupferberg 2013; Furey et al. 2014), reduced resources for tadpoles, and reduced insect abundance and food-web repercussions. If sufficiently high, reservoir management releases and flow releases to benefit salmonids during the spring of otherwise dry years could dislodge egg masses and displace larvae downstream (Railsback et al. 2016).

In addition to the association of decline with the presence of large dams, demographic patterns of decline also indicate that extirpations are more frequent downwind of regions with extensive aerial spraying of pesticides (Davidson et al. 2002). Experiments show that tadpoles of foothill-yellow legged frogs are sensitive to low concentrations of organophosphate pesticides and their oxon derivatives (Sparling and Fellers 2007, 2009). NatureServe (2011) also notes other threats to the foothill yellow-legged frog from non-selective logging practices, and other habitat degradation and disturbance caused by excessive livestock grazing and in-stream suction dredge mining.

A detailed examination of the threats faced by foothill yellow-legged frogs is provided by Olson and Davis (2009).

Climate Change

Climate change is already causing a rise in temperatures across the United States and an increase in extreme weather events, such as droughts and floods (Parmesan et al. 2000; NSC 2003; CCSP 2008; Karl et al. 2009). Climate change predictions for terrestrial areas in the Northern Hemisphere indicate warmer air temperatures, more intense precipitation events, and increased summer continental drying (Field et al. 1999; Cayan et al. 2005; IPCC 2007). California is likely to see average annual temperatures rise by 1.5 - 4.5°C in the next century, with summer stream flow and soil moisture required for plant growth likely to decrease (Field et al. 1999; Cayan et al. 2008). Since 1895, annual average air temperatures in California have increased by about 1.5 degrees Fahrenheit, with minimum temperatures increasing at a rate almost twice as fast as the increase in maximum temperatures, and warming accelerated over the past three decades in most regions of the state (Kadir et al. 2013). Climate models predict more variable annual precipitation, and decreased spring and summer runoff as a result of lower annual snowpack (Smith and Tirpak 1989; USEPA 1997; Johnson et al. 1999). It is predicted that precipitation will come earlier in the spring in the form of rain rather than snow. Consequently, the hydrograph will shift to earlier snowmelt, lower snowpack, more winter rain, and higher winter storm runoff events (Maurer et al. 2007; Stewart 2009; Young et al. 2009). The low flow season will likely be longer, so water temperatures may be higher which may result in stress for species adapted to more moderate temperature regimes. Spring snowmelt from the Sierra Nevada has already declined over the past century, with changes in the timing and amount of precipitation in California and changes in the timing of Sierra runoff (Aguado et al 1992; Kadir et al. 2013). The portion of Sierra runoff that occurs between April and June has declined by about 9 percent (Kadir et al. 2013).

Climate change is particularly problematic for amphibian populations because they are ectothermic. As such, they are sensitive to changes in air and water temperature, precipitation, and the hydroperiod (length of time and seasonality of water presence); their body temperatures and activity cycles are dependent on the presence of optimal environmental conditions (Lind 2008). Case et al. (2015) evaluated relative sensitivity to climate change of species in Northwestern North America, using a combination of scientific literature and expert knowledge to assess the relative sensitivity to climate change of 195 plant and animal species in the northwestern North America. Amphibians and reptiles were, as a group, estimated to be the most sensitive to climate change.

Some amphibians have shown a trend towards earlier breeding, apparently in response to global warming (Beebee 1995, Blaustein et al. 2001, Gibbs and Breisch 2001). If such shifts in activities occur inconsistently with other ecological events (e.g., emergence of their insect prey), growth and survival rates could be affected. Species associated with ephemeral waters, such as shallow ponds and intermittent streams, may be particularly vulnerable to altered precipitation patterns (Dodd 1997, Lind 2008, McMenamina et al. 2008).

Changes in frequency, duration, and magnitude of droughts or severe winters resulting from climate variability may have considerable negative impacts on foothill yellow-legged frog populations. Population declines of foothill yellow-legged frogs have been attributed in part to extended drought (Jennings and Hayes 1994). Decreases in summer runoff may result in the loss of foraging and refuge habitat for adults and juveniles. As frogs congregate at shrinking pools in rivers as they dry, densities become locally concentrated and the probability of transmission of diseases and parasites increases. Changes in temperature may affect parasite prevalence (Kupferberg et al. 2009a) and pathogen virulence (Carey et al. 1999), making foothill yellow-legged frogs more susceptible to disease. Further, experimental increase in stream water temperature has been shown to decrease invertebrate density and biomass in invertebrates (Hogg and Williams 1996) and may have a negative impact on the foothill yellow-legged frog prey base.

Changes in climatic patterns, particularly those linked to precipitation, may have substantial impacts on foothill yellow-legged frogs. Low precipitation and increased variability in precipitation were both negatively related to frog presence (Lind 2005). Evidence also suggests that low precipitation may increase dam effects (Lind 2005). Climate change is predicted to reduce the habitat suitability for foothill yellow-legged frogs at lower latitudes and elevations. Current foothill yellow-legged frog distribution may be an indication that climate change has already influenced the species (Lind 2005). Although other factors may confound the influence of climate change on distribution patterns, short-term oscillations and drought severity have been greater at lower latitudes in California (Cook et al. 2004), where foothill yellow-legged frogs appear to be in dramatic decline.

Climate change appears to already be a contributing factor in the decline of the foothill yellow-legged frog (Fellers 2005, Olson and Davis 2009). Continued climate change is likely to cause further range contraction for the foothill yellow-legged frog, with loss of southernmost populations, as well as potential habitat shift upward in elevation as temperatures increase and precipitation becomes more variable.

Davidson et al. (2002) examined the spatial patterns of declining frogs in California and hypotheses of spatial patterns of climate change. For foothill yellow-legged frogs, they found a north-to-south gradient of increasing frog losses, consistent with climate change hypotheses (more losses at drier sites to the south). Lind (2005) considered climate change as a potential threat to foothill yellow-legged frog, due to precipitation being associated with frog presence.

Kupferberg et al. (2009a) presented data supporting a link between periods of unusually warm summer water temperatures during 2006 and 2008 in a northern California river, outbreaks of the parasitic copepod *Lernaea cyprinacea*, and malformations in tadpoles and young of the year foothill yellow-legged frogs. *Rana boylei* are likely to have increased vulnerability to pathogens due to projected climate changes. Changes in climatic regimes are likely to increase pathogen virulence and amphibian susceptibility to pathogens (Pounds et al. 2006, Pounds et al. 2007, Gervasi et al. 2008, Alford 2011).

Dams, Water Development and Diversions

Water development and diversions are the primary (and most well-documented) cause of declines in foothill yellow-legged frogs and water developments on natural waterways

have greater potential to alter habitat for the foothill yellow-legged frog than any other risk factor (Hayes et al. 2013, 2016). In California, the Mediterranean climate produces a very distinct hydrologic signature with high and variable water flows in the fall, winter, and spring; and low, receding, stable flows in the summer. Foothill yellow-legged frogs are adapted to this flow regime, especially spring (rain or snowmelt) recession flows (Yarnell et al. 2010). Modifications to that hydrologic regime can disrupt species responses to environmental cues and have direct effects on survival of aquatic life stages.

Water development and diversions result in hydrological changes that chronically affect several aspects of the frog's life history. Hayes et al. (2013, 2016) note recent studies from both regulated and unregulated rivers have demonstrated that both landscape scale changes and small-scale changes in local habitat conditions, such as water velocities, depths, and temperatures, that often result from water management activities, can lead inconsistent environmental cues for frog breeding, lower growth rates for tadpoles, scouring and/or stranding of egg masses and tadpoles, reductions of overall habitat suitability for breeding and rearing, barriers to gene flow around reservoirs, and establishment of non-native predators in reservoirs that then spread into the rivers.

Water developments exist as two major types: impoundments and diversions. Impoundments block streams with a structure such that natural flows are impeded and water is pooled upstream. Impoundment size varies throughout the foothill yellow-legged frog range, ranging from smaller dams created for water gauging stations and improved fisheries to larger dams created for hydroelectric generation or flood control. Diversions are created for the purpose of removing and delivering water to off-site locations. Some diversions are associated with impoundments, whereas others involve pumping water directly from the waterway or indirectly through groundwater pumping. The California Water Plan Update (CDWR 1998) indicates that dams and diversions are found on most Sierra Nevada streams (Moyle and Randall 1998) and a majority of these alterations exist within the elevational range of the foothill yellow-legged frog.

Reservoir placement on Sierran streams has converted many lotic aquatic habitats to lentic conditions, resulting in habitat with reduced flows, increased depths, and altered temperature and dissolved oxygen regimes (Petts 1980, 1984; Mount 1995). These changes result in direct loss of required habitat for stream-dwelling foothill yellow-legged frogs, which have evolved to inhabit free-flowing, well-oxygenated water with coarse substrates. In an evaluation of the distribution of reservoirs in the Sierra Nevada, Kondolf et al. (1996) found reservoirs had eliminated an estimated 9,972 km (6,209 mi) of aquatic habitat. Given the distribution of reservoirs, foothill yellow-legged frogs could have been historically present in much of this lost habitat. Sierran reservoirs currently inundate at least eight sites once occupied by foothill yellow-legged frogs (Hayes et al. 2013, 2016).

Regulation of flows downstream of dams is associated with lower frog abundances, with *R. boylei* breeding populations on average 5 times smaller in regulated rivers than in unregulated rivers (Kupferberg et al. 2012). Lind (2005) previously found an impoundment effect on foothill yellow-legged frogs; the species was associated with streams lacking dams or with streams with small dams located far upstream of foothill yellow-legged frog occupied locations. Lind (2005) found that former yellow-legged frog localities throughout California where frogs are now extirpated were characterized by higher numbers of all dams upstream, greater number of very large dams upstream,

greater maximum height of dams upstream and closer proximity to upstream dams. Large regulated streams typically have substantially lower numbers of foothill yellow-legged frogs than unregulated streams (Hayes et al. 2013, 2016). At least one large reservoir ($\geq 100,000$ ac-ft) exists in the foothill region of every major Sierran stream below 600 m (1,968 ft) (Hayes et al. 2013, 2016). Several major streams (such as the Pit River, Feather River, American River, Mokelumne River, Tuolumne River and San Joaquin River) have two or more reservoirs (of varying size) in linear sequence, and a few large reservoirs also occur at higher elevations on major stream tributaries (Hayes et al. 2013, 2016). Additionally, several hundred medium-sized ($< 100,000$ ac-ft and $\geq 25,000$ ac-ft) and small reservoirs ($< 25,000$ ac-ft) are broadly distributed at elevations below 1,828 m (6,000 ft) over the Sierra Nevada (Mount 1995).

An extensive survey effort by Garcia and Associates (2002) on the North Fork Feather River failed to find foothill yellow-legged frogs in habitats which appeared suitable for this species in the Rock Creek reach. The study areas were below impoundments operated by Pacific Gas & Electric for hydroelectric power generation. However, foothill yellow-legged frogs are present in the Poe reach at decent numbers, and very low numbers in the Cresta reach of the North Fork Feather River (S. Kupferberg, pers. comm., 2016). On the main stem of the Trinity River, northern California, unnatural flow regimes and loss of habitat caused by dam construction are the greatest threats (Ashton et al. 1997). A study on the Trinity River below Lewiston Dam, reported a 94 percent loss of potential breeding habitat after construction of the dam (Lind et al. 1996). After Trinity River flood flows were reduced, there was encroachment by riparian vegetation and reduced cobble/gravel bar formation. Flow releases had been reduced to 10–30 percent of pre-dam operation flows in both total volume and in periodic high flows (i.e., storm runoff) (Lind et al. 1996). Egg masses have been scoured in several years by high late spring releases from Lewiston Dam (Lind et al. 1996). Ellis and Cook (2004) reported half of known egg masses were scoured by five days of high flow releases on the Pit River in California. They suggested duration of high flows and change in current direction (shearing) had a higher impact than overall magnitude. Jackman et al. (2004) also found pulsed flows scoured half of the egg masses on the North Fork Feather River, in only one day. Egg masses may be left to desiccate if receding high flows are poorly timed (Lind et al. 1996; Ashton 1998).

High aseasonal flow releases from dams in late spring sometimes result in scouring of egg masses, whereas receding high flows, if poorly timed, can leave egg masses stranded “high and dry” (Lind et al. 1996). Bobzien and DiDonato (2007) concluded from frog breeding surveys in Alameda Creek in Alameda County, California, that unnatural and consistently higher discharge and irregular flows associated with dam releases appears to be a major factor in poor reproductive conditions for the frog, when compared to stream reaches with natural hydrology.

Kupferberg et al. (2009b), based on review of the literature and FERC-related reports, found foothill yellow-legged frog egg masses are negatively affected by pulsed flows (large magnitude flow fluctuations in rivers with dams) via scouring if flows occur during or after oviposition and desiccation if oviposition occurs during high flows and subsequently drops. Tadpole stranding and potential negative effects on metamorphs have been documented in multiple studies. South Fork Eel River population monitoring shows that the magnitude and timing of spring pulse flows are key factors in survival of eggs and tadpoles. While large magnitude spring pulses decrease egg survival, smaller

pulses later in the spring cause even higher mortality. Fluctuations in population growth are associated with spring pulse events three years prior. Experiments suggest that during pulse flows tadpoles seek refuge from higher velocities in the substrate, but many are swept downstream. Tadpoles confined to refugia face energetic costs in terms of growth and development. Kupferberg et al. (2011) explored the effects of pulsed flows from dams on foothill yellow-legged frog tadpoles, and found that typical velocity increases in near shore habitats (provided for recreational flows for white water boating or peaking releases for hydroelectric power generation) caused tadpoles approaching metamorphosis to be displaced, and that tadpoles exposed to repeated sub-critical velocity stress grew significantly less and experienced greater predation than tadpoles reared at ambient velocities.

Dams not only eliminate habitat and cause local extirpations, and they also interfere with normal dispersal and movements, which can impede recolonization after local extirpations (Fellers 2005; Peek 2010). Kupferberg et al. (2009b) found that water control management that avoids aseasonal flow fluctuations would benefit foothill yellow-legged frogs, and other taxa, whose lifecycles are synchronous with the natural timing of runoff in California's rivers. Most recently, Kupferberg et al. (2012) found that the foothill yellow-legged frog is more likely to be absent downstream of large dams than in free-flowing rivers, and breeding populations are on average 5 times smaller in regulated rivers than in unregulated rivers.

Dam-controlled flows and lack of winter flooding likely results in stable pool areas with established aquatic vegetation (Lind et al. 1996, Kupferberg 1996), and this can increase suitable habitat for exotic species such as bullfrogs (Ashton et al. 1997). Decreased flows can force frogs into permanent pools where they are more susceptible to predation (Hayes and Jennings 1988).

Disease

Amphibian declines in the United States and Panama have been linked to the introduced fungus *Batrachochytrium dendrobatidis* (Bd), which causes chytridiomycosis (Fellers 2001). This disease causes abnormalities in jaw sheaths and teeth rows of tadpoles, and is invariably fatal in populations of some species. Foothill yellow-legged frogs are hosts to this amphibian fungal pathogen. There is conflicting evidence about the lethality of Bd infection for foothill yellow-legged frogs under laboratory conditions (Davidson et al. 2007; Padgett-Flohr pers. comm. to S. Kupferberg) and its population effects are unknown (Fellers 2005). Infection with Bd does appear to have negative effects on growth of *R. boylei* in the lab and in the field (Davidson et al. 2007; Lowe 2009). In laboratory experiments, Davidson et al. (2007) found that chytrid infection reduced growth of newly metamorphosed foothill yellow-legged frogs by approximately one-half and that exposure to the pesticide carbaryl likely increases susceptibility to chytrid infection.

Bd has been detected in *R. boylei* in California (Fellers 2001; Davidson et al. 2007; Johnson and Saulino 2007; Lowe 2007, 2009; Padgett-Flohr and Hopkins 2009; Adams et al. in press). Fellers (2001) sampled 25 counties in California and found chytridiomycosis in six species of amphibians including foothill yellow-legged frogs in 10 counties at 73 sites. Johnson and Saulino (2007) found Bd in all anuran species, including foothill yellow-legged frogs, in and around Pinnacles National Monument, and

at a few sites in the western foothills of the San Joaquin Valley. Foothill yellow-legged frogs infected with Bd have been found at 10 of 12 sites sampled in the Diablo Mountains, San Benito County, and western San Joaquin foothills, Fresno County, California, in 2006 (Lowe 2007, 2009). However, most of post-metamorphic frogs were not infected. All foothill yellow-legged frogs >40 mm were chytrid free.

Histological examination of museum specimens of the foothill yellow-legged frog indicates that Bd has been present since at least 1961 in the Alameda Creek watershed in Alameda County (Padgett-Flohr and Hopkins 2009). In the fall of 2013, foothill yellow-legged frogs in the Little Yosemite reach of Alameda Creek experienced an outbreak of Bd in which dead and dying juveniles were observed (Adams et al. in press), a location where annual amphibian breeding censuses have been conducted since 2003. Adams et al. (in press) attribute the die-off to an outbreak of chytridiomycosis, caused by Bd, in which recently metamorphosed frogs had the highest Bd loads among sampled individuals and was confirmed by necropsy and histological examination. Although chytrid infections have been documented by others over the last decade many miles upstream of Calaveras Reservoir, these were the first indications of the effects of Bd infection among the lotic-breeding frogs downstream of the dam. In contrast to laboratory investigations of *R. boylei* from a North Coast California source population by others, these observations corroborate observations that *R. boylei* is susceptible to the lethal consequences of chytridiomycosis. The outbreak coincided with extremely low stream flows that concentrated frogs in drying pools and expanded the spatial distribution of non-native bullfrogs in the stream network. Infection intensity and prevalence has varied through time, but over a 2 year sampling period (fall 2013 through fall 2015) the strongest predictor of Bd load was the presence of bullfrogs. Bullfrogs may represent a reservoir for Bd when foothill yellow-legged frogs in the population are Bd negative. Over three years of drought, the number of *R. boylei* egg masses per stream kilometer (an index of adult female population size), upstream of the dams that regulate flow, has decreased to the extent that abundance is no longer greater than in the regulated reaches where Bd and anthropogenic stressors occur.

An 11 year study on Bd infections of the related Sierra Nevada yellow-legged frog (*R. sierrae*) and southern mountain yellow-legged frog (*R. muscosa*) revealed the extirpation of over 100 populations since 1997, and after the introduction of Bd (Vredenburg et al. 2009). The disease spread at approximately 1 km a year in an easterly direction. Infections of frog populations reached 100 percent within weeks. All populations were stable prior to the onset of Bd invasion. Although mass die-offs from Bd have not yet been observed for *R. boylei* as they have for other ranid frogs in California (Vredenburg et al. 2010), the recent die-off in Alameda Creek and the potential for catastrophic, population level impacts is concerning.

In the main stem of the Trinity River, there is evidence of fungal infections on foothill yellow-legged frog egg masses (Ashton et al. 1997), possibly with the water mold *Saprolegnia* sp. which Blaustein et al. (1994) and Kiesecker and Blaustein (1997) found to be fatal for other species of frogs.

Known from related species are the bacterial disease “red leg” (*Aeromonas hydrophila*) (e.g., *R. muscosa*, Bradford 1991) and iridoviruses (Ranavirus species), which are a complex of viruses found in frogs and fish (Mao et al. 1999).

Invasive Species

A host of native vertebrates (e.g. birds and snakes) and aquatic invertebrates (e.g. dragonfly nymphs) feed on foothill yellow-legged frogs and their tadpoles (Fitch 1936, 1941; Everdon 1948; Zweifel 1955; Milne and Milne 1980; Nussbaum 1983; Jennings and Hayes 1994; Lind and Welsh 1994; Duellman and Trueb 1986; Jennings 1988; Moyle and Brown 1997; Ashton et al. 1998; Fellers 2005; Olson and Davis 2009), but nonnative predators are a primary threat to the species.

It is well documented that foothill yellow-legged frog adults, larvae, and/or eggs are vulnerable to an array of non-native predators such as predatory fishes, bullfrogs, and crayfish (Moyle 1973; Hayes and Jennings 1986; Lind et al. 1996; Kupferberg 1996b; Ashton et al. 1997; Lind et al. 2003; Fellers 2005; Paoletti 2009; Paoletti et al. 2011). Invasive bullfrogs and crayfish negatively affect amphibian populations in general through direct predation and competition for resources (Hayes 1985; Hayes and Jennings 1986; Jennings 1988; Kupferberg 1996b; Kats and Ferrer 2003). Centrarchid fishes readily eat *Rana* eggs (Werschkul and Christensen 1977) and where introduced into foothill streams may contribute to the elimination of *R. boylei*. Rombough et al. (2005) found that foothill yellow-legged frog abundance and production was inversely related to abundance of smallmouth bass (*Micropterus dolomieu*) and American bullfrogs (*R. catesbeiana*). Borisenko and Hayes (1999) found exotic bullfrogs and fishes occurred significantly more often at historic yellow-legged frog sites in Oregon that lacked yellow-legged frogs during their resurveys. Rombough (2006b) found smallmouth bass were the best predictor of absence of yellow-legged frog in Cow Creek, Oregon, having an inverse relation to the yellow-legged frogs; Rombough also found bullfrogs were negatively correlated with yellow-legged frog distributions. Bullfrogs have been linked to the observed reduction of foothill yellow-legged frog populations in the Sierra Nevada (Moyle 1973). Kupferberg (1997) found foothill yellow-legged frogs to have decreased abundance in stream reaches in northern California occupied by bullfrogs; and that bullfrog larvae perturbed aquatic community structure and exerted detrimental effects on foothill yellow-legged frog populations. Interspecific matings between male foothill yellow-legged frog and female bullfrogs have been observed; these interactions with non-native bullfrogs might reduce the reproductive output of foothill yellow-legged frogs (Lind et al. 2003).

Predation by feral pigs is a concern for *R. boylei* in some locations (Ely 1993, 1994).

The invasive New Zealand mudsnail (*Potamopyrgus antipodarum*) is an emerging concern for California waterways, and according to the United States Geological Survey's Non-Indigenous Aquatic Species Database (Foster et al. 2016) these snails occur in watersheds with extant populations of *R. boylei*. The influences of this snail on *R. boylei* are not known, but experiments have shown that they have negative competitive effects on western toad (*Anaxyrus boreas*) tadpole survival (Bennett et al. 2015). The mudsnails have the ability to reproduce quickly, grow rapidly and mass in high densities, and can alter macroinvertebrate community composition and food web function (Alonso and Castro-Díez 2008).

Kupferberg et al. (2009a) found evidence between unusually warm summer water temperatures and outbreaks of the parasitic non-native copepod *Lernaea cyprinacea*, and malformations in *R. boylei* tadpoles and young of the year.

Livestock Grazing

The potential negative effects of intensive livestock grazing on foothill yellow-legged frogs include: the possibility for cattle directly crushing individuals; trampling of stream banks resulting in soil compaction, loss or reduction in vegetative bank cover, stream bank collapse, and increased instream water temperatures from loss of shade; and added sedimentation of stream segments at crossings or other stream areas if used in a very concentrated manner by livestock for watering or grazing on riparian vegetation.

Further investigation is needed to evaluate the extent of livestock grazing impacts on instream habitat quality and population dynamics of *R. boylei* because research on this topic had been focused on different taxa and the results are equivocal. Grazing of livestock at high densities in the absence of off-channel water sources can result in bank erosion, degrading shorelines and increasing stream sedimentation (Davis and Olson 2009). The Sierra Nevada Ecosystem Project, an assessment of the Sierra Nevada ecoregion, concluded that more open vegetation resulting from overgrazing can expose amphibians in general to predation and desiccation, and direct trampling by livestock is likely an important cause of mortality for some taxa (SNEP 1996). A subsequent fencing/cattle exclusion experiment conducted in Sierran wet meadows, however, showed no effect of grazing on toad populations (McIlroy et al. 2013). Masters (1997b) described the negative impacts of cattle grazing on habitat used by foothill yellow-legged frogs in Jackson Creek, in the Umpqua National Forest, Oregon:

Direct impacts of cattle in riparian areas include crushing eggs and tadpoles of foothill yellow-legged frogs, as well as juveniles and adults...Indirect impacts include alteration and/or elimination of vegetation, alteration of the microhabitat conditions, degradation of water quality, alteration of the structure and composition of the vegetation, and introduction of non-native vegetative species...Increased sedimentation covers up the cobble-sized rocks that the foothill yellow-legged frog requires for breeding, tadpole development, and juvenile and adult habitat. The cowpies and urine degrade the water quality...sedimentation, resulting from cattle grazing...reduces the interstitial spaces available for use by tadpoles and it may inhibit attachment of egg masses.

In some locations and under certain circumstances, managed cattle grazing can help keep vegetation from encroaching into the active channel; too much canopy cover can make sites unsuitably shady for foothill yellow-legged frogs (S. Kupferberg, pers. comm., 2016). Thus, controlled livestock exclosure experiments are needed to determine the net effects of grazing on *R. boylei*.

Logging

Timber harvest without sufficient riparian buffer zones can decrease populations of aquatic amphibians such as the foothill yellow-legged frog by increasing water temperatures to lethal levels and by causing siltation of streambeds (Corn and Bury 1989). Even partial removal of stream canopy can increase water temperatures and decrease relative humidity along the stream corridor in headwater reaches which can

make these areas unsuitable for amphibians (Bury and Corn 1988). High levels of silt inhibit the attachment of frog egg masses to the substrate (Applegarth 1994, Ashton et al. 1997), and excessive accumulation of silt on the egg masses likely has adverse effects on embryo development (Jennings and Hayes 1994). Silt also reduces the interstitial spaces available for use by tadpoles, reduces algal growth on which the tadpoles feed (Power 1990), and can have a significant negative impact on adult frog food resources such as aquatic macro-invertebrates (Petts 1984). Sediment impacts likely adversely affect preferred foothill yellow-legged frog habitat through bed aggradation, surface texture fining or changes in hydraulic geometry (Yarnell 2000).

Marijuana Cultivation

Cultivation of *Cannabis*, i.e. marijuana, is a threat to foothill yellow-legged frogs and their habitat, by the direct effects of illegal and legal water extraction that de-waters the streams where frogs live, introduction of pesticides and chemical fertilizers into waterways, denuding terrestrial habitat adjacent to streams and terracing the slopes, and by promoting the growth of toxic cyanobacteria (Gonsolin 2010; Bauer et al. 2015; Carah et al. 2015; Power et al. 2015). This impact is a major threat to *R. boylei* in Northern California, where marijuana cultivation is concentrated, and the severity of the impacts is exacerbated by ongoing drought conditions. Gonsolin (2010) noted the decline of a *R. boylei* population in the upper Coyote Creek watershed, Santa Clara County, due to impacts from illegal marijuana cultivation.

Mining

Ashton et al. (1997) explained that mining can have deleterious effects on egg masses and tadpoles, as well as disturbing postmetamorphic behavior patterns.

Suction-dredge mining, in which water, sediment, and rocks are vacuumed from portions of streams and rivers, and the spoils re-deposited in the stream (CSLC 1993; Harvey and Lisle 1998), may increase suspended sediment, modify stream geomorphology, directly remove aquatic organisms, and rearrange the substrate of streams (CDFG 1994, 2012). This form of mining may have effects on frog reproduction by disturbing adults during courtship and breeding activities, or disrupting habitat during the reproductive season. Dredging up stream substrates can result in displacement, burial, or suffocation of eggs or tadpoles (CDFG 1994, Harvey and Lisle 1998). Depending on the size and stage of foothill yellow-legged frog tadpoles, they would not be able to swim away from the strong vacuums created by suction dredging as they can be entrained into currents as slow as 0.33 feet per second (Kupferberg et al. 2011). In response to elevated currents, these tadpoles seek shelter in interstitial spaces in the substrate. Because of this behavior, this species is particularly vulnerable to suction of sediments.

Sweet (1992) observed mortality of eggs and larvae of the stream-breeding arroyo toad (*Anaxyrus californicus*); mortality was a direct effect of increased sedimentation that resulted from suction-dredge mining. Suction-dredging may cause movement of instream habitat features such as rock substrates and woody debris, which may be used by foothill yellow-legged frogs for overwintering. Dredging may also affect foothill yellow-legged frog prey base.

Although a moratorium in California currently prohibits CDFW from issuing suction dredge permits (California Fish & Game Code §5653.1, subdivision a), and use of

related equipment in any river, stream, or lake through 30 June 2016 (California Fish & Game Code §5653.1, subdivision b), suction-dredge mining may be permitted in the future. Many of the foothill streams in the northern Sierra Nevada have regulated and unregulated recreational gold mining activities, which alter the streambed and are likely having a serious, negative impact on the frog fauna (Lannoo 2005).

Gravel mining that removes stream substrates puts all life history stages of foothill yellow-legged frogs at risk of direct mortality if such mining occurs at occupied sites (Olson and Davis 2009).

The tailings of abandoned mines and settling ponds often have contaminants such as mercury that could be harmful to frogs (Olson and Davis 2009). Hothem (2007) discussed harmful mercury levels found in foothill yellow-legged frogs in Harley Gulch, within the Cache Creek watershed in Lake County, California, an area with abundant geologic sources of mercury and a long history of mercury mining and contamination. Hothem (2007) documented that mercury concentrations of 100% of the 13 foothill yellow-legged frogs collected in 1997-1998 from Harley Gulch exceeded the EPA mercury criterion (0.3 µg/g) for issuance of health advisories for human fish consumption and 100% also exceeded the methylmercury criterion for the protection of piscivorous wildlife; and that 31% exceeded FDA criterion (1.0 µg/g) for regulation of commercial fish.

Mining activities likely contributed to the extirpation of the yellow-legged frog population from Baja (Welsh 1988).

Off-road Vehicles

Damage to montane stream habitat from off-road vehicles is credited as a partial cause of the extirpation of the foothill yellow-legged frog from some southern California coastal streams (Sweet 1983). Off-road vehicle activity also likely eliminated a frog population from Corral Hollow in San Joaquin County (Jones & Stokes 2000). M.R. Jennings documented motorcycle use in riparian zones that crushed juvenile and adult foothill yellow-legged frogs (SNEP 1996).

Pollution

In the Sierra Nevada foothills of California, air-borne pesticides that move east on the prevailing winds blowing across the highly agriculturalized Central Valley are likely to be the primary threat to foothill yellow-legged frogs (LeNoir et al. 1999; Sparling et al. 2001; Hayes et al. 2002b; Fellers 2005; Sparling and Fellers 2007; Sparling and Fellers 2008). Pesticide drift from the Central Valley to the Sierra Nevada, and high pesticide levels in the bodies of Sierra Nevada amphibians, have been well documented in California by Davidson et al. (2002). They found a strong positive association between declines of both California red-legged frogs and foothill yellow-legged frogs in areas downwind of agricultural land use. The populations of foothill yellow-legged frogs in greatest decline are all downwind of highly impacted (mostly agriculturalized) areas, while the largest, most robust frog populations are along the Pacific coast (Fellers 2005).

Davidson et al. (2002) found evidence that airborne agrochemicals have played a significant role in the decline of the foothill yellow-legged frog. Davidson (2004)

examined the association between the spatial patterns of declines for five California amphibian species and historical patterns of pesticide use in California from 1974 to 1991, and found that historical pesticide use was a strong, significant variable in population declines for the foothill yellow-legged frog, especially so for organophosphates and carbamates. In particular, Davidson et al. (2002) found that sublethal exposure to the pesticide carbaryl likely inhibits the innate immune defense of foothill yellow-legged frogs and increases susceptibility to disease. Kerby and Sih (2015) conducted three separate laboratory studies examining the toxicity of the insecticide carbaryl on foothill yellow-legged frogs and found that *R. boylei* are more susceptible to pesticide exposure than Pacific tree frogs (*Pseudacris regilla*), and exposure to carbaryl reduced the ability of *R. boylei* to compete with tree frogs. Kerby and Sih (2015) also showed a remarkable increase in mortality (50%) for *R. boylei* exposed to carbaryl with an invasive crayfish (*Pacifastacus leniusculus*) predator present. Buck et al. (2015) demonstrated that exposure of larval and metamorphic amphibians to ecologically relevant concentrations of pesticide mixtures (chlorpyrifos, carbaryl, permethrin, and endosulfan) or herbicides (glyphosate, acetochlor, atrazine, and 2,4-D) altered post-metamorphic susceptibility to Bd load in Pacific treefrogs, spring peepers and western toads.

Fellers et al. (2007) exposed tadpoles for long periods of time in a laboratory to environmentally realistic concentrations of pesticides still in use. They concluded these pesticides are at sufficient concentration levels in the Sierra Nevada to cause a significant decrease in survival rates. Sparling and Fellers (2007) found that environmental concentrations of the pesticides chlorpyrifos, malathion and diazinon and their oxons can be harmful to populations of the foothill yellow-legged frog. Compounds from the breakdown of chlorpyrifos, malathion, and diazinon were found to be 10 to 100 times more toxic than the parent compounds (Sparling and Fellers 2007). Chlorpyrifos was three times more toxic and Endosulfan was 40 times more toxic to foothill yellow-legged frogs. Sparling and Fellers (2009) established the chronic toxicity of chlorpyrifos and endosulfan, two of the insecticides most commonly used in the Central Valley and found in the mountains, which likely contributes to observed declines in the frog. Kerby (2007) examined the sublethal effects of four pesticides on foothill yellow-legged frogs and found significant alteration of behavior and development.

Hayes et al. (2002) found hermaphroditism and other deformities in leopard frogs (*Rana pipiens*) exposed to commonly occurring levels of the widely used herbicide atrazine, both in the laboratory, and in the field. Hayes et al. (2006) found mixtures of pesticides to have much greater effects on frogs than single pesticides, and suggested that studies examining single pesticides may underestimate pesticide impacts on amphibians. Colborn and Clement (1992) attributed foothill yellow-legged frog population declines to endocrine mimicking chemicals that entered the ecosystem through pesticides and fungicides. Foothill yellow-legged frogs are far more susceptible to pesticides than Pacific chorus frogs (Fellers and Kleeman 2009).

Agricultural fertilizers have been linked to amphibian deaths, including in a study showing that several frog, toad, and other amphibian species in Oregon can be highly susceptible to fairly low levels of nitrate and nitrite exposure, especially at more vulnerable larval stages (Marco et al. 1999; Marco and Blaustein 1999). Marco et al. (1999) found that moderate exposure to nitrates and nitrites resulted in reduced feeding activity, disequilibrium, physical abnormalities, paralysis, and even death among some

tadpoles and young frogs. Levels of nitrite considered safe for human drinking water killed over half of Oregon spotted frog (*Rana pretiosa*) tadpoles after 15 days of exposure. Nitrates are of low toxicity but can cause health problems when reduced to nitrites. Nitrite levels can become high in specific areas such as shore sites with high contents of organic matter and can be concentrated due to waste from livestock. Nitrate can be reduced to nitrite in the gastrointestinal tract of amphibians, especially in younger animals (Marco et al. 1999; Marco and Blaustein 1999). Additionally, nitrate deposition from air pollution can greatly alter lake ecosystems, and may shift the normal ecological balance in a manner that increases the ability for disease to take hold in amphibians (V. Vredenburg, pers. comm., 2000).

Ashton et al. (1997) mentioned the potential for spills of toxic materials into streams along roads along the Trinity River in northern California. Bury (1972) found that spilled diesel fuel had negative impacts on foothill yellow-legged frog tadpoles and partially transformed individuals but apparently little impact on adults.

Mercury contamination is another threat to the frog. Hothem et al. (2010) found mercury concentrations in the foothill yellow-legged frog that were high enough to pose a potential hazard to human or wildlife consumption, with the total Hg concentration exceeding the FDA criterion (1.0 µg/g) for regulation of commercial fish in at least one sample at 24 percent of the yellow-legged frog sites, with 13 of the sites (62 percent) exceeding the EPA Hg criterion (0.3 µg/g) for issuance of health advisories for fish consumption. Research shows that mercury likely adversely affects amphibian development and can decrease survival through metamorphosis (Unrine et al. 2004). Other effects can include impaired reproduction, growth inhibition, behavioral modification, and various sublethal effects (Zillioux et al. 1993).

Recreation

There are potential threats to foothill yellow-legged frogs related to recreation (Olson and Davis 2009). Jet boats create waves that could potentially result in dislodgement and loss of egg masses, stranding of tadpoles, disruption of adult basking behavior, and erosion of shorelines (Borisenko and Hayes 1999).

Vehicles driven along stream gravel bars and recreationists fishing, swimming, walking or camping along shores likely adversely affects frogs, including disruption of frog basking opportunities (Borisenko and Hayes 1999). The Marin Municipal Water District documented that people and dogs have been known to squash the eggs of foothill yellow-legged frogs in Little Carson Creek in Marin County (Prado 2005). A concern in the Little Yosemite area of Alameda Creek in Alameda County is intensive disturbance to yellow-legged frog breeding habitat by humans and dogs (S. Kupferberg, J. Miller, pers. observ.).

Roads and Urbanization

Roads and urbanization are logical potential threats to this frog (Davis and Olson 2009). The human population continues to increase within its range and this results in continued expansion of urban and agricultural areas and construction of new roads. Road construction crossing streams likely adversely affects frogs due to sedimentation during road building, maintenance or failures. As explained above, sediments can

embed stream substrates and remove interstitial spaces used by these frogs. The use of culverts that do not easily pass frogs also impacts population connectivity. Lind (2006) found that foothill yellow-legged frog presence was associated with less urban development nearby, using data from both Oregon and California.

INADEQUACY OF EXISTING REGULATORY MECHANISMS

Federal Regulatory Mechanisms

Existing federal regulatory mechanisms that have the potential to provide some form of protection for the foothill yellow-legged frog include occurrence on federally protected land, consideration under the National Environmental Policy Act or the Clean Water Act, and coverage under Habitat Conservation Plans.

Occurrence on National Forests/BLM Lands

Populations of foothill yellow-legged frogs occur on national forest lands and Bureau of Land Management (BLM) lands in California. The foothill yellow-legged frog is listed on the USDA Forest Service Region 5 (California) Sensitive Species List. However, this designation as a “sensitive species” offers little protection for individual frogs, frog populations or frog habitat. The designation merely requires that the impacts to the species be considered, but does not prevent agency actions, such as logging, road building, cattle grazing or mining, that could harm the species or its habitat. Sensitive species cannot be impacted without an analysis of significance of adverse effects on the populations, their habitat, and on the viability of the species as a whole. All Forest Service planned, funded, executed, or permitted programs and activities are reviewed under NEPA for possible effects on sensitive species, through a Biological Assessment and Evaluation. Yet the Forest Service can conclude in a Biological Evaluation that even though individual frogs or frog populations will be harmed or destroyed by an action, it can still carry out this action.

The Forest Service adopted the Sierra Nevada Forest Plan Amendment in 2001 after more than a decade of scientific study, to direct the management of 11.5 million acres of California's national forest lands in the Sierra. The Sierra Nevada Forest Plan Amendment represented a shift in Forest Service management to ecosystem management principles. However, as it has been implemented, the Sierra Nevada Forest Plan arguably has not provided adequate protection for the foothill yellow-legged frog from water withdrawals, river flow regulation by dams, hydrologic alteration of wet meadows by livestock grazing, and sedimentation from forest roads, which are all permitted or agency-directed actions on national forest lands (C. Frissell, pers. comm., 2015).

The Sierra Nevada Forest Plan Amendment committed the Forest Service to completing a Conservation Assessment for the foothill yellow-legged frog in cooperation with other federal agencies, state agencies, universities, and research scientists (USDA Forest Service 2001). The Conservation Assessment was published in 2016 (Hayes et al. 2016). Conservation Assessments provide only management recommendations, not mandated habitat protections. The Conservation Assessment is envisioned as the first of a three-phase process that also includes a Conservation Strategy and a Conservation Agreement. However, this process is moving far too slowly to provide protection for

foothill yellow-legged frogs. The Conservation Assessment alone took more than a decade to produce. The Sierra Nevada Plan's primary emphasis is on terrestrial species, but it also contains an Aquatic Conservation Strategy focused on reducing some threats to amphibians, including the foothill yellow-legged frog. Some of these measures include changes to livestock grazing and exotic fish stocking practices. Yet at the same time, the plan contains proposed management activities (such as fire and fuels management) that may increase risk of habitat degradation for yellow-legged frogs. For example, extensive fuels treatments (e.g., prescribed burning and mechanical thinning of trees) are proposed at lower elevations because these areas contain large wildland/urban interface zones. Some of these treatments may occur within riparian areas, resulting in unknown effects on the foothill yellow-legged frog.

In addition, the Sierra Nevada Forest Plan Amendment has been under attack since its adoption, with ongoing efforts by legislators and industry to increase the amount of logging allowed, limit protections for forests, water quality and wildlife, and to weaken forest monitoring requirements by reducing the management indicator species lists that are tracked across Sierra Nevada national forests.

In 1994 the U.S. Forest Service and Bureau of Land Management adopted the Northwest Forest Plan (USDA and USDI 1994), establishing different land allocations and standards and guidelines intended to conserve and restore both terrestrial and aquatic ecosystems. The Northwest Forest Plan established "riparian reserves" and includes an Aquatic Conservation Strategy that is generally protective of habitat for the foothill yellow-legged frog, in that it sets buffers on logging for year round and intermittent streams, among other measures.

However, as it has been implemented, the Northwest Forest Plan arguably has not provided adequate protection for the foothill yellow-legged frog from water withdrawals, river flow regulation by dams, hydrologic alteration of wet meadows by livestock grazing, and sedimentation from forest roads, which are all permitted or agency-directed actions on national forest lands (C. Frissell, pers. comm., 2015). Frissell (2013, 2014) discussed ongoing efforts by the U.S. Forest Service, BLM and political leaders to alter the Northwest Forest Plan by reducing the area of Riparian Reserves, while also increasing the basis for commercial logging from near-stream and potentially unstable lands. Frissell (2013, 2014) and Heiken (2013) evaluated the potential environmental consequences of altering Riparian Reserve protections in the Northwest Forest Plan to allow more systematic and aggressive logging within Riparian Reserves (mostly as commercial thinning), including alteration of thermal regimes and increased summer stream temperatures, increased erosion and sediment delivery to streams, and diminished capacity of riparian forests to filter nutrients loads that are a threat to water quality. The Forest Service and BLM are moving forward with attempts to revise the Northwest Forest Plan to reduce stream buffers and weaken the Aquatic Conservation Strategy (USBLM 2015b).

The three National Parks (Yosemite, Kings Canyon, and Sequoia National Parks) that encompass a small portion of the historical range of the foothill yellow-legged frog all have guiding principles, management goals and management plans that are beneficial for aquatic ecosystems, but the species is already extirpated from all three of these parks.

Even on federal lands that are protected for ecological values, foothill yellow-legged frogs are not protected from threats such as drifting pesticides or impacts from nonnative predators.

National Environmental Policy Act

The National Environmental Policy Act of 1969 (NEPA) (42 U.S.C.4321-4370a) requires federal agencies to consider the environmental impacts of their actions. The NEPA process requires these agencies to describe a proposed action, consider alternatives, identify and disclose potential environmental impacts of each alternative, and involve the public in the decision-making process. Most actions taken by the federal agencies such as the National Park Service, U. S. Forest Service, and the Bureau of Land Management that could affect the foothill yellow-legged frog are subject to the NEPA process. NEPA does not, however, prohibit these agencies from choosing alternatives that will negatively affect individual frogs, populations of foothill yellow-legged frogs, or potential foothill yellow-legged frog habitat. De facto evidence of NEPA's inability to protect the foothill yellow-legged frog is that the species has declined precipitously in spite of the existence of NEPA for more than 45 years.

Clean Water Act

Under Section 404 of the Clean Water Act, 33 U.S.C. §§ 1251 et seq. ("CWA"), discharge of pollutants, including dredged or fill material, into "Waters of the U.S." is prohibited absent a permit from the U.S. Army Corps of Engineers. Theoretically the CWA should provide some protection for stream and wetland habitats used by foothill yellow-legged frogs. However, the implementation of the CWA regulatory scheme and the Section 404 program in particular have fallen far short of Congress's intent to protect wetlands and water quality. A National Research Council report entitled "Compensating for Wetland Losses Under the Clean Water Act" concluded that the goal of no net loss of wetlands has not been achieved through the Army Corps regulatory program, and that applicants often do not follow through on promised mitigation packages (National Research Council 2001). These failures of the Army Corps regulatory scheme are due in part because the Corps' implementation of the individual permitting process has allowed too much development while requiring too little avoidance and mitigation. Also, in permitting projects, the Army Corps often takes a very limited view of a project, looking only at impacts in the project footprint. The CWA has been and will continue to be inadequate to ensure the continued survival of the foothill yellow-legged frog.

Habitat Conservation Plans

There are only four Habitat Conservation Plans in California within the range of the species that include the foothill yellow-legged frog as a covered species: the San Joaquin County Multi-Species Habitat Conservation and Open Space Plan; East Contra Costa County HCP/NCCP; Humboldt Redwood Company (formerly Pacific Lumber, Headwaters) HCP; and Santa Clara Valley HCP/NCCP (USFWS 2015).

Two of these HCPs – the San Joaquin County HCP and the East Contra Costa County HCP - no longer have extant foothill yellow-legged frog populations within their plan areas. The San Joaquin County HCP referred to four historic records of *R. boylei* from San Joaquin County, with the most recent observation in the Corral Hollow area in 1977, and defined the status of the species in the county as unknown (San Joaquin County

2000). Though the HCP set out some incidental take minimization measures for foothill yellow-legged frogs, the species is likely already extirpated from lower Corral Hollow Creek, western San Joaquin County and the HCP plan area. The East Contra Costa County HCP noted that there are no recent documented occurrences of foothill yellow-legged frogs within the eastern Contra Costa Habitat Conservation Plan inventory area, covering major portions of eastern Contra Costa County (CCC 2006).

The Humboldt Redwood Company HCP (HRCHCP) noted that foothill yellow-legged frogs were “commonly observed” in the HCP area along the Eel River and Van Duzen River; also reported from the Yager and Bear-Mattole watershed areas, and were “suspected to occur” in suitable habitat in the Humboldt watershed area (HRC 1999). While the HRCHCP’s amphibian and reptile conservation plan contains vague promises of retention of habitat diversity and mix of forest types post-logging (HRC 1999), no concrete measures are specified to protect or enhance habitat specifically for foothill yellow-legged frogs, and the plan allows levels of timber harvest and road building likely to be detrimental to the species. The HRCHCP allows logging of a substantial portion of remaining late-successional forest on their lands; of an estimated 26,147 acres of old-growth (12% of their total lands), 57% is available for harvest (USDI et al. 1999).

The Santa Clara Valley HCP (SCVHCP) noted that foothill yellow-legged frogs had essentially disappeared from the farmed and urbanized lowland areas of Santa Clara County, as well as many of the perennial streams below major reservoirs; but still occurred in Santa Clara County in the upper reaches of Coyote Creek, nearly all of the streams in the Pajaro River watershed, Penitencia Creek, and in the Santa Cruz Mountains west of Gilroy (ICF 2012).

The SCVHCP anticipated potential direct impacts from projects in stream channels that could result in removal of cobblestone substrate or riparian vegetation, increase in erosion and sediment discharge, creation of pooling habitat where higher risk of predation exists for frogs, and dewatering of breeding locations. The SCVHCP anticipated permanent impacts in up to 5.7 stream miles and temporary impacts in up to 2.0 miles of modeled primary and secondary habitat for foothill yellow-legged frogs.

The SCVHCP included no conservation efforts within the study area that directly target the recovery of this species, but noted that stream restoration projects that return creeks and streams to natural flow regimes would benefit the species. The SCVHCP proposed to acquire a minimum of 80 miles of primary and secondary modeled habitat for the foothill yellow-legged frog for the SCVHCP’s Reserve System; but only proposes to protect 32-44% of the 690 miles of foothill yellow-legged frog modeled primary and secondary habitat within the study area.

The HCP Reserve System was expected to protect only 4 known occurrences of the foothill yellow-legged frog: 3 on Llagas Creek above Chesbro Reservoir and 1 on San Felipe Creek above Anderson Reservoir. The SCVHCP notes that the species could possibly occur on the Reserve System on Upper Penitencia Creek, Uvas Creek below Uvas Reservoir, and Little Arthur Creek, but the species presence had not been documented. The SCVHCP prioritizes acquisition of streams for the Reserve System by: sites with documented records of breeding foothill yellow-legged frog; sites with known occurrences, but no documented breeding; and sites without known occurrences of foothill yellow-legged frogs but with western pond turtle habitat and known occurrences of other covered amphibian species. The SCVHCP proposed to “restore” from 1.0 to

10.4 miles of streams, with a goal of supporting breeding of yellow-legged frogs, by adding sediment to stream courses so that sand bars will form to create egg laying substrate, or adding large rocks to the stream course for the same purpose. Management will include selectively applying herbicides or other treatments to control nonnative invasive vegetation along creek corridors that might inhibit sediment movement and restrict the creation of egg laying habitat. Proposed management techniques and tools include stream channel rehabilitation, planting native understory and overstory riparian vegetation, adding rocky substrate to the stream channel, and translocation to help establish new populations.

Very few extant foothill yellow-legged populations will gain any protection from the two HCPs which overlap with the current range of the species.

In addition, coverage by an HCP is not a guarantee of protection for a species. Kareiva et al. (1999) thoroughly discussed the failure of most HCPs to protect and recover listed species that are covered under HCP agreements. Kareiva et al. (1999), in a nationwide study of HCPs by the National Center for Ecological Analysis & Synthesis and the American Institute of Biological Sciences, found that most HCPs contributed to habitat losses for the targeted species, failed to meet recovery goals, and suffered from poor planning and plan evaluation. Kareiva et al. (1999) documented that nearly 30% of HCPs “take” 100% of the focal species’ populations or habitat in the permit area; about 50% of HCPs allowed 50% or more of the species’ populations or habitat in the plan area to be taken; 43% of the time, HCPs failed to provide sufficient mitigation measures; 23% of the time, species and their habitats were taken before mitigation measures were implemented and found effective (most HCPs failed to reduce allowed take levels or use other more conservative approaches in the face of inadequate information or uncertainties); 33% of HCPs failed to secure up-front funding to ensure that mitigation actually occurs; and 81% of HCPs studied had irreversible impacts. Not surprisingly, HCPs that failed to adequately conserve species also tended to lack rigorous impact assessments and planning. Kareiva et al. (1999) found that: 75% of the time, impacts to species were not adequately studied by HCPs; 42% to 49% of the time, HCPs failed to quantify how much of a species’ habitat and population, respectively, would be taken; most HCPs used low quality data to evaluate their mitigation measures; and 25% of the time, sufficient information did not exist to determine how HCPs would affect the species’ viability.

Several other studies have documented a variety of shortcomings regarding the scientific foundation and conservation promises of HCPs (e.g. Hood 1998; Smallwood et al. 1998; Bowler 2000; Smallwood 2000; Harding et al. 2001; Wilhere 2002). Rahn et al. (2006) found that the conservation benefits of multispecies HCPs (MSHCPs) to individual covered species may be overestimated. Rahn et al. (2006) reviewed the species selected for coverage in 22 MSHCPs from USFWS Region 1, and found that conservation measures were often not clearly defined, and that the presence of the species in the planning area was not even confirmed for 41% of covered species. Owley (2015) used a case study approach to illustrate the concerns associated with whether federal and local agencies are able to locate, understand, track and adequately enforce HCP mitigation requirements and conservation easements, focusing on four HCPs in California (San Bruno Mountain HCP in San Mateo County, approved in 1983; Lytle Creek Turnout Low-Effect HCP in San Bernardino County, approved in 2009; Cushenbury Sand and Gravel HCP in San Bernardino County, approved in 1996; and

Wildcat Line Property HCP in Monterey County, approved in 2001). Owley (2015) found that the government entities involved in these HCPs struggled to locate and understand the permits themselves, let alone the details of the compensatory mitigation projects, and that county offices charged with recording property restrictions on conservation easements often had inadequate records of land use restrictions.

State Regulatory Mechanisms

The state of California lists the foothill yellow-legged frog as a “Species of Special Concern.” This is an administrative designation and carries no formal legal status.

Natural Community Conservation Plans

Of the 9 approved Natural Community Conservation Plans (NCCP) in California, only 2 are within the range of the foothill yellow-legged frog: the East Contra Costa County NCCP and Santa Clara Valley NCCP (CDFW 2015). These plans are joint NCCP/HCPs, and the limitations of these plans for protecting foothill yellow-legged frogs was discussed above. Of NCCPs in the planning phase, only one, the Butte Regional Conservation Plan, lists the foothill yellow-legged frog as a proposed covered species (CDFW 2015).

California Environmental Quality Act

The environmental review process under the California Environmental Quality Act (“CEQA”, California Public Resources Code §§ 21000-21177) requires state agencies, local governments and special districts to evaluate and disclose impacts from “projects” in the state. CEQA declares that it is the policy of the state to prevent “the elimination of fish or wildlife species due to man’s activities, ensure that fish and wildlife populations do not drop below self-perpetuating levels, and preserve for future generations representations of all plant and animal communities” (California Public Resources Code, section 21001(c)). The CEQA process is triggered when discretionary activities of state agencies may have a significant effect on the environment. When the CEQA process is triggered, it requires full disclosure of the potential environmental impacts of proposed projects. The operative document for major projects is usually the Environmental Impact Report.

Under CEQA, Species of Special Concern must be considered during the environmental review process, with an analysis of the project impacts on the species, only if they meet the criteria of sensitivity under Section 15380 of the CEQA Guidelines. However, project impacts to foothill yellow-legged frogs are often not analyzed because project proponents are able to claim insignificant impacts to non-listed species if the project does not have population-level or regional effects or impacts a small proportion of the species’ range.

Theoretically, besides ensuring environmental protection through procedural and informational means, CEQA also has substantive mandates for environmental protection. The most important of these is the provision requiring public agencies to deny approval of a project with significant adverse effects when feasible alternatives or feasible mitigation measures can substantially lessen such effects. In practice, however, this substantive mandate is rarely implemented, particularly with regard to instream

projects, water diversions, mining permits, grazing permits and projects causing pollution and sedimentation that have impacted and continue to impact habitat for foothill yellow-legged frogs. If significant impacts remain after all mitigation measures and alternatives deemed feasible by a lead agency have been adopted, a lead agency is allowed under CEQA to approve a project despite environmental impacts if it finds that social or economic factors outweigh the environmental costs. It is important to note that CEQA is not, nor was it ever intended to be, a habitat protection mechanism.

Regional and Local Government Plans

Madera County adopted a Yellow-legged Frog Conservation Program in 1997, which contained some measures to protect suitable primary frog habitat “from the direct significant impact of human activity,” however the foothill yellow-legged frog may already be extirpated from Madera County.

Summary: The perilous status of the foothill yellow-legged frog reflects the overall failure or inability of existing federal and state regulatory mechanisms to protect foothill yellow-legged frog habitat and provide for the conservation of the species. Neither CEQA nor any other state or local regulatory mechanism provide protection from factors adversely impacting foothill yellow-legged frogs such as invasive species, pollutants and pesticides, disease or climate change. Without state listing, reintroduction of the species at unoccupied historic sites and implementation of confirmed frog habitat enhancement methods (e.g. Kupferberg 1996a; Lind et al. 1996; Lind and Shaffer 2005; Yarnell 2005) are unlikely to be utilized.

RECOMMENDED MANAGEMENT AND RECOVERY ACTIONS

Recommended management actions for foothill yellow legged frogs will vary depending on the type of river system where a given extant population remains, either with flows regulated by dams or in free-flowing systems that may be subject to other forms of human perturbation (such as illegal diversion of flows in summer for *Cannabis* cultivation or excessive sedimentation and hillslope erosion due to road building and other types of land use in the upland portions of the watersheds).

Require frog-friendly flow regimes: In rivers with dams, a-seasonal flow fluctuation which could cause stranding and scouring of egg masses and tadpoles should be prohibited. Limit pulsed flows from dams during or after oviposition and during tadpole development. For rivers with frog populations below dams, flow regimes should be developed that mimic natural seasonal flows to which the various life stages of *R. boylei* are adapted to. Maintaining thermal regimes below dams that are conducive to larval survival and rapid development will also be important.

Stream channel habitat restoration: Where upstream dam operations have artificially cooled downstream river reaches, suppressed flood disturbance, limited sediment supply, and allowed encroachment of woody riparian vegetation into the active channel (thus reducing frog opportunities for thermoregulatory behaviour), create thermal habitat heterogeneity by restoring gently sloping and sun-lit gravel bars used for breeding and by enhancing edgewater habitats.

Eradication of invasive predators: Conduct active eradication and management efforts to decrease the abundance of bullfrogs (removing egg masses, netting tadpoles, hunting adult frogs, etc.). Develop eradication or control programs for non-native fish and crayfish which are predators of frogs, tadpoles, and egg masses. In managed rivers, manipulate stream flows to negatively affect non-native taxa not adapted to a winter-flood/summer drought flow regime.

Mitigate impacts of *Cannabis* cultivation: Direct some of the money that will be gained from new *Cannabis* taxes to rehabilitation of *R. boylei* streams, including for law enforcement to end illegal dewatering of creeks, and to conduct clean-up of spilled diesel fuel, fertilizers and rodenticides.

Prohibit habitat damage: Ensure that state regulations for timber harvest within watersheds with yellow-legged frogs adequately prevent siltation of streambeds or increase of water temperatures to lethal levels. Prohibit instream gravel mining or dredging in stream reaches with yellow-legged frogs. Ensure that all state-managed off-road vehicle areas are not negatively impacting yellow-legged frogs or their habitat.

Restrict pesticides: Determine where and which pesticide uses should be restricted to prevent harm to yellow-legged frogs.

Reintroduction: Explore reintroduction of foothill yellow-legged frogs into stream systems with appropriate habitat within the historical range of the species. Start with streams in the National Parks once the stressors have been removed, such as post-eradication of bullfrogs in Yosemite National Park.

Curate locality data: The California Department of Fish and Wildlife should take responsibility for, or find a curator to maintain a repository of, all the foothill yellow-legged frog survey data collected by agencies, utilities and researchers, and submitted to the CNDDB.

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State of California
Natural Resources Agency
Department of Fish and Wildlife

**REPORT TO THE FISH AND GAME COMMISSION
CALIFORNIA ENDANGERED SPECIES ACT STATUS REVIEW OF
SOUTHERN CALIFORNIA STEELHEAD (ONCORHYNCHUS MYKISS)**

January 2024



Southern California Steelhead Rainbow Trout, CDFW photo

Prepared by
California Department of Fish and Wildlife



Attachment C: Report to the Fish and Game Commission California Endangered Species Act Status Review of Southern California Steelhead (Oncorhynchus mykiss) January 2024

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TABLE OF CONTENTS

REPORT TO THE FISH AND GAME COMMISSION CALIFORNIA ENDANGERED SPECIES ACT STATUS
REVIEW OF SOUTHERN CALIFORNIA STEELHEAD (ONCORHYNCHUS MYKISS) i

TABLE OF CONTENTS.....	2
LIST OF FIGURES.....	4
LIST OF TABLES.....	5
LIST OF APPENDICES	7
LIST OF ABBREVIATIONS, ACRONYMS, AND TERMS.....	7
EXECUTIVE SUMMARY	9
1. INTRODUCTION.....	11
1.1 <i>Petition History</i>	11
1.2 <i>Status Review Overview</i>	12
1.3 <i>Federal Endangered Species Act Listing History</i>	13
2. BIOLOGY AND ECOLOGY	14
2.1 <i>Species Description</i>	14
2.2 <i>Taxonomy and Systematics</i>	17
2.3 <i>Range and Distribution</i>	18
2.4 <i>Life History</i>	19
2.5 <i>Genetics and Genomics</i>	23
3. ASSESSMENT OF PROPOSED CESA LISTING UNIT	33
3.1 <i>DPS and ESU Criteria</i>	34
3.2 <i>Southern SH/RT Evaluation under the Joint DPS Policy</i>	36
4. POPULATION TRENDS AND ABUNDANCE	39
4.1 <i>Structure and Function of Viable Salmonid Populations</i>	39
4.2 <i>Sources of Information</i>	40
4.3 <i>Historical and Current Distribution</i>	41
4.4 <i>Abundance and Trends</i>	61
4.5 <i>Productivity</i>	75
4.6 <i>Population Spatial Structure</i>	78

4.7 Diversity.....	80
4.8 Conclusions.....	82
5. HABITAT THAT MAY BE ESSENTIAL TO THE CONTINUED EXISTENCE OF SOUTHERN SH/RT	85
5.1 Migration	85
5.2 Spawning.....	85
5.3 Instream Residency	86
5.4 Egg and Larval Development and Fry Emergence	87
5.5 Rearing and Emigration	88
5.6 Ocean Growth	91
6. FACTORS AFFECTING THE ABILITY TO SURVIVE AND REPRODUCE.....	92
6.1 Changes in Ocean Conditions.....	92
6.2 Effects of Climate Change	93
6.3 Disease	102
6.4 Hatcheries	102
6.5 Predation.....	104
6.6 Competition.....	105
6.7 Genetic Diversity	106
6.8 Habitat Conditions	108
6.9 Fishing and Illegal Harvest.....	118
7. INFLUENCE OF EXISTING MANAGEMENT EFFORTS	119
7.1 Federal and State Laws and Regulations.....	119
7.2 Recovery Plans and Regional Management Plans.....	128
7.3 Habitat Restoration and Watershed Management.....	131
7.4 Commercial and Recreational Fishing.....	134
7.5 Research and Monitoring Programs	134
8. SUMMARY OF LISTING FACTORS	136
8.1 Present or Threatened Modification or Destruction of Habitat.....	136
8.2 Overexploitation.....	137
8.3 Predation.....	138

8.4 Competition.....	138
8.5 Disease	139
8.6 Other Natural Occurrences or Human-related Activities.....	139
9. SUMMARY OF KEY FINDINGS.....	139
10. RECOMMENDATION FOR THE COMMISSION	141
11. PROTECTION AFFORDED BY LISTING	141
12. MANAGEMENT RECOMMENDATIONS AND RECOVERY MEASURES	142
ACKNOWLEDGEMENTS.....	145
LITERATURE CITED	145
<i>Literature</i>	145
<i>Personal Communication</i>	187
APPENDIX A: Annual <i>O. mykiss</i> observations and data sources for three extant populations in the Conception Coast BPG.	188
APPENDIX B: Annual adult steelhead observations and data sources for three extant populations in the Conception Coast BPG.	189
APPENDIX C: Annual <i>O. mykiss</i> observations and data sources for four extant populations in the Monte Arido Highlands BPG.	190
APPENDIX D: Annual adult steelhead observations and data sources for four extant populations in the Monte Arido Highlands BPG.	190
APPENDIX E: Comments from tribes and affected and interested parties on the petitioned action	191
APPENDIX F: Peer Review summary	Error! Bookmark not defined.

LIST OF FIGURES

FIGURE 1. Consensus relationships of <i>Oncorhynchus</i> species from morphological, allozyme, ribosomal RNA, mitochondrial DNA, and short interspersed repetitive elements data across multiple studies.....	18
FIGURE 2. Majority-rule consensus tree, with genetic data bootstrapped 1,000 times, showing chord distances and neighbor-joining trees for 62 coastal California steelhead populations.....	25
FIGURE 3. Unrooted neighbor-joining chord distance tree of 84 coastal <i>O. mykiss</i> populations in California	27

FIGURE 4. Schematic of indirect genetic control of migratory behavior. Genetic variation and the environment influence physiology, which then impacts migratory behavior	29
FIGURE 5. Unrooted neighbor-joining dendrogram showing chord distances between 24 sampled naturally spawning populations both above and below barriers.....	30
FIGURE 6. Map of the known and suspected current and historic distribution of Southern SH/RT. BPGs represented are the Monte Arido Highlands, Conception Coast, Santa Monica Mountains, Mojave Rim, and Santa Catalina Gulf Coast	42
FIGURE 7. Map of the Monte Arido Highlands BPG	43
FIGURE 8. Map of the Conception Coast BPG	51
FIGURE 9. Map of the Santa Monica Mountains BPG	53
FIGURE 10. Map of the Map of the Mojave Rim BPG.....	55
FIGURE 11. Map of the Santa Catalina Gulf Coast BPG	58
FIGURE 12. Adult steelhead and <i>O. mykiss</i> abundances for the Monte Arido Highlands BPG	63
FIGURE 13. Adult steelhead and <i>O. mykiss</i> abundances for the Santa Monica Mountains BPG .65	
FIGURE 14a. Ln-Cohort Replacement Rates for <i>O. mykiss</i> populations, A) Santa Ynez River, B) Ventura River, and C) Santa Clara River.....	77
FIGURE 14b. Ln-Cohort Replacement Rates for <i>O. mykiss</i> populations, D) Topanga Creek, and E) Malibu Creek.....	78
FIGURE 15. Temperature trend and departure from average graphs for California.....	94
FIGURE 16. The distribution and progression of drought conditions in California from 2011 to 2016, depicting the level of drought at the beginning of each Water Year (October 1)	96
FIGURE 17. Example southern California stream (Arroyo Hondo Creek, Santa Barbara County), showing seasonal desiccation across 60% of its study area wetted length during February-October 2015	97

LIST OF TABLES

TABLE 1. Common nomenclature for <i>Oncorhynchus mykiss</i>	15
TABLE 2a. Minimum and maximum adult abundance for the Santa Ynez River over three-time frames: 1995 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year)	67
TABLE 2b. Minimum and maximum adult abundance for the Ventura River over three-time frames: 2006 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year)	67

TABLE 2c. Minimum and maximum adult abundance for the Santa Clara River over three-time frames: 1994 to 2018 (long-term), 2007 to 2018 (12-year), and 2014 to 2018 (5-year)	68
TABLE 3. Long-term, medium-term, and short-term geometric mean abundance of adult steelhead in the Monte Arido Highlands BPG	68
TABLE 4a. Minimum and maximum <i>O. mykiss</i> abundance for the Santa Ynez River over three-time frames: 2001 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year).....	68
TABLE 4b. Minimum and maximum <i>O. mykiss</i> abundance for the Ventura River over three-time frames: 2005 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year)	69
TABLE 4c. Minimum and maximum <i>O. mykiss</i> abundance for the Santa Clara River over three-time frames: 1994 to 2014 (long-term), 2003 to 2014 (12-year), and 2010 to 2014 (5-year).....	69
TABLE 5. Long-term, medium-term, and short-term geometric mean abundance of <i>O. mykiss</i> in the Monte Arido Highlands BPG	69
TABLE 6a. Table 7a. Minimum and maximum adult abundance for Arroyo Sequit Creek over three-time frames: 2005 to 2018 (long-term), 2007 to 2018 (12-year), and 2014 to 2018 (5-year)	70
TABLE 6b. Minimum and maximum adult abundance for Malibu Creek over three-time frames: 2004 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year)	70
TABLE 6c. Minimum and maximum adult abundance for Topanga Creek over three-time frames: 2001 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year)	70
TABLE 7. Long-term, medium-term, and short-term geometric mean abundance of adult steelhead in the Santa Monica Mountains BPG.	71
TABLE 8a. Minimum and maximum <i>O. mykiss</i> abundance for Arroyo Sequit Creek over three-time frames: 2005 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year).....	71
TABLE 8b. Minimum and maximum <i>O. mykiss</i> abundance for Malibu Creek over three-time frames: 2004 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year)	71
TABLE 8c. Minimum and maximum <i>O. mykiss</i> abundance for Topanga Creek over three-time frames: 2001 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year)	72
TABLE 9. Long-term, medium-term, and short-term geometric mean abundance of <i>O. mykiss</i> in the Santa Monica Mountains BPG. Data used are sum of the average number of <i>O. mykiss</i> observed per month	72
TABLE 10. Trends in adult abundance using slope of ln-transformed time series counts for three Monte Arido Highland BPG populations.	73

TABLE 11. Trends in <i>O. mykiss</i> abundance using slope of ln-transformed time series counts for three Monte Arido Highland BPG populations.....	74
TABLE 12. Trends in adult abundance using slope of ln-transformed time series counts for the Santa Monica Mountains BPG populations	74
TABLE 13. Trends in <i>O. mykiss</i> abundance using slope of ln-transformed time series counts for the Santa Monica Mountains BPG populations.....	75

LIST OF APPENDICES

APPENDIX A. Annual *O. mykiss* observations and data sources for three extant populations in the Conception Coast BPG.

APPENDIX B. Annual adult steelhead observations and data sources for three extant populations in the Conception Coast BPG.

APPENDIX C. Annual *O. mykiss* observations and data sources for four extant populations in the Monte Arido Highlands BPG.

APPENDIX D. Annual adult steelhead observations and data sources for four extant populations in the Monte Arido Highlands BPG.

APPENDIX E. Comments from Tribes and Affected and Interested Parties on the Petitioned Action

APPENDIX F. Peer Review Summary

LIST OF ABBREVIATIONS, ACRONYMS, AND TERMS

BEUTI – Biologically Effective Upwelling Transport Index

BPG – Biogeographic Population Group

CATEX – Categorical Exclusion

CCE – California Current Ecosystem

CESA – California Endangered Species Act

CEQA – California Environmental Quality Act

CFS – cubic feet per second

CMP – California Coastal Monitoring Program

CMWD – Casitas Municipal Water District

COMB – Cachuma Operations and Maintenance

Commission – California Fish and Game Commission

Creeks Division – City of Santa Barbara Creeks Restoration and Water Quality Improvement Division

CRR – cohort replacement rate

CUTI – Cumulative Upwelling Transport Index
CWA – Federal Clean Water Act
Department – California Department of Fish and Wildlife
DIDSON - dual-frequency identification sonar
DO – dissolved oxygen
DPS – Distinct Population Segment
DWR – California Department of Water Resources
EA – Environmental Assessment
EIR – Environmental Impact Report
EIS – Environmental Impact Statement
EPA – United States Environmental Protection Agency
ESA – Federal Endangered Species Act
ESU – Evolutionary significant unit
FERC – Federal Energy Regulatory Commission
FONSI – Finding of No Significant Impact
FRGP – Fisheries Restoration Grant Program
GSA – Groundwater sustainability agency
GSP – Groundwater sustainability plan
HCP – Habitat Conservation Plan
LWD – large woody debris
NCCP – Natural Community Conservation Plan
NEPA – National Environmental Policy Act
NGO – Non-Governmental Organization
NMFS – National Marine Fisheries Service
RCDSMM – Resource Conservation District of the Santa Monica Mountains
SCCWRP – Southern California Coastal Water Research Project
SCWRP – Southern California Wetlands Recovery Project
SGMA – Sustainable Groundwater management Act
SNP – single nucleotide polymorphism
SST – sea surface temperature
SWRCB – California State Water Resources Control Board
TMDL – Total Maximum Daily Load
USACE – United States Army Corp of Engineers
USBR – United States Bureau of Reclamation
USFWS – United States Fish and Wildlife Service
UWCD – United Water Conservation District
WSRA – Federal Wild and Scenic Rivers Act
YOY – young-of-the-year

EXECUTIVE SUMMARY

This status review of southern California steelhead (*Oncorhynchus mykiss*) (Status Review) has been prepared by the California Department of Fish and Wildlife (Department) for the California Fish and Game Commission (Commission) pursuant to the requirements of the California Endangered Species Act (CESA; Fish & G. Code, § 2050 et seq.). This Status Review is based on the best scientific information currently available to the Department regarding each of the components listed under Section 2072.3 of the Fish and Game Code and Section 670.1 of Title 14 of the California Code of Regulations. In addition, this Status Review includes a preliminary identification of habitat that may be essential to the continued existence of the species, the Department's recommendations for management activities, and other recommendations for the recovery of the species (Fish & G. Code, § 2074.6). This Status Review has been independently reviewed by scientific peers pursuant to Fish and Game Code Section 2074.6.

In this Status Review, southern California steelhead are defined as "all *O. mykiss* below manmade and natural complete barriers to anadromy, including anadromous and resident life histories, from and including the Santa Maria River (San Luis Obispo and Santa Barbara counties) to the U.S.-Mexico Border." This range encompasses five biogeographic population groups of *O. mykiss* (from north to south): Monte Arido Highlands, Conception Coast, Santa Monica Mountains, Mojave Rim, and Santa Catalina Gulf Coast. To capture the life history variability that is included in the scope of the CESA listing unit evaluated in this Status Review, "southern California steelhead rainbow trout" (Southern SH/RT) is used to describe the proposed CESA listing unit.

The Department recommends that the Commission find the petitioned action to list Southern SH/RT as an endangered species under CESA to be warranted. The Department further recommends implementation of the management recommendations and recovery measures described in this Status Review.

The scientific data available to the Department indicates a long-term declining trend of Southern SH/RT and low range-wide abundances. The decline of Southern SH/RT can be attributed to a wide variety of human activities, including, but not limited to, urbanization, agriculture, and water development. These activities have degraded range-wide aquatic habitat conditions and limited the amount of suitable and accessible spawning and rearing habitats. Dams and other impediments obstruct access to a significant portion of historical Southern SH/RT habitats in many rivers within the proposed listing area, some of which have multiple major dams on a single mainstem.

Climate change projections for Southern SH/RT range predict an intensification of typical climate patterns, such as more intense cyclic storms, droughts, and extreme heat. These projections suggest that Southern SH/RT will likely experience more frequent periods of adverse conditions and continued selection pressure against the anadromous life-history form. Impacts of the most recent prolonged period of drought from 2012 – 2017 resulted in significant reductions in all life-history forms and stages of Southern SH/RT, and few populations have rebounded as current abundance estimates remain low relative to pre-drought conditions. The ability of Southern SH/RT to persist will likely depend on the successful recruitment of migrants from resident populations in refugia habitats. However, virtually all refugia populations are currently above impassable barriers. Furthermore, many southern California watersheds do not contain upstream drought refugia. In these instances, recolonization of Southern SH/RT from source populations in other watersheds is likely the only mechanism for these populations to rebound (Boughton et al. 2022a).

1. INTRODUCTION

1.1 Petition History

On June 14, 2021, the California Fish and Game Commission (Commission) received a petition (Petition) from California Trout to list southern California steelhead (*Oncorhynchus mykiss*) as endangered pursuant to the California Endangered Species Act (CESA; Fish & G. Code, § 2050 et seq.).

On June 23, 2021, pursuant to Fish and Game Code Section 2073, the Commission referred the Petition to the California Department of Fish and Wildlife (Department) for evaluation.

On July 16, 2021, pursuant to Fish and Game Code Section 2073.3, the Commission published notice of receipt of the Petition in the California Regulatory Notice Register (Cal. Reg. Notice Register 2021, No. 29-Z, p. 921-922).

On August 18, 2021, pursuant to Fish and Game Code Section 2073.5, the Commission approved the Department's request for a 30-day extension to complete its petition evaluation report.

On October 29, 2021, the Department provided the Commission with a report, "Evaluation of the Petition from California Trout to List Southern California Steelhead (*Oncorhynchus mykiss*) as Endangered under the California Endangered Species Act" (Evaluation). Based upon the information contained in the Petition, the Department concluded, pursuant to Fish and Game Code Section 2073.5, that sufficient information exists to indicate that the petitioned action may be warranted and recommended to the Commission that the Petition be accepted and considered.

On April 21, 2022, at its public meeting pursuant to Fish and Game Code Sections 2074 and 2074.2, the Commission considered the Petition, the Department's Evaluation and recommendation, comments received, and oral testimony. The Commission found that sufficient information exists to indicate the petitioned action may be warranted and accepted the Petition for consideration.

On May 13, 2022, pursuant to Fish and Game Code Section 2074.2, the Commission published its Notice of Findings for southern California steelhead in the California Regulatory Notice Register, designating southern California steelhead as a candidate species (Cal. Reg. Notice Register 2022, No. 19-z, p. 541).

On October 12, 2022, pursuant to Fish and Game Code Section 2074.6, the Commission approved the Department's request for a six-month extension to complete its status review report.

1.2 Status Review Overview

Pursuant to Fish and Game Code Section 2074.6 and the California Code of Regulations, Title 14, Section 670.1, the Department has prepared this status review to indicate whether the petitioned action to list southern California steelhead as endangered under CESA is warranted (Status Review). An endangered species under CESA is "a native species or subspecies . . . which is in serious danger of becoming extinct throughout all, or a significant portion, of its range due to one or more causes, including loss of habitat, change in habitat, overexploitation, predation, competition, or disease" (Fish & G. Code, § 2062). A threatened species under CESA is "a native species or subspecies . . . that, although not presently threatened with extinction, is likely to become an endangered species in the foreseeable future in the absence of the special protection and management efforts required by [CESA]" (*id.* at § 2067). A species' range for CESA purposes is the species' California range (Cal. Forestry Assn. v. Cal. Fish and Game Com. (2007) 156 Cal.App.4th 1535, 1551).

Using the best scientific information available to the Department, this Status Review includes information on each of the following components pursuant to Fish and Game Code Section 2072.3 and Title 14 of the California Code of Regulations Section 670.1: population trend(s), range, distribution, abundance, life history, factors affecting the species' ability to survive and reproduce, the degree and immediacy of threats, the impact of existing management efforts, the availability and sources of information, habitat that may be essential to the continued existence of the species, and the Department's recommendations for future management activities and other recovery measures to conserve, protect, and enhance the species.

Southern California steelhead, as defined in the Petition, means all *O. mykiss*, including anadromous and resident life histories, below manmade and natural complete barriers to anadromy from and including the Santa Maria River (San Luis Obispo and Santa Barbara counties) to the U.S.-Mexico Border (CDFW 2021a Petition Evaluation). The Department accepts the taxonomy as published by Behnke (1992) that identifies southern California *O. mykiss* as being included in the range of Coastal Rainbow Trout (*O. mykiss irideus*), which have a broad distribution extending from Alaska to Baja California (Moyle 2002). The Department has long referred to these fish as "steelhead rainbow trout" (Shapovalov and Taft 1954), which captures the life history variability that is included in the scope of this status review for both anadromous and resident forms of the species. Thus, the Department will refer to the Petitioner's proposed listing unit as southern California steelhead rainbow trout (*O. mykiss*;

Southern SH/RT) throughout the remainder of this Status Review. This naming convention is slightly different than what was used by the Petitioner in the Petition, but the Department asserts the importance of recognizing the full scope of life history diversity included in the listing unit.

This Status Review report is not intended to be an exhaustive review of all published scientific literature relevant to the Southern SH/RT. Rather, it is intended to summarize the best scientific information available relevant to the status of the species, provide that information to the Commission, and serve as the basis for the Department's recommendation to the Commission on whether the petitioned action is warranted. Specifically, this Status Review analyzes whether there is sufficient scientific information to indicate that the continued existence of Southern SH/RT throughout all or a significant portion of its range is in serious danger or is threatened by one or a combination of the following factors: present or threatened modification or destruction of its habitat; overexploitation; predation; competition; disease; or other natural occurrences or human-related activities (Cal. Code Regs., tit. 14, § 670.1, subd. (i)(1)(A)).

1.3 Federal Endangered Species Act Listing History

The federal Endangered Species Act (ESA) defines "species" to include "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature" (16 U.S.C. § 1532). In 1991, the National Marine Fisheries Service (NMFS) adopted its policy on how it would apply the definition of "species" to Pacific salmon stocks for listing under the ESA (ESU Policy). Under the ESU Policy, a salmon stock is considered a distinct population segment (DPS) if it constitutes an evolutionary significant unit (ESU) of the biological species (NMFS 1991). In February 1996, the United States Fish and Wildlife Service (USFWS) and NMFS published a joint DPS policy for the purposes of ESA listings (DPS Policy) (NMFS 1996a). Section 3.1 of this Status Review describes the ESU Policy and DPS Policy in greater detail.

In 1997, NMFS listed the Southern California Steelhead ESU as endangered under the federal ESA. The Southern California Steelhead ESU only included naturally spawned populations of anadromous *O. mykiss* (and their progeny) residing below long-term, natural and manmade impassable barriers in streams from the Santa Maria River, San Luis Obispo County (inclusive) to Malibu Creek, Los Angeles County (inclusive) (NMFS 1997). In 2002, NMFS extended the geographic range of the Southern California Steelhead ESU listed under the federal ESA south to the U.S.-Mexico border (NMFS 2002).

In 2001, the U.S. District Court in Eugene, Oregon, ruled that NMFS improperly excluded certain hatchery stocks from the listing of Oregon Coast Coho Salmon after NMFS had concluded that

those hatchery stocks were part of the ESU being considered for listing but not essential for recovery (*Alsea Valley Alliance v. Evans* (D. Or. 2001) 161 F. Supp. 2d 1154, 1162). Based in part on the *Alsea* decision, in 2002 NMFS announced that it would conduct an updated status review of 27 West Coast salmonid ESUs, including the Southern California Steelhead ESU (NMFS 2006). In 2004, NMFS proposed to continue applying its ESU Policy to the delineation of DPSs of *O. mykiss* and to include resident *O. mykiss* that co-occur with the anadromous form of *O. mykiss* in 10 *O. mykiss* ESUs, including the Southern California Steelhead ESU (NMFS 2006).

In 2005 USFWS wrote to NMFS stating USFWS's "concerns about the factual and legal bases for [NMFS's] proposed listing determinations for 10 *O. mykiss* ESUs, specifying issues of substantial disagreement regarding the relationship between anadromous and resident *O. mykiss*" (NMFS 2006). After discussions with USFWS regarding the relationship between anadromous and non-anadromous *O. mykiss*, in 2006 NMFS decided to depart from their past practice of applying the ESU policy to *O. mykiss* stocks and instead apply the joint DPS Policy (NMFS 2006). Concurrent with that decision, NMFS relisted the Southern California Steelhead ESU as the Southern California Steelhead DPS under the federal ESA (NMFS 2006). As part of its 2006 relisting of southern California steelhead, NMFS concluded that the anadromous life form of *O. mykiss* is markedly separate from the non-anadromous life form of *O. mykiss* within the geographic boundary of the Southern California Steelhead DPS—as well as the geographic boundaries of the other nine *O. mykiss* ESUs that NMFS was relisting as DPSs at that time—due to "physical, physiological, ecological, and behavioral factors" (NMFS 2006). The Southern California Steelhead DPS only includes the anadromous life-history component of *O. mykiss* and is defined as including all naturally spawned anadromous *O. mykiss* (steelhead) populations below natural and manmade impassible barriers in streams from the Santa Maria River, San Luis Obispo County (inclusive) to the U.S.-Mexico border (Table 1) (NMFS 2006).

2. BIOLOGY AND ECOLOGY

2.1 Species Description

The species *O. mykiss* is one of the most widely distributed of Pacific salmonids, occupying nearly all coastal streams from Alaska to southern California and from Russia's Kamachotka Peninsula to South Korea in the western Pacific. Steelhead is the common name for the anadromous form of *O. mykiss*, while Rainbow Trout is the common name applied to the freshwater resident form (Behnke 1993; Moyle 2002). *O. mykiss* possess 10–12 dorsal fin rays, 8–12 anal fin rays, 9–10 pelvic fin rays, 11 – 17 pectoral fin rays, and a slightly forked caudal fin (Moyle 2002). They have 9–13 branchiostegal rays and 16–22 gill rakers on each arch (Moyle 2002). Teeth are present on both upper and lower jaws, the tip and shaft of the vomer, as well

as on the tip of the tongue (Fry 1973; Moyle 2002). Between 110–180 small, pored scales make up the first row above the lateral line (Fry 1973; Moyle 2002).

*Table 1. Common nomenclature for *Oncorhynchus mykiss* (adapted from Boughton et al. 2022b).*

Term	Description
<i>Oncorhynchus mykiss</i>	A species of Pacific salmonid composed of both anadromous and freshwater-resident forms, which all spawn in freshwater rivers and streams.
Steelhead	Individuals: <i>O. mykiss</i> that are anadromous (individuals that migrate to and spend one or more seasons in the ocean); here used to mean adult steelhead.
Rainbow Trout	Individuals: <i>O. mykiss</i> that are freshwater resident (individuals that complete their life cycle in freshwater), here used to mean adult Rainbow Trout.
Steelhead Rainbow Trout	Population(s): contains both steelhead individuals and Rainbow Trout individuals.
Juvenile <i>O. mykiss</i>	Immature fish whose fate as steelhead or Rainbow Trout cannot yet be established.
Anadromous waters	Stream reaches that are accessible to migrating steelhead (those not blocked by complete natural or artificial barriers). It is important to note that <i>Oncorhynchus mykiss</i> individuals, occurring in anadromous waters, may or may not express the anadromous life history type (e.g., smoltification).

The steelhead life history form is thought to be named for the sometimes silvery-metallic appearance of its back and head. The steelhead body profile is fusiform, with typically “bullet-shaped” heads and distinct narrowing at the base of a powerful tail, suited for often-demanding and lengthy upstream spawning migrations. In the marine environment, steelhead body coloration includes a blueish-green dorsum (back) and silver or white coloration over the rest of the body (Fry 1973; Moyle 2002). Black spots typically cover the dorsal, adipose, and caudal fins, as well as the head and back (Fry 1973). When adult steelhead return to spawn in freshwater, their silver sheen fades and a pink or red lateral band develops along the sides and on the opercula, while the silvery-blue coloration on the back transitions to an olive green or brown (Barnhart 1986). These characteristics are very similar to those exhibited by resident Rainbow Trout (Fry 1973); thus, it can be difficult to differentiate the anadromous and resident

forms based only on outward appearance. Adult steelhead, however, are generally larger than adult Rainbow Trout in a given stream system since they spend time feeding and growing in the ocean (NWF 2020; USFWS 2020).

Juvenile *O. mykiss* have body coloration similar to that of resident adults, while also exhibiting 5–13 oval parr marks along the lateral line on both sides of the body (Moyle 2002). These parr marks are dark bluish-purple in coloration and are widely spaced, with the marks themselves being narrower than the spaces between them (Moyle 2002). A total of 5–10 dark spots also line the back, typically extending from the head to the dorsal fin. There are usually few to no marks on the caudal fin, and the tips of the dorsal and anal fins are white to orange (Moyle 2002).

After a year or more of development, some *O. mykiss* undergo the transitional process of smolting, which is a series of morphological, physiological, and behavioral changes that prepare the fish for entry into brackish estuaries and then ocean environments (Fessler and Wagner 1969; McCormick 2012). Smolting is the primary physiological characteristic that distinguishes the anadromous life history variant from the resident one within the species. Smolts lose their parr marks and develop silver coloration during the downstream migration process. After entering the ocean, young steelhead will reside in the saltwater environment for 1–4 years while feeding and growing quickly (Moyle 2002). Juvenile *O. mykiss* that do not smolt and remain in freshwater generally lose their parr marks as they grow and develop into adult Rainbow Trout.

The sexual maturation process for anadromous steelhead involves the development of secondary sex characteristics such as bright coloration and sexual dimorphism, including the development of a hooked snout, or kype, in males. These secondary sex characteristics are typically reabsorbed once spawning is complete, although jaw shape may never fully revert to the pre-spawn condition (Shapovalov and Taft 1954).

Different populations of *O. mykiss* can exhibit variations in growth rate, size, and body shape depending on their life histories and habitats utilized. For example, Bajjaliya et al. (2014) studied morphometric variation between four California steelhead DPSs and found that coastal steelhead (populations with adults migrating less than 160 km from the ocean to their sample site) were significantly larger in size and had a more robust body type than steelhead found in California's Central Valley drainages and the Klamath-Trinity basin (populations with adults migrating more than 160 km from the ocean to their sample site). These morphological differences provided the basis for recognizing "coastal type" and "inland type" steelhead in California (Bajjaliya et al. 2014). However, the morphometric variation in populations of steelhead occurring in more southerly DPSs, such as the Southern California Steelhead DPS,

may include features of both the large, coastal type as well as smaller, inland-type *O. mykiss* that occur in interior drainages (Bajjaliya et al. 2014).

2.2 Taxonomy and Systematics

Steelhead and Rainbow Trout are members of the bony fish class Osteichthyes, in the order Salmoniformes and family Salmonidae. In 1792, J. J. Walbaum classified Rainbow Trout from populations on the Kamchatka Peninsula in Russia as *Salmo mykiss* (Moyle 2002). During the next century, using J. Richardson's description of Columbia River steelhead as *S. gairdneri* and Gibbons's description of juvenile steelhead from San Leandro Creek as *S. iridea*, both the biology and fishing communities began referring to resident Rainbow Trout and steelhead as *S. irideus* and *S. gairdneri*, respectively. It was ultimately discovered that Rainbow Trout and steelhead are the same species, and North American scientists applied the original species name, *mykiss*, to North American populations (Moyle 2002).

In the 1970s, analyses of polymorphic proteins, or allozymes, were utilized to determine the degree of species relatedness and evolutionary divergence among salmonids (Quinn 2018). These studies indicated that Coho and Chinook salmon (*O. kisutch* and *O. tshawytscha*, respectively) were most closely related to Pink, Chum, and Sockeye salmon, and that Rainbow and Cutthroat trout were most closely related to each other (Quinn 2018). This phylogeny was assumed until researchers analyzed relatedness by looking at differences in mitochondrial DNA, which showed that Coho and Chinook salmon were related more closely to steelhead than they were to the other three genera of salmon (Quinn 2018). Based on this study, Smith and Stearley (1989) reorganized the taxonomy to reflect both the use of the name *mykiss* for North American Rainbow Trout and the inclusion of Rainbow and Cutthroat trouts in the Pacific salmon genus *Oncorhynchus*, but with their own distinct lineages.

Pacific salmonid lineages continue to be studied using a variety of genetic and statistical methods (Quinn 2018). There has been debate over the relationship between Rainbow and Cutthroat trouts with regards to genetics versus morphology and behavior. Stearley and Smith (1993) and Esteve and McLennan (2007) found that the idea of monophyly (a group descending from a most recent common ancestor) of these two trout species is not supported by either morphological or behavioral traits, even though mitochondrial DNA suggests otherwise. Esteve and McLennan (2007) attribute this contradiction to hybridization events that have led to a high rate of genetic introgression between the two species (Chevassus 1979). This introgression can dilute the distinctiveness of these close relatives and convolute phylogenetic reconstruction (Esteve and McLennan 2007). Although some uncertainty remains surrounding these evolutionary relationships, it is now accepted that within the genus *Oncorhynchus*, Coho and Chinook salmon have the closest relationship to each other, with Pink (*O. gorbuscha*), Chum (*O.*

keta), and Sockeye (*O. nerka*) salmon in their own group, and Rainbow (*O. mykiss*) and Cutthroat (*O. clarkii*) trout in another group (Kitano et al. 1997; Crête-Lafrenière et al. 2012; Quinn 2018; Figure 1).

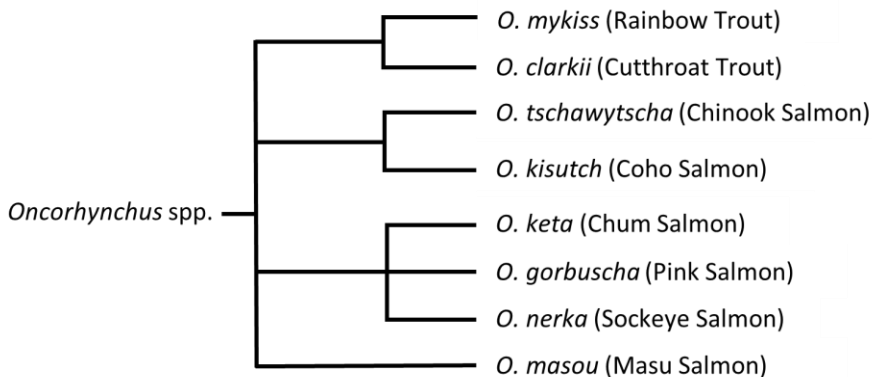


Figure 1. Consensus relationships of *Oncorhynchus* species from morphological, allozyme, ribosomal RNA, mitochondrial DNA, and short interspersed repetitive elements data across multiple studies. Adapted from Figure 1 in Kitano et al. (1997)

2.3 Range and Distribution

Range is the general geographical area in which an organism occurs. For purposes of CESA and this Status Review, the range is the species' California range (*Cal. Forestry Assn. v. Cal. Fish and Game Com.* (2007) 156 Cal.App.4th 1535, 1551). Distribution describes the actual sites where individuals and populations of the species occur within the species' range.

Oncorhynchus mykiss is native to both coastlines of the Pacific Ocean and spawns in freshwater streams, from the Kuskokwim River in Alaska, south to Baja California along the eastern Pacific, and from Russia's Kamchatka Peninsula in the western Pacific (Moyle 2002). The species is widely distributed throughout the northern Pacific Ocean during its ocean phase. Coastal steelhead within the state historically occupied all perennial coastal streams, from the Oregon/California border to the U.S.-Mexico border (Moyle 2002). Steelhead are also native to the Central Valley, including both the Sacramento and San Joaquin River basins, and have been found as far upstream as the Pit and McCloud rivers (Moyle 2002). It is likely that most suitable streams in the Sacramento and San Joaquin River basins with ocean access have historically supported runs of steelhead (Moyle 2002).

Southern SH/RT currently occupy fluvial habitat from the Santa Maria River at the border of San Luis Obispo and Santa Barbara counties south to the U.S.-Mexico border. This range encompasses five biogeographic population groups (BPGs), collectively described by NMFS as the Southern California steelhead DPS (Boughton et al. 2007; NMFS 2012a). BPGs are steelhead

subpopulations within a DPS that occupy contiguous areas that share broadly similar physical geography and hydrology, generally within a single watershed unit. The combinations of these physical characteristics represent the suite of differing natural selective regimes across the watersheds occupied by Southern SH/RT. These varying selective pressures have led to life history and genetic adaptations that enable subpopulations to persist in distinctive and dynamic habitats that have shaped each BPG. The purpose of delineating BPGs for steelhead populations is to ensure the preservation of the range of genetic and natural diversity within each DPS for recovery and conservation purposes (NMFS 2012a). The BPGs that form the Southern SH/RT DPS are (from north to south): Monte Arido Highlands, Conception Coast, Santa Monica Mountains, Mojave Rim, and Santa Catalina Gulf Coast.

While some near-coastal populations of Southern SH/RT are small, there are likely dispersal dynamics that contribute to their stability and persistence (Boughton et al. 2007). The movement of spawning adults between BPGs may be an important mechanism for maintaining the viability of steelhead populations (NMFS 2012a). Dams and other impediments obstruct access to a significant portion of historical Southern SH/RT habitats in many rivers within the proposed listing area, some of which have multiple major dams on a single mainstem. There is evidence that loss of access to upstream habitat has resulted in a northward range contraction of anadromous Southern SH/RT (Boughton et al. 2005), whose study also found a strong correlation between steelhead population extirpations and anadromous barriers, as well as urban and agricultural development.

2.4 Life History

An individual fish's genotype, condition, and a variety of environmental factors influence the expression of anadromy versus stream residency (Sloat et al. 2014; Busby et al. 1996; Pascual et al. 2001; Courter et al. 2013). Juvenile *O. mykiss* prior to the smolting life stage are difficult to distinguish without genetic, morphological, or physiological evaluations (Negus 2003; Beeman et al. 1995; Haner et al. 1995; Pearse et al. 2014). Adult steelhead returning to streams from the ocean are often easier to identify due to their larger size relative to most resident Rainbow Trout adults in the same stream system and their overall steel-gray color (Dagit et al. 2020). While anadromy and residency are the two primary life histories, *O. mykiss* life history expression is notably plastic and can be quite variable (Moyle 2002). For example, individuals may exhibit the lagoon-anadromous life history, spending their first or second summer rearing in seasonal lagoons in the estuaries of streams before outmigrating to the ocean (Boughton et al. 2007).

Unlike other Pacific salmonids, which are semelparous and perish almost immediately after spawning, *O. mykiss* can be iteroparous (Moyle 2002), with the potential to spawn up to four

times but typically not more than twice (Shapovalov and Taft 1954). Steelhead that spawn and return to the sea are called “kelts.” These fish can either spawn consecutively, returning the next season after their first spawn, or they may return a year later after spending an extra year at sea (Light et al. 1989). Reportedly, females survive spawning events more frequently than males (Shapovalov and Taft 1954; Ward and Slaney 1988; Busby et al. 1996; Marston et al. 2012), although males can repeat spawn in significant numbers, especially in smaller, near-coastal stream systems (Marston et al. 2012).

Steelhead exhibit two seasonal migratory patterns, or run types: 1) winter, also called “ocean-maturing” or “mature-migrating;” and 2) summer, also called “stream-maturing” or “premature-migrating.” The names of these two runs are reflective of the seasonal timing when adult steelhead reenter estuaries and rivers to reproduce (Busby et al. 1996; Moyle 2002). Only the winter-run form of steelhead occurs in southern California streams, consistent with what is believed to be the historical condition (Moyle 2002). Southern SH/RT typically begin migrating upstream from December through May, with returning adults often reliant upon winter rainstorms to breach sandbars at the mouths of stream estuaries and lagoons, providing seasonal upstream spawning passage (California Trout 2019). Steelhead age-at-maturity is dependent on a number of factors, including time spent in either or both freshwater and marine environments; however, adult returning spawners are usually 3 or 4 years old, having spent 1-3 years in freshwater and 1-2 years at sea (Shapovalov and Taft 1954). Southern SH/RT steelhead spawning runs are dominated by age 3+ fish, with 2 years spent in fresh water and 1 year in the ocean, although many smolt after only 1 year in fresh water (Busby et al. 1996). Shapovalov and Taft (1954) found that the average age of male spawners (about 3.5 years) was lower than that of female spawners (close to 4 years) in Waddell Creek, CA. Non-anadromous Rainbow Trout can mature anywhere between 1 and 5 years but are commonly age 2+ or 3+ years, with a fork length of ≥ 13 cm (Moyle 2002). Rainbow Trout typically spawn during the spring months, from February through June (Moyle 2002).

Spawning usually occurs in shallow habitats with fast-flowing water and suitable-sized gravel substrates, often found in riffles, faster runs, or near the tail crests of pool habitats. When female *O. mykiss* are ready to spawn, they will select a suitable spawning site and excavate a nest, or redd, in which they deposit their eggs to incubate (Moyle 2002). Adequate stream flow, gravel size, and low substrate embeddedness are crucial for egg survival, as these conditions allow oxygenated water to permeate through sediments to the egg (Coble 1961). During redd construction, the female may be courted by multiple males. Following completion of the redd, the most dominant males fight for position alongside the female, depositing milt while the female deposits her eggs (Quinn 2018). Immediately following fertilization, females cover their eggs with gravel (Barnhart 1986). Females dig multiple smaller pits within the broader redd where they deposit a portion of eggs into each pocket until all the eggs are expelled

(Shapovalov and Taft 1954; Quinn 2018). Adult steelhead are often accompanied by resident male Rainbow Trout during spawning, as they attempt to participate by quickly swimming, or darting, in and out of steelhead redds (Shapovalov and Taft 1954). These fish are sometimes referred to as “egg-eaters,” although it is generally accepted that the main purpose of their presence is to contribute to spawning rather than consume newly laid eggs (Shapovalov and Taft 1954). If adult steelhead cannot emigrate back to the ocean after spawning, they require large, deep pools that provide refuge during the hot summer months (Boughton et al. 2015).

Fecundity, among other biological and environmental factors, contributes substantially to reproductive success. Egg production is positively correlated with fish length, although there is wide variation in female steelhead fecundity at a given size (Shapovalov and Taft 1954; Quinn 2018). Larger females tend to produce larger and greater numbers of eggs; however, energy demands for gonad development create a physiological tradeoff between the number and size of eggs produced (Quinn 2018). Thus, females generally produce either many smaller eggs or fewer larger eggs. Quinn (2018), referencing multiple sources of data, showed that female steelhead of average size produce slightly over 5,000 eggs. Moyle (2002) provides a range of eggs per female from 200 to 12,000 and states that steelhead generally produce about 2,000 eggs per kilogram of body weight. Rainbow Trout less than 30 cm in total length usually have under 1,000 eggs per kilogram of body weight (Moyle 2002).

Multiple factors contribute to egg development and incubation time; however, eggs generally incubate in stream gravels for up to several months. Temperature has the greatest effect on the incubation period; colder water slows development, and warmer water increases the rate of development (Quinn 2018). Incubation can take from 19 days at an average temperature of 60°F (15.6°C) to 80 days at an average temperature of 40°F (4.4°C) (Shapovalov and Taft 1954). Dissolved oxygen (DO) levels in surrounding waters also influence life stage development rates in Southern SH/RT and other salmonids. Higher DO levels lead to more rapid egg development, while eggs exposed to low levels of DO during incubation produce much smaller alevins (yolk-sac fry) than those exposed to high DO (Quinn 2018). Fry emerge from the gravel 2-3 weeks after hatching, once the yolk sac is fully or almost entirely absorbed, at which time they form schools along stream banks (Shapovalov and Taft 1954). During their first year of life, *O. mykiss* juveniles develop small territories and defend them against other individuals in their age class (Shapovalov and Taft 1954; Barnhart 1986). Juvenile *O. mykiss* generally feed on many different species of aquatic and terrestrial insects, sometimes cannibalizing newly emerged fry (Barnhart 1986). Further north, feeding generally peaks during the summer months and is depressed during the winter months; however, *O. mykiss* in California typically have higher growth rates in the winter and spring than summer and fall (Hayes et al. 2008; Sogard et al. 2009; Krug et al. 2012). As they grow, juveniles will move into deeper, faster water and are often found in riffle or swift-run habitats (Shapovalov and Taft 1954; Barnhart 1986). Larger juvenile *O. mykiss* can

outcompete and displace their smaller counterparts from ideal habitats, such as deep pools or run complexes, leaving smaller individuals to often inhabit suboptimal habitats, such as riffles (Barnhart 1986).

Parr will ultimately begin transitioning into smolts and migrate downstream to estuaries and lagoons, where they complete the process of smolting. Smolt outmigration to the ocean typically occurs from March–May in southern California but can vary depending on factors such as connectivity between the ocean and estuary or lagoon and streamflow (Booth 2020). Compared to other Pacific salmonids, steelhead have the greatest variability in the timing and duration of freshwater inhabitance, ocean entry, time spent at sea, and return to freshwater (Barnhart 1986). Resident Rainbow Trout early life stages mirror those of anadromous steelhead, up until their life history strategies diverge (Moyle 2002). Rather than migrating out to the ocean like steelhead, resident *O. mykiss* will reside in freshwater for the remainder of their lives.

Little is known regarding steelhead stock-specific utilization of and distribution in the ocean environment. While much is known about the status and abundance of commercially important ocean stocks of Pacific salmon, steelhead-specific research on this topic is lacking and hampered by the inability to differentiate individual stocks using standard sampling methods (Barnhart 1986; Light et al. 1989; Moyle 2002). Unlike Pacific salmon species, steelhead are rarely captured in the ocean; therefore, information specific to Southern SH/RT ocean distribution is not available. Limited tag recoveries by North American fisheries research and management agencies showed no differences in the ocean distribution of steelhead by stock (Light et al. 1989). Attempts to distinguish steelhead population units from one another in terms of ocean distribution are confounded by findings that all steelhead apparently congregate in shared ocean feeding grounds, regardless of their origin or run type (Light et al. 1988).

Pacific steelhead smolts quickly migrate offshore after entry into the ocean (Daly et al. 2014) and, once in the open water, generally move in a northwestern trajectory from spring to summer and follow a southeastern pattern from fall to winter (Okazaki 1983; Light et al. 1989). In the winter, steelhead are found in the eastern North Pacific (Myers et al. 2016) and tend to be closer to shore than during other times of the year (Light et al. 1989). California steelhead do not appear to migrate any farther west than the Gulf of Alaska (Light et al. 1989), and, overall, steelhead migration patterns appear to be strongly tied to “thermal avoidance.” Migratory-based thermal avoidance involves fish movement patterns that remain within a narrow range of tolerable sea surface temperatures, suggesting that steelhead ocean migration may be largely influenced by physiological responses to temperature (Hayes et al. 2016). Ocean

steelhead are typically found within seven meters of the sea surface, within the epipelagic zone, although they have been found at more than three times that depth (Light et al. 1989).

Studies addressing steelhead ocean behavior, distribution, and movement are limited; however, as with other salmonids, steelhead tend to exhibit strong homing behavior to their natal streams, with some exceptions. Evidence of straying has been documented in central California steelhead populations (Donohoe et al. 2021), while genetic population structure analyses suggest that historical (natural) exchange of genetic information occurred between coastal populations of steelhead (Garza et al. 2014).

2.5 Genetics and Genomics

2.5.1 Role of Genetics and Genomics in Evaluating Steelhead Population Structure

To date, most genetic studies focused on quantifying the population structure of salmonid species have used neutral genetic markers (e.g., microsatellite DNA). Neutral markers are not directly linked with a particular life history trait, and it is assumed that they are not under direct selection. This class of genetic marker continues to be used to investigate and define salmonid listing units and population structure (e.g., Busby et al. 1996) in both California and across the Pacific Northwest. These types of markers have also been successfully used for decades to delineate populations and ESUs based primarily on reproductively isolated lineages. These markers remain valuable, in that they are the standard for determining the genetic structure and relatedness of species and, thus, their evolutionary histories.

More recently, the advent and rapid development of “adaptive” genetic markers have provided fishery managers and geneticists with a new suite of tools. Adaptive genetic markers provide putative associations with specific life history characteristics, and the “genetic type”, or “variant” infers information about a phenotype of interest. Specific genes, or genomic regions, within individuals or subgroups may vary from the overall pattern exhibited by a species. Of particular relevance to Southern SH/RT is the role that adaptive genetic variation plays in migratory behavior. This relationship is still being evaluated, and uncertainties remain regarding the level of influence genetics may have on migration phenotype. See Section 2.6.5 for more information.

*2.5.2 Patterns of *O. mykiss* Genetic Population Structure*

Geography and local environmental factors influence the genetic structure of *O. mykiss* populations, a pattern referred to as "isolation by distance". Evidence of isolation by distance is shown in *O. mykiss* populations throughout their range. Studies based on neutral mitochondrial DNA analysis have demonstrated a pattern of isolation by distance in populations spanning the

western coast of the United States, including among coastal California steelhead populations (Hatch 1990; Reisenbichler et al. 1992; McCusker et al. 2000). Nielsen (1999) found a pattern of isolation by distance when looking at the microsatellite loci of southern California and northern California steelhead populations. Bjorkstedt et al. (2005) suggested that genetic variation in salmonid populations generally increases with greater distances between watersheds. Pearse et al. (2007) analyzed geographic structure within the Klamath-Trinity River basin and consistently found a positive relationship between geographic distance and genetic relatedness—specifically, that genetic divergence between populations increased as a function of geographic distance.

Garza et al. (2004) evaluated population structure across coastal California populations using microsatellite loci to understand the relationship between genetic distance and the geography of coastal steelhead populations. This study's results included a bootstrap consensus tree showing clustering of geographic locations corresponding to five DPS assignments in coastal California steelhead (Figure 2). The long terminal branches in this consensus tree demonstrate that, while migration is important to the populations in this study, the conflicting evolutionary processes of random genetic drift and local adaptation were likely responsible for the genetic differentiation between the populations. The general isolation-by-distance pattern of genetic diversity is also visually apparent.

Aguilar and Garza (2006) found a significant relationship between geographic distance and genetic distance in coastal *O. mykiss* using both major histocompatibility complex genes, which can be helpful in identifying salmonid population structure, and microsatellite loci. This significant relationship represented isolation through distance. Garza et al. (2014) reaffirmed that genetic variation is associated with isolation by distance using microsatellite loci from samples of coastal California steelhead. Across all coastal California steelhead populations sampled, there was evidence that population structure is dependent on geographic distance. Their phylogeographic trees also suggested that population structure was almost entirely consistent with geographic proximity.

Populations within a watershed, even those disconnected by barriers, have been shown through microsatellite DNA analyses to be more genetically similar than those in adjacent watersheds (Clement et al. 2009; Garza et al. 2014). However, anthropogenic impacts including stocking, barrier construction, and habitat destruction have resulted in weaker relationships between geographic proximity and relatedness in modern *O. mykiss* populations (Pearse et al. 2011).

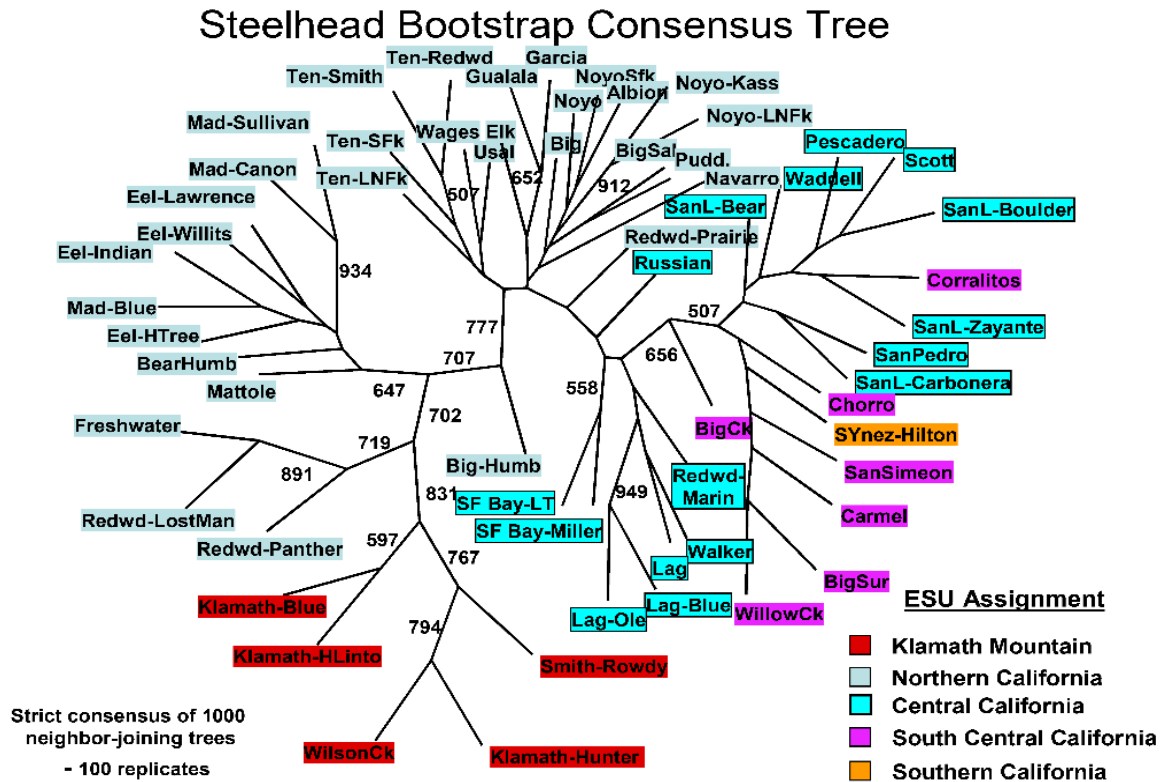


Figure 2. Majority-rule consensus tree, with genetic data bootstrapped 1,000 times, showing chord distances and neighbor-joining trees for 62 coastal California steelhead populations. (from Garza et al. 2004).

2.5.3 Genetics of the Southern California SH/RT

Busby et al. (1996) posited that the extreme environmental conditions found in southern California could result in both substantial local adaptations of and gene flow impediments between *O. mykiss* populations in the region. Nielsen (1999) hypothesized that the substantial interpopulation genetic diversity found in southern California's mostly small and somewhat isolated *O. mykiss* populations could be the result of a transitional ecotone, where two adjacent Pleistocene source populations have met and blended. Allozymes, mitochondrial DNA, and microsatellites have uncovered significant and unique genetic diversity in southern California steelhead, with traits not found in more northern populations. Busby et al. (1996) noted that a mitochondrial DNA type exists in steelhead populations between the Santa Ynez River and Malibu Creek that is rare in populations to the north, and samples from Santa Barbara County were found to be the most genetically unique of any wild coastal steelhead populations analyzed. In general, *O. mykiss* at the extreme southern end of their range have low genetic diversity (Clemento et al. 2009; Pearse et al. 2009; Jacobson et al. 2014; Abadía-Cardoso et al. 2016; Apgar et al. 2017). Loss of genetic diversity is often a consequence of declines in

population size (Allendorf et al. 1997), which have been observed in Southern SH/RT populations.

2.5.4 South-Central and Southern California Genetic Relationships

Clemento et al. (2009) conducted a genetic analysis of steelhead populations in California south of Monterey Bay using microsatellite data to elucidate patterns of genetic differentiation and gene flow. In terms of coastwide population structure, the authors found that southern California steelhead populations were grouped with all other steelhead populations south of San Francisco Bay and were well-distanced from populations north of San Francisco Bay. Population genetic structure does not correspond with geographic management boundaries because genetically based population clusters are not separated by current federal-ESA-listed DPS boundaries. Overlap in clustering was detected between populations from nearby watersheds, and genetic differentiation between populations in the South-Central California Coast steelhead DPS and the southern California steelhead DPS could not be detected. Additionally, the construction of phylogeographic trees did not result in the separation of populations from the two DPSs into distinct genetic lineages based on their current ancestry (Figure 3). In populations south of San Francisco Bay, no apparent isolation by distance pattern corresponding with DPS boundaries was detected. This may be a result of metapopulation dynamics occurring between these *O. mykiss* populations. Although a lack of genetic differentiation was observed across these southern DPSs, allozymes, mitochondrial DNA, and microsatellites have uncovered significant and unique genetic diversity in southern California steelhead (see Section 3.2.2 for more information). Further, the Department recognizes other factors that define Southern SH/RT, such as unique regional biogeography, ecology, physiology, and behavior of the population groups (Boughton et al. 2007).

2.5.5 Role of Genetics in Life History Expression

Many *O. mykiss* populations are considered “partially migratory,” meaning they contain both migratory (e.g., anadromous) and non-migratory (e.g., resident) individuals (Chapman et al. 2011). It is widely accepted that migratory behavior and migration-associated traits are heritable in partially migratory populations (Pearse et al. 2014; Hecht et al. 2015; Phillis et al. 2016). In recent years, studies have revealed that important migration-related characteristics in *O. mykiss*, such as maturation, growth, development, and smolting, are linked to specific genomic regions that are under natural selection (Nichols et al. 2008; Martínez et al. 2011; Hecht et al. 2012; Miller et al. 2012; Pearse et al. 2014). Phenotypic expression of anadromy vs. residency has since been found to be strongly associated with a large genomic region on *O. mykiss* chromosome 5 (*Omy5*) (Martínez et al. 2011; Hecht et al. 2012; Pearse et al. 2014; Leitwein et al. 2017; Kelson et al. 2019). This *Omy5* migration-associated region exhibits unique

alleles, associated with either anadromy or residency as their phenotypic expression, and these *Omy5* genetic variants are thought to be the result of a chromosomal inversion (Pearse et al. 2014; Leitwein et al. 2017).

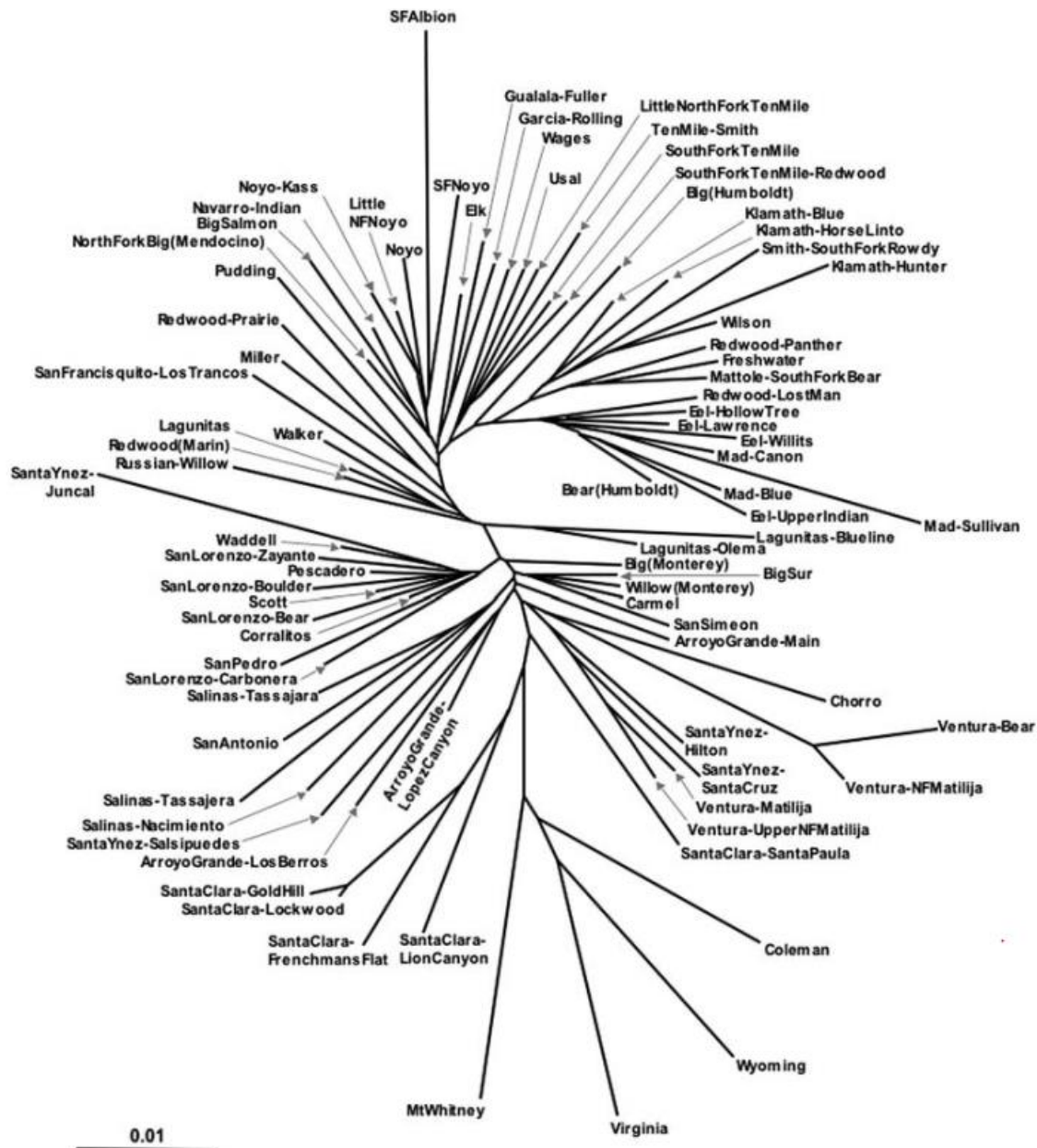


Figure 3. Unrooted neighbor-joining chord distance tree of 84 coastal *O. mykiss* populations in California (from Clemente et al. 2009).

Chromosome *Omy5* is associated with multiple life history characteristics related to migration vs. residency in *O. mykiss*, explaining morphological and developmental variation between the two life history forms (Nichols et al. 2008; Martínez et al. 2011; Hecht et al. 2012). Nichols et al.

(2008) used quantitative trait loci analysis to locate specific loci associated with smolting and found several genomic regions that were linked with morphological and physiological smolting indicators. The study was the first of its kind in terms of finding connections between specific genomic loci and the migration characteristics of a species of fish. In addition, Martínez et al. (2011) found multiple microsatellite markers on *Omy5* that were correlated with differential selection between anadromous and resident *O. mykiss*, while Hecht et al. (2012) identified associations between *Omy5*, body morphology, and skin reflectance, which are linked to the smolting process and the anadromous phenotype. Pearse et al. (2014) found that specific *Omy5* loci diverged between above-barrier and below-barrier *O. mykiss* populations that had differing frequencies of the anadromous phenotype.

Populations with higher potential to support anadromous or migratory individuals typically have a higher population-wide frequency of the anadromous variant of *Omy5* than populations that have a higher frequency of the resident rainbow trout, such as those above manmade and natural barriers (Pearse et al. 2014; Leitwein et al. 2017). This suggests that utilizing comparative anadromous *Omy5* variant frequency data between steelhead populations may indicate which populations have a higher likelihood of producing anadromous offspring, as well as having utility in identifying above-barrier populations with the genetic potential to support or bolster downstream anadromous populations. Results from Kelson et al. (2020) suggest that the *Omy5* genomic region also regulates physiological traits, such as juvenile growth, which will subsequently influence residency vs. anadromy (Figure 4).

Sex determination has also been genetically linked to the migratory phenotype of *O. mykiss* (Rundio et al. 2012). Migratory ecotype composition within a population is typically female-dominated, a phenomenon that has been observed in multiple salmonid species (Jonsson et al. 1998; Páez et al. 2011; Ohms et al. 2014; Kelson et al. 2019) and may be due to a strong correlation between fecundity and body size (Hendry et al. 2004; Quinn 2018). Female steelhead that migrate to the ocean can grow larger in the highly productive marine environment than their counterparts in the less productive freshwater environment and, as a result, produce greater numbers of embryos. Their genetic traits, which control the anadromous ecotype, are therefore predominant in most populations.

Alternate life history ecotypes within a given watershed are typically more closely related to each other than to their life history stage equivalents in other watersheds (Nielsen and Fountain 1999; Docker and Heath 2003; Narum et al. 2004; Olsen et al. 2006; McPhee et al. 2007; Leitwein et al. 2017). These close genetic relationships indicate some degree of gene flow between sympatric life history forms of *O. mykiss* (Olsen et al. 2006; McPhee et al. 2007; Heath et al. 2008), although the level of gene flow is dependent on environmental, physiological, and genetic factors, such as watershed size and degree of reproductive isolation between life

history forms (Heath et al. 2008). Regardless, the close genetic relationships between sympatric populations of steelhead and Rainbow Trout suggest that the populations interbreed and that close relatives, including full siblings, may express alternative ecotypes (or other life-history variation, e.g., adfluvial or lagoon migration). Therefore, managing individual fish with different life histories separately is biologically unjustified, and the two life history variants should be considered a single population when found coexisting in streams (McPhee et al. 2007). Additionally, freshwater resident populations can retain alleles associated with anadromy (Nielsen and Fountain 1999; Phillis et al. 2016; Apgar et al. 2017) and can contribute to the viability of anadromous *O. mykiss* populations.

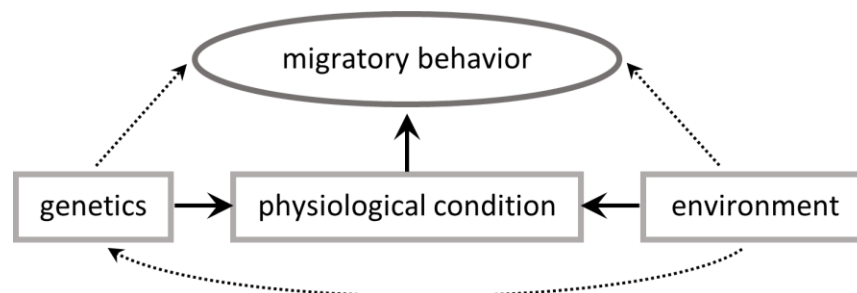


Figure 4. Schematic of indirect genetic control of migratory behavior. Genetic variation and the environment influence physiology, which then impacts migratory behavior (adapted from Kelson et al. 2020).

2.5.6 Above-Barrier vs. Below-Barrier Genetic Relationships

Studies have shown that populations of *O. mykiss*, above and below barriers within the same drainage, are closely related to one another (Heath et al. 2008; Clemento et al. 2009; Pearse et al. 2009; Leitwein et al. 2017; Fraik et al. 2021). Clemento et al. (2009) used microsatellite data to evaluate steelhead population structure above and below barriers in southern California streams and determined that populations separated by barriers are typically more closely related to each other than to populations in adjacent watersheds, consistent with many previous barrier studies. This relationship had strong bootstrap support, especially for natural-origin steelhead populations. For example, populations from the Santa Clara River formed a monophyletic lineage on the unrooted neighbor-joining tree constructed from samples taken in five main southern California watersheds (Figure 5).

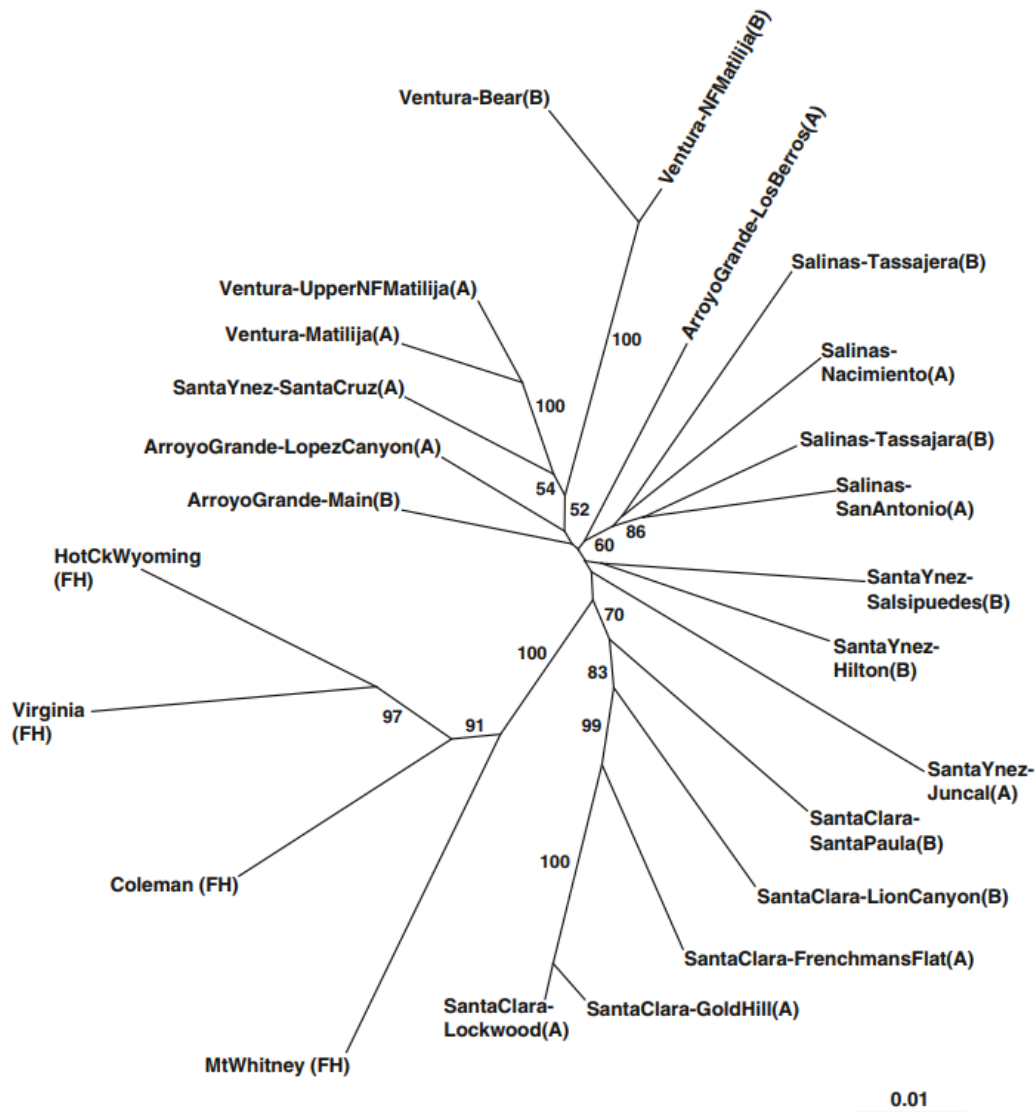


Figure 5. Unrooted neighbor-joining dendrogram showing chord distances between 24 sampled naturally spawning populations both above and below barriers, denoted with A and B, respectively. Strains of Rainbow Trout from Fillmore Hatchery used for regional stocking are indicated with FH. Numbers associated with branches indicate percentage >50% of the 10,000 bootstrap replications in which the branch appeared (from Clemento et al. 2009).

Fraik et al. (2021) recently studied patterns of genetic diversity both before and after dam removal on the Elwha River (in Washington state) and determined that populations separated by natural barriers had greater genetic differentiation than those separated by long-standing dams. Following the removal of major artificial dams on the Elwha, they also detected admixture of above- and below-dam lineages and recolonization of upstream areas by steelhead.

While many fish populations separated by barriers within the same watershed have been shown to be closely related (Heath et al. 2008; Clemento et al. 2009; Pearse et al. 2009; Leitwein et al. 2017), major barriers to anadromy, both natural and artificial, have been found to prevent gene flow between populations upstream and downstream of the obstruction (Pearse et al. 2009; Abadía-Cardoso et al. 2019; Fraik et al. 2021). Multiple studies have demonstrated that there is often a discrepancy between life history expression (Nielsen 1999; Pearse et al. 2009) and associated adaptive genetic variation (Leitwein et al. 2017; Phillis et al. 2016; Apgar et al. 2017; Abadía-Cardoso et al. 2019) across major fish passage barriers. In a number of California watersheds, *O. mykiss* populations above major barriers, especially permanent artificial barriers, have shown decreased anadromous allelic frequency when compared with the population below (Leitwein et al. 2017; Phillis et al. 2016; Abadía-Cardoso et al. 2019). Likewise, in San Francisco Bay Area study streams, most above-dam *O. mykiss* populations, have significantly lower frequencies of the anadromous *Omy5* genotype than populations downstream of barriers (Leitwein et al. 2017). Abadía-Cardoso et al. (2019) also found decreased frequencies of anadromous alleles above barrier dams in the American River drainage.

Reduced migratory allelic frequency in fish populations above longstanding natural barriers is the expected condition since the population is fragmented and gene flow is unidirectional. Fish can almost always move, either passively or volitionally, over barriers in the downstream direction, potentially contributing genes to the downstream population. Those that inhabit waters upstream of permanent barriers either assume a resident life history or must migrate downstream, taking migratory alleles with them and further reducing their frequency in the upstream population (Leitwein et al. 2017). It is also important to note that some above-barrier fish populations exhibit less genetic diversity (lower heterozygosity) than their below-barrier counterparts within the same drainage (Martínez et al. 2011). In some cases, however, fish carrying anadromous alleles may not be able to move downstream over barriers, especially large artificial dams and other complete barriers, which may help maintain anadromous *Omy5* variants in some above-dam populations (Leitwein et al. 2017; Pearse et al. 2014). It also appears that some large, above-barrier reservoirs can act as “surrogate oceans” and may assist in the retention of anadromous genotypes and the expression of the adfluvial life history type (Leitwein et al. 2017). However, a reservoir environment imposes different selective pressures than migration to the northern Pacific Ocean, and therefore we would expect the anadromous genotype to be changed over time and eventually lose its ability to express a successful anadromous phenotype.

Apgar et al. (2017) recently investigated the effects of climate, geomorphology, and fish passage barriers on the frequency of migration-associated alleles in *O. mykiss* populations across four California steelhead federal-ESA-listed DPSs (Southern California, South-Central

California Coast, Central California Coast, and Northern California). Long-term natural barriers and artificial dams that provide no fish passage had the most pronounced negative impact on migration-associated allele frequency. Southern California DPS populations had the lowest frequency of *Omy5* haplotypes associated with anadromy of all California DPSs sampled. The Southern California DPS also exists in a number of heavily developed watersheds, with the greatest average number of partial and complete artificial barriers of the DPSs sampled. Removal of these barriers was predicted to substantially increase the frequency of anadromous alleles in southern California watersheds (Apgar et al. 2017).

2.5.7 Genetic Impacts of Historical Stocking

Clemento et al. (2009) conducted a genetic analysis using microsatellite loci to elucidate the genetic population structure of *O. mykiss* in southern California, with an emphasis on above- and below-barrier genetic relationships. Their analysis included an evaluation of genetic influences of long-standing Fillmore Hatchery stocking on naturally spawned populations in the region. In regional population structure analysis, Fillmore Hatchery Rainbow Trout strains clustered separately from all wild populations, both above and below barriers. This dispersal pattern indicates that there was no evidence of hatchery introgression with wild *O. mykiss* within the Southern SH/RT range (Clemento et al. 2009).

Abadía-Cardoso et al. (2016) used microsatellite and SNP loci to elucidate *O. mykiss* ancestry at the extreme southern extent of its range. Most samples collected for this study were from populations above anadromous barriers, which mostly precludes any analysis of Southern SH/RT genetic lineage pertinent to the proposed CESA listing unit, which includes only below barrier *O. mykiss*. The evaluated southern California *O. mykiss* populations had lower genetic diversity than other California steelhead populations and, genetically, most resembled hatchery Rainbow Trout. The most northern of the evaluated populations of the Southern SH/RT exist in the Santa Maria, Santa Ynez, and Santa Clara rivers, all of which exhibit genetics associated with the native coastal steelhead lineage, matching the results of Clemento et al. (2009) and Nielsen et al. (1997). Many of the more southern populations have been almost entirely replaced by hatchery produced Rainbow Trout, and only select populations in the San Luis Rey River, Coldwater Canyon Creek, the Santa Ana River watershed, and the San Gabriel River were found to have significant native coastal steelhead ancestry. Based upon these findings, the authors recommended that conservation planning focus on these populations for the preservation of native coastal lineages. These populations also had shared ancestry with the native coastal *O. m. nelsoni* from Baja California. Secondly, they identified Bear Creek and Devil's Canyon Creek as high value populations with remnant, detectable levels of native ancestry. Also, in contrast to northern coastal steelhead populations, southern California *O. mykiss* showed low allelic frequency correlated with anadromy at *Omy5* loci, again consistent with extensive

introgressive hybridization with hatchery Rainbow Trout and limited opportunities to express the anadromous life history. Low genetic variation, observed in populations with predominantly native ancestry, may not allow them to endure changes in environmental conditions, particularly rapid and dramatic changes like those being driven by escalating climate change impacts to the region. Abadía-Cardosa et al (2016) further recommended a managed translocation strategy between the few remaining southern populations with native ancestry to help slow the erosion of native genetic diversity. They found a high variability in the frequency of alleles associated with anadromy, suggesting that many populations of Southern RT/SH may maintain the capability to express the anadromous phenotype.

Nuetzel et al (2019) examined population genetic structure of *O. mykiss* populations in the Santa Monica Mountains BPG using a set of SNP markers. Specifically, they conducted genetic analyses of *O. mykiss* from Topanga, Malibu and Arroyo Sequit creeks and compared SNP data to the existing data from the Abadía-Cardosa et al (2016) study, including Omy5 genetic marker data. Their results indicate that Malibu Creek trout are almost entirely of native ancestry. The analysis of Topanga Creek trout was more complex, suggesting that Topanga Creek is a predominantly unique native population with some introgressive hybridization with hatchery Rainbow Trout. The authors did not have a sufficient sample size from Arroyo Sequit Creek to draw meaningful inferences about the ancestry of that population. Both Malibu and Topanga creeks were also found to have relatively high frequencies of the anadromous Omy5 alleles. Together, both of these populations can be a valuable genetic resource for recovery of southern California native coastal *O. mykiss*.

3. ASSESSMENT OF PROPOSED CESA LISTING UNIT

The Commission has authority to list species or subspecies as endangered or threatened under CESA (Fish and G. Code, §§ 2062, 2067). The Legislature left to the Department and the Commission, which are responsible for providing the best scientific information and for making listing decisions, respectively, the interpretation of what constitutes a “species or subspecies” under CESA (*Cal. Forestry Assn. v. Cal. Fish and G. Com.* (2007) 156 Cal.App.4th 1535, 1548-49). The Department has recognized that similar populations of a species can be grouped for efficient protection of bio- and genetic diversity (*id.* at 1546-47). Further, genetic structure and biodiversity in California populations are important because they foster enhanced long-term stability (*id.* at p. 1547). Diversity spreads risk and supports redundancy in the case of catastrophes, provides a range of raw materials that allow adaptation and persistence in the face of long-term environmental change, and leads to greater abundance (*ibid.*).

Courts should give a “great deal of deference” to Commission listing determinations supported by Department scientific expertise (*Central Coast Forest Assn. v. Fish & Game Com.* (2018) 18

Cal.App.5th 1191, 1198-99). Courts have held that the term “species or subspecies” includes ESUs (*id.* at 1236, citing *Cal. Forestry Assn.*, 156 Cal.App.4th at pp. 1542 and 1549). The Commission’s authority to list necessarily includes discretion to determine what constitutes a species or subspecies (*id.* at p. 1237). The Commission’s determination of which populations to list under CESA goes beyond genetics to questions of policy (*ibid.*). The Department and Commission’s determinations of what constitutes a species or subspecies under CESA are not subject to the federal ESA, regulations based on the federal ESA, or federal ESA policies adopted by NMFS or USFWS, but those sources may be informative and useful to the Department and Commission in determining what constitutes a species or subspecies under CESA.

The ESU designation has been used for previous Pacific salmon listings under CESA, including the Sacramento River Winter-run Chinook Salmon ESU (Endangered, 1989), the Central Valley Spring-run Chinook Salmon ESU (Threatened, 1999), Southern Oregon-Northern California Coast Coho Salmon ESU (Threatened, 2005), and the Central California Coast Coho Salmon ESU (Endangered, 2005). In 2022, the Commission listed northern California summer steelhead as endangered under CESA. In support of that listing, the Commission determined that the petitioned listing unit qualified as a subspecies under CESA “based on the discreteness (when compared to other ecotypes) and significance of that listing unit within the state of California” (Cal. Fish and G. Com. 2022).

3.1 DPS and ESU Criteria

The federal ESA defines “species” to include “any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature” (16 U.S.C. § 1532). In 1991, NMFS adopted its policy on how it would apply the definition of “species” to Pacific salmon stocks for listing under the ESA. Under the NMFS ESU Policy, a salmon stock is considered a DPS if it constitutes an ESU of the biological species. To be considered an ESU, the salmon stock must meet two criteria (NMFS 1991):

1. “It must be substantially reproductively isolated from other conspecific population units; and
2. It must represent an important component in the evolutionary legacy of the species.”

Generally, reproductive isolation does not have to be absolute, but it must be strong enough to permit evolutionarily important differences to accrue in different population units (NMFS 1991). The evolutionary legacy of a species refers to whether the population contributes substantially to the ecological and genetic diversity of the species as a whole (NMFS 1991).

In February 1996, USFWS and NMFS published a joint DPS policy for the purposes of ESA listings. Three elements are evaluated in a decision regarding the determination of a possible DPS as endangered or threatened under the ESA. These criteria are (NMFS 1996a):

1. “Discreteness of the population segment in relation to the remainder of the species to which it belongs;
2. The significance of the population segment to the species to which it belongs; and
3. The population segment’s conservation status in relation to the [federal ESA’s] standards for listing (i.e., is the population segment, when treated as if it were a species, endangered or threatened [under the federal ESA’s standards]).”

A population segment is discrete if it meets either of two conditions specified in the DPS Policy (NMFS 1996a):

1. “It is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors. Quantitative measures of genetic or morphological discontinuity may provide evidence of this separation.
2. It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of Section 4(a)(1)(D) of the [ESA].”

If a population segment is determined to be discrete based on physical, physiological, ecological, or behavioral factors, its significance and status are then evaluated based on several characteristics specified in the joint DPS Policy. These include, but are not limited to (NMFS 1996a):

1. “Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon.
2. Evidence that loss of the discrete population segment would result in a significant gap in the range of a taxon.
3. Evidence that the discrete population segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historic range.
4. Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics.”

Under the DPS Policy, if a population segment is found to be both discrete and significant, its status is then evaluated for listing based on listing factors established by the federal ESA.

3.2 Southern SH/RT Evaluation under the Joint DPS Policy

The proposed listing unit (Southern SH/RT) in the Petition is “all *O. mykiss* below manmade and natural complete barriers to anadromy, including anadromous and resident life histories, from and including the Santa Maria River (San Luis Obispo and Santa Barbara counties) to the U.S.-Mexico Border.” Southern SH/RT is a subtaxon of the species *O. mykiss*. The anadromous life history of Southern SH/RT is not markedly separate from the non-anadromous life history of Southern SH/RT below manmade and natural barriers to anadromy. To determine whether Southern SH/RT is a subspecies for the purposes of CESA listing, the Department used the joint DPS Policy to determine whether Southern SH/RT is a DPS. The Department evaluated the proposed listing unit by applying the first (discreteness) and second (significance) criteria of the joint DPS Policy but not the third criterion (the population segment’s conservation status in relation to the federal ESA’s standards). The Department did not apply the third criterion because after using the discreteness and significance criteria to determine whether Southern SH/RT is a DPS and hence a subspecies for purposes of CESA, the Department will assess the listing unit’s status in relation to CESA’s standards rather than the federal ESA’s standards.

In 2006 NMFS concluded that application of the joint DPS Policy to West Coast *O. mykiss*, including the Southern California Steelhead DPS, was logical, reasonable, and appropriate (NMFS 2006). Further, NMFS concluded that use of the ESU Policy, which was originally intended for Pacific salmon, should not continue to be applied to *O. mykiss*, a type of salmonid with characteristics not typically exhibited by Pacific salmon (NMFS 2006). The Department finds that the application of the discreteness and significance DPS criteria from the DPS Policy is appropriate, logical, and reasonable for identifying whether Southern SH/RT is a subspecies for purposes of CESA because the taxon exhibits characteristics that are not typically exhibited by other Pacific salmonids, for which the ESU policy was developed.

3.2.1 Discreteness

Markedly Separate: Yes. The Department considers Southern SH/RT to be markedly separate from other populations of the taxon along the West Coast of North America based on unique regional biogeography, ecology, physiology, and behavior of Southern SH/RT. Point Conception in southern California is a well-studied biogeographic boundary that separates different physical oceanographic processes and the abundance and distribution of many marine species (Horn and Allen 1978; Horn et al. 2006; Miller 2023). The coastal areas north of Point Conception have cooler water temperatures, stronger upwelling, high nutrient concentrations, and the coastline is generally rocky. Within the southern California Bight, water temperatures are warmer, upwelling is weaker, and the coastline is typically sandy. While intraspecific genetic breaks do not always coincide with biogeographic boundaries near Point Conception (Burton

1998), the Department maintains that the DPS standards for discreteness do not require absolute separation of a DPS from other members of this species, because this can rarely be demonstrated in nature for any population of organisms (NMFS 1996a).

The life history of Southern SH/RT relies more heavily on seasonal precipitation than populations of the same taxon occurring farther north (Busby et al. 1996). Because average precipitation is substantially lower and more variable and erratic in southern California than regions to the north, Southern SH/RT are more frequently exposed to adverse environmental conditions in marginal habitats (i.e., warmer water temperatures, droughts, floods, wildfire) (Busby et al. 1996). Morphologically, anadromous forms of Southern SH/RT are typically longer in length and more streamlined in shape than more northern populations to enable passage through southern California's erratic and low streamflow watersheds (Moyle et al. 2017).

The Department also considers Southern SH/RT to be markedly separate from above-barrier populations of *O. mykiss* in watersheds that are within the geographic scope of the proposed listing unit, because these above-barrier populations do not contribute substantially to the below-barrier populations of Southern SH/RT. Despite several studies showing that above and below barrier *O. mykiss* populations within the same drainage are closely related, major artificial and natural barriers to anadromy prevent migration and gene flow between these populations (Heath et al. 2008; Clemente et al. 2009; Pearse et al. 2009; Abadia-Cardoso et al. 2019; Fraik et al. 2021). Disconnection between populations is further illustrated by the fact that a number of above-barrier *O. mykiss* populations exhibit reduced migratory allelic frequency compared to below-barrier Southern SH/RT. This is particularly true for *O. mykiss* populations in southern California, where long-standing natural and artificial barriers that impede fish passage have led to a lower frequency of migratory alleles associated with anadromy than in populations further north (Apgar et al. 2017).

International Border: No.

3.2.2 Significance

Unique Ecological Setting: Yes. The range of Southern SH/RT represents one of the southernmost regions of the taxon's entire West Coast Range of North America. Within this range, the watersheds that occur south of the Santa Monica Mountains have a semi-arid climate that is characterized by low precipitation, high evaporation rates, and hot and dry summers (CDFW 2021d). This climate type represents a unique ecological setting for Southern SH/RT relative to most *O. mykiss* populations along the West Coast of North America that occur in Mediterranean climates characterized by summer fog.

The ecological setting for Southern SH/RT is characterized by significant urbanization which is unique among many other federally listed steelhead DPSs that occur in coastal regions of California that are not as highly developed or populated. For example, approximately 22 million people reside in the southern California counties of Santa Barbara, Ventura, Los Angeles, Orange, San Bernardino, Imperial, and San Diego, whereas the population in the South-Central coast counties of Santa Cruz, Santa Clara, Monterey, San Benito, and San Luis Obispo is approximately 2.8 million people (NMFS 2012a; NMFS 2013). Furthermore, almost all Southern SH/RT-bearing watersheds contain dams and water diversions that have blocked access to most historic spawning and rearing habitats. Of the four DPSs sampled by Apgar et al. (2017), the Southern California Steelhead DPS contained the highest average number of partial anthropogenic barriers per watershed ($n = 4.7$) and the highest total number of complete anthropogenic barriers ($n = 8$). For context, the neighboring, and more northern South-Central Coast DPS contains a significantly lower average number of partial anthropogenic barriers per watershed ($n = 1.6$) and complete anthropogenic barriers ($n = 1$). Moreover, nearly all estuary and lagoon ecosystems in southern California have been severely degraded, thereby limiting the ability of juvenile Southern SH/RT to utilize these critical nursery habitats (Moyle et al. 2017). While these anthropogenic threats are not necessarily unique to the southern California coastal area, the region's highly variable and erratic hydrologic cycle and relatively arid climate, combined with the impacts of climate change, make Southern SH/RT increasingly vulnerable to extinction and less resilient to disturbance events and catastrophic events such as major wildfires and floods.

Gap in Range: Yes. The Department maintains that the loss of Southern SH/RT would result in a significant truncation of the southern range of the taxon along the West Coast of North America. The range of Southern SH/RT encompasses approximately 12,700 square miles with 25,700 miles of streams (NMFS 2012a).

Only Surviving Natural Occurrence: No.

Markedly Different Genetic Characteristics: No. Individuals from populations of Southern SH/RT have been shown to not be genetically isolated from populations of *O. mykiss* in the south-central California coast (Clemento et al. 2009). Evidence of straying has been documented in steelhead in central California (Donohue et al. 2021), and genetic population structure analyses suggest that there was historical exchange of genetic information between coastal populations (Garza et al. 2014). Although many steelhead populations can be partially isolated, at least a small amount of exchange between different populations of steelhead is to be expected due to natural straying. This connectivity results in a level of genetic similarity, which is more pronounced between neighboring populations, and prevents most populations from being completely isolated (Bjorkstedt et al. 2005; Garza et al. 2014; Arciniega et al. 2016).

Nonetheless, allozymes, mitochondrial DNA, and microsatellites have uncovered significant and unique genetic diversity in southern California steelhead, including traits not found in more northern populations. Busby et al. (1996) noted that a mitochondrial DNA type exists in *O. mykiss* populations between the Santa Ynez River and Malibu Creek that is rare in populations to the north, while samples from Santa Barbara County were found to be the most genetically unique of any wild coastal steelhead populations analyzed. Conservation of both neutral and adaptive genetic diversity, such genetic variation associated with migratory life history, is crucial in maintaining the ability of *O. mykiss* populations to adapt to altered environments. Given that Southern SH/RT populations have the lowest frequencies of anadromous genotypes, it is critical to preserve this genetic variation and ensure no more of it is lost.

3.2.3 Conclusion

Southern SH/RT satisfies the first (discreteness) and second (significance) criteria of the joint DPS Policy: i.e., Southern SH/RT is markedly separate and biologically significant to the taxon to which it belongs. Accordingly, the Department concludes that Southern SH/RT is a DPS and hence a subspecies for the purposes of CESA listing.

4. POPULATION TRENDS AND ABUNDANCE

4.1 Structure and Function of Viable Salmonid Populations

In this review, we use the definition of “population” from McElhany et al. (2000): “An independent population is a group of fish of the same species that spawns in a particular lake or stream (or portion thereof) at a particular season and which, to a substantial degree, does not interbreed with fish from any other group spawning in a different place, or in the same place at a different season.” In other words, a population as defined by McElhany et al. (2000) is a group of fish that experiences a substantial degree of reproductive isolation.

Steelhead have strong fidelity to their natal stream, which can lead to substantial reproductive isolation and, as a result, create local adaptation within somewhat isolated populations (Waples et al. 2008). Isolation can expose these local populations to varying degrees of genetic drift as well as different environmental pressures that ultimately lead to the development of genetic and phenotypic differences. Although many steelhead populations can be partially isolated, at least a small amount of exchange between different populations of steelhead is to be expected due to natural straying. This connectivity results in a level of genetic similarity, which is more pronounced between neighboring populations, and prevents most populations from being completely isolated (Bjorkstedt et al. 2005; Garza et al. 2014; Arciniega et al. 2016).

The concept of viable salmonid populations was introduced by McElhany et al. (2000). A viable salmonid population is defined as, “an independent population of any Pacific salmonid (genus *Oncorhynchus*) that has negligible risk of extinction due to threats from demographic variation, local environmental variation, and genetic diversity changes over a 100-year time frame,” and an independent population is defined as, “any collection of one or more local breeding groups whose population dynamics or extinction risk over a 100-year time period are not substantially altered by exchanges of individuals with other populations.”

McElhany et al. (2000) introduced four criteria for assessing viability of salmonid populations: abundance, productivity, population spatial structure, and diversity. These parameters form the foundation for evaluating population viability because they serve as reasonable predictors of extinction risk, reflect general processes important to all populations of species, and are measurable. Abundance is a key parameter because smaller populations are at greater risk of extinction than larger populations. Productivity, which is associated with abundance, serves as an indicator of population growth rate either over an entire life cycle or stage-specific life-history stage. Population spatial structure represents the distribution of individuals in habitats they use throughout their life cycle, as well as the processes that generate that distribution. Spatial structure often reflects the amount of suitable habitat available for a population as well as demographic stability and the level of straying among habitats. Diversity represents variation in traits such as anadromy, run-timing, and spawning behavior and timing. Typically, a more diverse population is more likely to contain individuals that will survive and reproduce in the face of environmental variation (McElhany et al. 2000). In this chapter, we evaluate, to the best of our ability, these four criteria for Southern SH/RT populations.

4.2 Sources of Information

We reviewed many sources of information for this Status Review, including primary research and literature review articles, the CESA listing petition, previous federal status reviews, recovery plans, viability assessments, Department reports and documents, annual reports from ongoing Southern SH/RT monitoring efforts, and historical reports. Agency staff with knowledge of watersheds supporting Southern SH/RT were also consulted for information.

Data limitations and uncertainties associated with historical accounts for Southern SH/RT limits our ability to understand their complete historical abundance and distribution in their range. The majority of available historical data are in reports, technical memos, and other documents that have not undergone a formal peer-review process. These types of historical sources are not necessarily at a high level of scientific rigor and have not been subject to peer review, but they represent the best information available at the time of this review regarding the historical distribution and abundance of Southern SH/RT populations.

Multiple data sources were used to evaluate viability metrics of Southern SH/RT populations. These data are mostly derived from monitoring reports from several single-basin annual survey efforts. For example, data for the Santa Ynez River population was sourced from monitoring reports developed by the Cachuma Operations and Maintenance Board (COMB). Data for the Ventura River was sourced from annual monitoring reports produced by Casitas Municipal Water District (CMWD), and data contained in Booth (2016) for the United Water Conservation District (UWCD) was used for the Santa Clara River population (See Appendices A – D for full data sources). Although data from these monitoring reports represent the best available scientific information in many southern California watersheds, the data may be derived from different monitoring approaches and designs, contain detection bias, and vary in the level of monitoring effort through time and geographic areas. These constraints may limit the power of statistical analyses to assess trends in viability criteria. Therefore, the results of the analyses conducted in subsequent portions of this chapter should be interpreted in the context of these limitations.

Dagit et al. (2020) describes the occurrences of adult steelhead from 1994-2018 and was also used as a source of peer-reviewed information to provide insight into the abundance trends of Southern SH/RT, particularly for the basins south of Los Angeles where historically no monitoring of steelhead occurred. Additional information on the data sources used in this chapter can be found in Appendices A - D. and Dagit et al. (2020).

4.3 Historical and Current Distribution

This section discusses the historical and current distribution of Southern SH/RT within their range. The section is structured on the five BPGs, which are a federal delineation based on a suite of environmental conditions (e.g., hydrology, local climate, geography) and watershed characteristics (i.e., large inland or short coastal streams) (NMFS 2012a). Separate watersheds within each BPG are considered to support individual populations of southern SH and RT; therefore, single BPGs encompass multiple watersheds and populations (Figure 6). Additional information on southern SH/RT distribution in watersheds not included in this section can be found in Good et al. (2005), Becker and Reining (2008) and Titus et al. (2010). In general, estimates of historical population abundance are based on sparse data and assumptions that are plausible but have yet to be adequately verified or tested. While the following historical estimates are likely biased either upward or downward, the examination of historical records of adult run size in southern California show consistent patterns of abundance that are at least two or three orders of magnitude greater in size than in recent years.

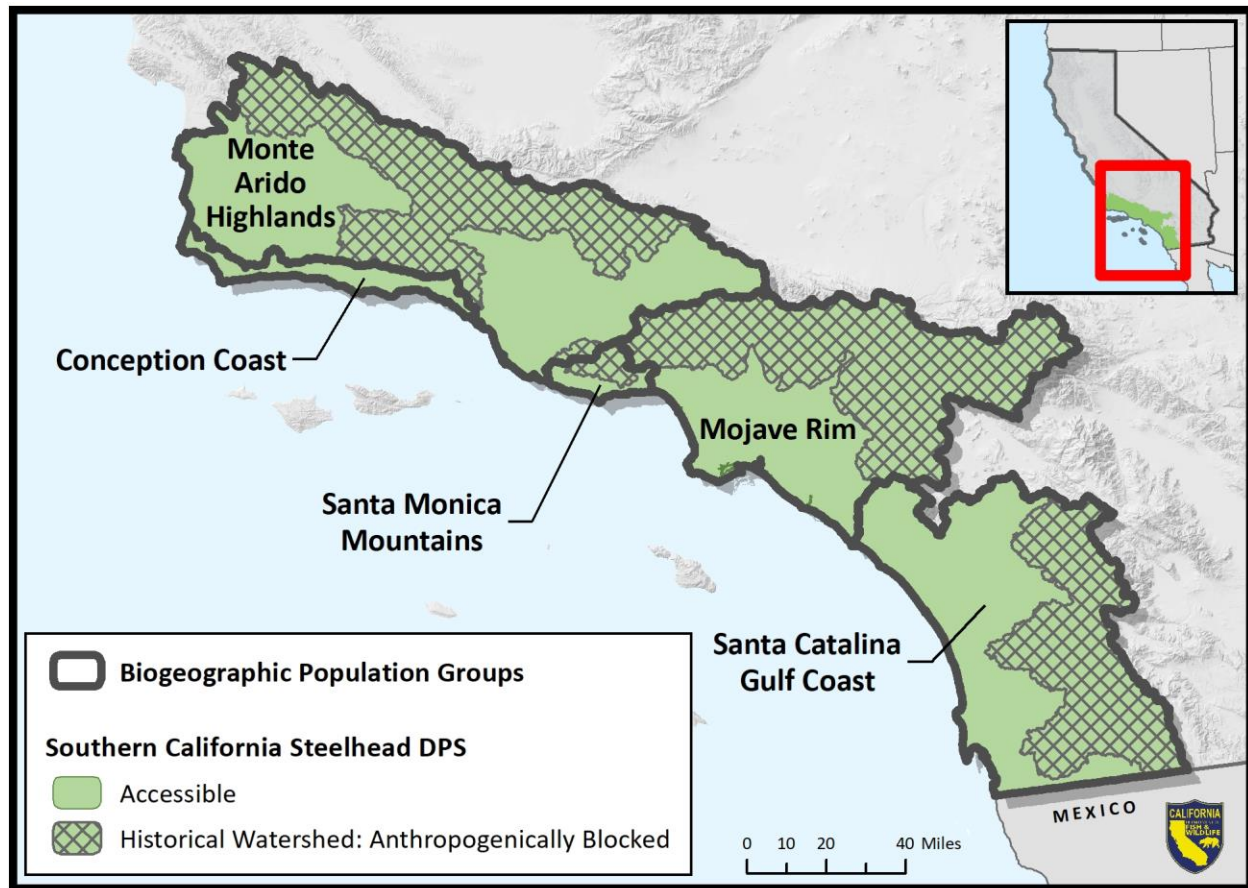


Figure 6. Map of the current and historical distribution of Southern SH/RT. BPGs represented are the Monte Arido Highlands, Conception Coast, Santa Monica Mountains, Mojave Rim, and Santa Catalina Gulf Coast.

4.3.1 Monte Arido Highlands Biogeographic Population Group

The Monte Arido Highlands BPG includes four watersheds spanning San Luis Obispo, Santa Barbara, Ventura, and northern Los Angeles counties draining the west side of the Transverse Range and terminating at the Pacific Ocean (NMFS 2012a; Figure 7). Inland stretches of these watersheds are high in elevation and mountainous, but otherwise the watersheds contain different geographic features. Watersheds in this BPG are susceptible to “flashy” flows with seasonal storms and can also dry during the summer even in mainstem reaches. Perennial flows are mainly found in the upper reaches of tributaries that still retain groundwater connection (NMFS 2012a).

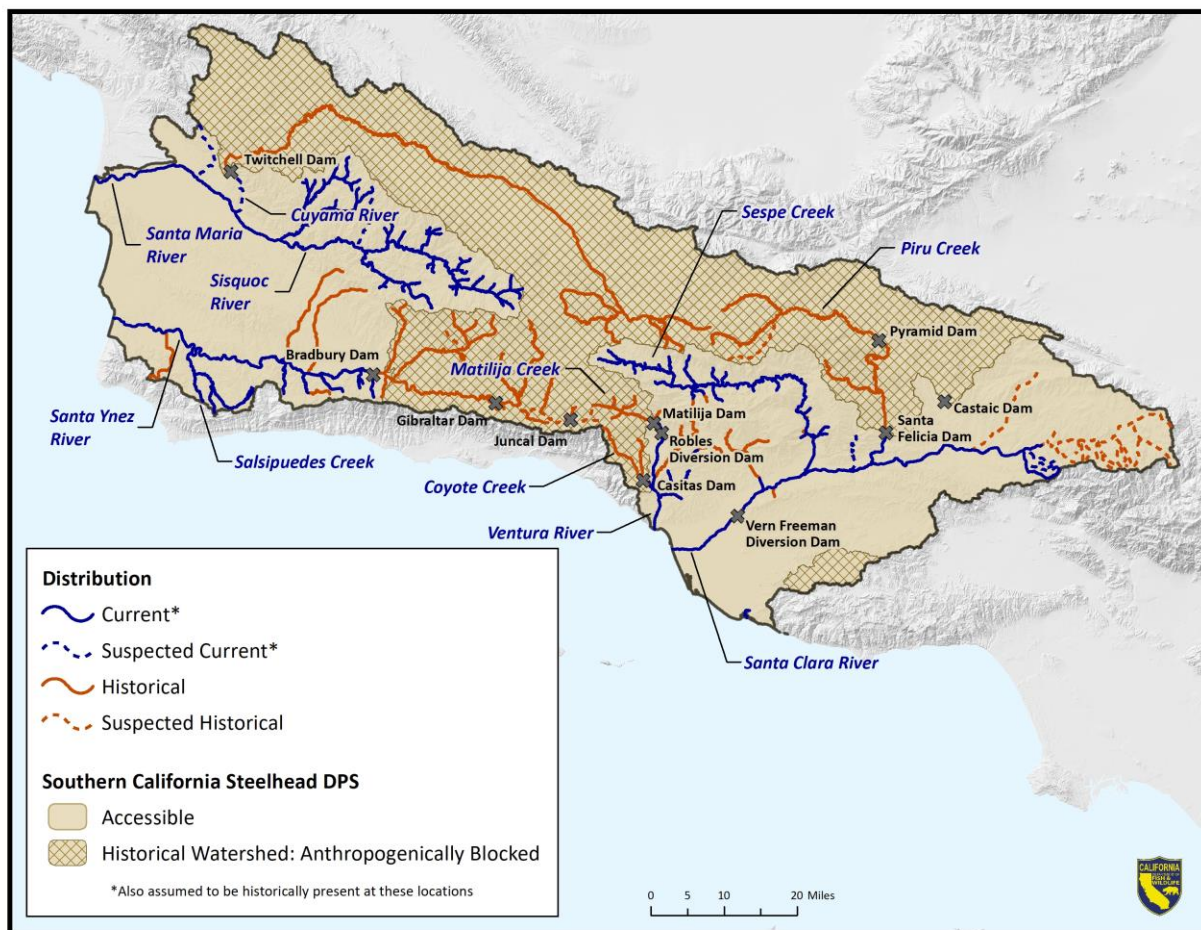


Figure 7. Map of the Monte Arido Highlands BPG depicting known and suspected current and historical distribution.

4.3.1.1 Santa Maria River

The Santa Maria River runs from the confluence of the Cuyama and Sisquoc rivers to the ocean and encompasses 1,790 square miles of watershed (Becker and Reining 2008). Historically, the Santa Maria River served mainly as a corridor for steelhead migrating to and emigrating from the Cuyama and Sisquoc rivers, rather than as habitat for spawning and rearing (Titus et al. 2010).

Hatchery stocking of *O. mykiss* occurred in the early 1930s in the Sisquoc and Cuyama watersheds (Titus et al. 2010). However, local newspaper records from the late 1800's reported abundant harvests of *O. mykiss* in the Sisquoc River watershed well before hatchery stocking occurred (Camm Swift, Emeritus, Section of Fishes, Natural History Museum of Los Angeles County, personal communication). In the early to mid-1940s, juvenile steelhead from the Santa Ynez River were rescued and translocated to the Santa Maria River. Tributaries of the Cuyama

River were stocked with Rainbow Trout in the 1940s to support recreational fishing; however, it is unknown if there was a historical run of anadromous Southern SH/RT in the Cuyama River tributaries (Titus et al. 2010). Starting in 1950, there was essentially no steelhead fishery for at least a decade (Titus et al. 2010).

The Sisquoc River had a robust population of resident *O. mykiss* in 1959 (Becker and Reining 2008) and fish were seen in smaller numbers in 1964 (Titus et al. 2010). Southern SH/RT of multiple age classes were also observed in the upper river during the 1990s (Becker and Reining 2008). In 2005, substantial numbers of young-of-the-year (YOY) *O. mykiss*, as well as some older age classes, were observed in the upper Sisquoc watershed during a population survey (Stoecker 2005).

Other smaller tributaries in the Santa Maria watershed, mostly tributaries of the Sisquoc and Cuyama rivers, have had limited historical and present *O. mykiss* observations from surveys, although some anecdotal sightings have occurred (Becker and Reining 2008). The streams include Deal Canyon Creek, Reyes Creek, Beartrap Creek, Tepusquet Creek, La Brea Creek, North Fork La Brea Creek, Manzana Creek, Davy Brown Creek, Munch Canyon Creek, Sunset Valley Creek, Fish Creek, Abel Canyon Creek, South Fork Sisquoc River, White Ledge Canyon Creek, Rattlesnake Canyon Creek, and Big Pine Canyon Creek. Some of these *O. mykiss* observations were made in tributaries of the Cuyama River post-dam construction (Becker and Reining 2008); however, it is possible that anadromous Southern SH/RT were able to access and inhabit these areas historically. Notably, many of these small tributaries were stocked with thousands of hatchery-raised *O. mykiss* in the mid-1900s for fishery supplementation (Titus et al. 2010).

Twitchell Dam was built on the Cuyama River in the late 1950s, almost 8 miles upstream from the confluence with the Santa Maria River. The dam currently impacts hydrologic function of the Santa Maria system by increasing the frequency of “false positive” migration flows in the Sisquoc River, reducing the frequency of downstream passable migration conditions, increasing the number of days with upstream passable flows that are not followed by additional days of passable flows, and reducing the frequency of long-duration migration flows (Becker and Reining 2008; Stillwater Sciences 2012). Twitchell Dam is a complete barrier to anadromy, and historically, water releases have not been regulated to provide instream flows for upstream and/or downstream steelhead migration in the Santa Maria River during the winter and spring migration periods (Stoecker 2005). Following construction of the dam, the Santa Maria and Cuyama rivers continue to have intermittent flows (Becker and Reining 2008). Currently, the lower mainstem of the Santa Maria River, which serves as a migration corridor for Southern SH/RT, is dry most of the year in most years due to managed aquifer recharge in the Santa Maria Valley (NMFS 2012a). The U.S. Court of Appeals for the Ninth Circuit recently held that

under the legislation authorizing construction of Twitchell Dam, the U.S. Bureau of Reclamation and the Santa Maria Water District have discretion to manage and operate Twitchell Dam for the purpose of preventing take of Southern California Steelhead under the federal ESA, which may include adjusting water discharges to support their migration and reproduction (*San Luis Obispo Coastkeeper v. Santa Maria Valley Water Conservation Dist.* (9th Cir. 2022) 49 F.4th 1242, 1244). The case was remanded to the U.S. District Court for the Central District of California (*id.* at 1250), which adopted a pilot project involving supplemental flow releases, to be implemented while consultation under the federal ESA is conducted (*San Luis Obispo Coastkeeper et al. v. Santa Maria Valley Water Conservation Dist. et al.*, Case No. 2:19-CV-08696-AB-JPR, Dkt. No. 167 (October 12, 2023)).

4.3.1.2 Santa Ynez River

The Santa Ynez River is a major watershed spanning approximately 900 square miles and 90 river miles (Becker and Reining 2008). The river is thought to have supported the largest anadromous Southern SH/RT run (Titus et al. 2010). The earliest records of Southern SH/RT in the Santa Ynez occurred in the late 1800s prior to any stocking of the river with hatchery trout (Alagona et al. 2012). Upstream migration of Southern SH/RT past river km 116 was impeded in 1920 resulting from the construction of Gibraltar Dam (Titus et al. 2010). The reservoir supported landlocked steelhead following dam construction and was stocked in the 1930s with hatchery *O. mykiss* as well as steelhead rescued from the Santa Ynez River in 1939, 1940, and 1944 (Titus et al. 2010).

Upstream migration typically occurred from December to March following precipitation events. Southern SH/RT were seen spawning in all tributaries as well as the mainstem below Gibraltar Dam during the spring in the mid-1930s, though flow was observed to limit suitable spawning habitat (Titus et al. 2010). Most spawning in the Santa Ynez River occurred in the upper reaches between Buellton and Gibraltar Dam as well as the tributaries to the mainstem such as Alisal, Santa Cota, Cachuma, Tequepis Canyon, and Santa Cruz creeks. Fish rescues were required during the summer due to intermittent flows and drying of downstream tributary areas as well as the mainstem (DFG 1944).

Tens of thousands of hatchery *O. mykiss* were stocked in Gibraltar Reservoir in the 1930s, and over 100,000 hatchery-reared juvenile steelhead were planted in the Santa Ynez River from 1930-1935. In the 1940s, about 2.5 million juvenile Southern SH/RT were translocated from various areas of the watershed to the lower river (DFG 1944). An approximate run size of at least 13,000 spawners was inferred by a Department staff member based on comparisons with Benbow Dam counts on the South Fork Eel River, California in the 1930s and 1940s (Becker and Reining 2008; Titus et al. 2010). However, it is possible that the Santa Ynez steelhead

population may have increased during this period due to ongoing rescue operations that resulted in lower mean mortality rates during the early to mid-1940s (Good et al. 2005). Nonetheless, these estimates may underestimate historical abundance because they were produced 24 years after a significant portion of spawning and rearing habitat had been blocked by Gibraltar Dam.

Construction of Bradbury Dam, originally named Cachuma Dam, downstream of Gibraltar Dam was finished in 1953. Bradbury Dam forms the Lake Cachuma reservoir, blocks Southern SH/RT access to upstream habitat, and alters natural flow regimes and sediment dynamics (Becker and Reining 2008; Titus et al. 2010). Even before the dam was built, the lack of precipitation limited upstream migration due to the sandbar at the mouth of the river remaining intact (Titus et al. 2010). Steelhead run size declined significantly after 1946 and only small numbers were seen in the stream reaches below Bradbury Dam in following decades (Titus et al. 2010). Anadromous Southern SH/RT were effectively extirpated by 1975 due to lack of flows below Bradbury Dam especially during summer months, though steelhead have occasionally been observed over the past few decades (Becker and Reining 2008).

Recently, Reclamation's permit to operate releases from Bradbury Dam was modified to require releases from the dam for purposes of protecting fishery resources in accordance with the 2000 NMFS Biological Opinion during wetter years. This modification also included additional measures to benefit Southern SH/RT, including opportunities to provide fish passage above and below Bradbury Dam, measures to reduce the impacts of predation, and restoration of stream and bankside habitat (SWRCB 2019).

Department staff have monitored steelhead in Salsipuedes Creek, Hilton Creek, and the mainstem Santa Ynez River and have found that most years can support a small steelhead run. However, zero adult steelhead have been found in the Santa Ynez River since 2012 (Boughton et al. 2022a). COMB has conducted uncalibrated, single pass snorkel surveys each year since the 1990s at multiple index sites to determine *O. mykiss* densities in the Santa Ynez River. Until 2012, fish densities were consistent but declined sharply in the following years due to drought conditions (Boughton et al. 2022a). The past few years have seen numbers rebound somewhat in response to wetter conditions. Similar trends were observed in the migrant traps on Hilton and Salsipuedes creeks and the mainstem Santa Ynez River, which have been in operation since 2001 (COMB 2022).

4.3.1.3 Ventura River

The Ventura River watershed encompasses 228 square miles and 16.5 stream miles (Becker and Reining 2008). Matilija Creek and North Fork Matilija Creek intersect to form the headwaters of the Ventura River. Multiple large storage and diversion dams occur in this watershed, altering

the natural flow regime and causing negative impacts to Southern SH/RT habitat quantity and quality. About 2 miles downstream of the Ventura River headwaters is the Robles Diversion Dam, which was constructed in 1958 to direct water for storage into Lake Casitas (Becker and Reining 2008; Titus et al. 2010). Both Matilija Dam on Matilija Creek and Casitas Dam on Coyote Creek, are also attributed to population declines of Southern SH/RT on the Ventura River (Titus et al. 2010).

In the 1930s, tens of thousands of juvenile *O. mykiss* were stocked in the Ventura River, as well as thousands of fish that were transplanted from rescues conducted on the Santa Ynez River (Titus et al. 2010). Department staff estimated that the Ventura watershed supported 4,000 to 5,000 steelhead spawners in 1946. In 1973, Department staff estimated a run of between 2,500 and 3,000 steelhead (Becker and Reining 2008). However, the methodologies used to make these estimates were likely based on expert opinion. Similar to the Santa Ynez River, ongoing rescues may have had a small effect on the Ventura River steelhead populations in the 1940s. By the mid-1970s, the steelhead run size was estimated at approximately 100 fish, likely due to limited suitable rearing habitat below Robles Diversion Dam (Becker and Reining 2008).

There are four key tributaries to the Ventura River that historically provided substantial suitable spawning and rearing habitat for *O. mykiss*. These tributaries were Matilija Creek, San Antonio Creek, Coyote Creek, and Santa Ana Creek (Capelli 1974). Coyote Creek likely had a strong run of steelhead with up to 500 adult returns being probable prior to construction of Casitas Dam. Currently, the few returning Southern SH/RT spawners may use the lower reaches of the 13-mile stream for spawning (Becker and Reining 2008; Titus et al. 2010). Matilija Creek, which extends for almost 15 miles from its confluence with the Ventura River, contains ideal spawning and rearing habitat. However, access to the upper reaches of the creek was impeded with the construction of Matilija Dam (Becker and Reining 2008). Before completion of the dam, it is estimated that the creek could have supported runs of 2,000 to 2,500 spawners (Becker and Reining 2008). The removal of Matilija Dam, which is an important element of the Matilija Dam Ecosystem Restoration Project, is currently in the process of environmental review. Tributaries of Matilija Creek contain high quality habitat that continue to support resident *O. mykiss* (Becker and Reining 2008). The removal of Matilija dam will allow access to about 20 miles of stream habitat for Southern SH/RT (MDERP 2022). Historical presence of steelhead in San Antonio Creek is unknown, but the stream is thought to have produced steelhead in the 1980s and 1990s (Titus et al. 2010). Santa Ana Creek was home to *O. mykiss* in the headwater reaches during the 1930s through the 1940s as well as in 1979 (Becker and Reining 2008).

Construction on the Robles Fish Passage Facility, which allows fish passage through the Robles Diversion Dam, was completed in 2006. As a requirement of their federal Biological Opinion, CMWD monitors fish migration through the facility (CMWD 2019). A downstream migrant trap

is also operated to evaluate if smolts can pass through the facility without injury (CMWD 2019). A weir trap is then used to evaluate success of smolt migration through the reach downstream of the facility (CMWD 2019). Small numbers of out-migrating smolts have been captured since operation of the weir trap began. However, during the most recent drought (2012-2017), trapping did not occur due to low flow conditions. Since 2017, zero to only a few fish have been observed per year in the vicinity of the passage facility. Presence/absence and redd surveys for *O. mykiss* have also been conducted by CMWD each year and numbers have declined substantially since the beginning of the drought (CMWD 2018).

4.3.1.4 Santa Clara River

The Santa Clara River is a major river that flows into the Pacific Ocean near Ventura, California. The watershed drains an area of approximately 1,600 square miles with 75 stream miles (Becker and Reining 2008). The historical steelhead run was estimated to be around 9,000 fish based on comparisons of habitat suitability metrics produced for the Ventura River (Moore 1980). Numerous instream water diversions have impeded anadromous migration since the 1950s (Becker and Reining 2008; Titus et al. 2010).

In 1991 UWCD built the Vern Freeman Diversion Dam across the Santa Clara River at about 10 river miles from the Pacific Ocean, near the unincorporated community of Saticoy. The Vern Freeman Diversion Dam includes a fish passage facility (Titus et al. 2010), however, in 2019 the U.S. District Court for the Central District of California issued an order that stated, in a factual summary, “the structure and operation of [Vern Freeman Diversion Dam] significantly hampers the migration of steelhead in the Santa Clara River to and from the Pacific Ocean” because it “reduces the availability of water downstream for steelhead migration” and “it is difficult for adult steelhead to successfully pass through the fish ladder” (*Wishtoyo Found., et al. v. United Water Conservation Dist.*, Case No. 2:16-CV-03869-DOC-PLA, Dkt. No. 254 (Mar. 5, 2019); see also NMFS 2012a). Operations of a downstream migrant trap at the Vern Freeman Diversion Dam began in 1993 and typically occur from January to June when flows in the river are sufficient to maintain consistent water levels at the fish trap. A total of 16 adult steelhead and 839 smolts were observed at the Vern Freeman Diversion Dam from 1993-2014 (Booth 2016).

In 2018, the U.S. District Court for the Central District of California issued a judgment in *Wishtoyo Foundation, et al. v. United Water Conservation District* finding that “[UWCD’s] operation and maintenance of Vern Freeman Dam (‘VFD’), including its operation and maintenance of the fish ladder at the VFD, and [UWCD’s] diversion of water from the VFD, constituted ‘take’ of the Distinct Population Segment of Southern California Steelhead . . . in violation of section 9 of the Endangered Species Act” (Case No. 2:16-CV-03869-DOC-PLA, Dkt. No. 248 (December 1, 2018)). In that judgment, the court issued a permanent injunction

requiring UWCD to adhere to the water diversion operating rules set forth in a 2008 NMFS Biological Opinion until such time as UWCD obtains incidental take authorization from NMFS for the maintenance and operation of the Vern Freeman Diversion Dam (*ibid.*). The injunction further requires UWCD to design, construct, and obtain certain permits and authorizations for a new fish passage facility at the Vern Freeman Diversion Dam that is reasonably likely to meet NMFS criteria as specified in the judgment (*ibid.*). In September 2023, UWCD issued a Notice of Preparation under CEQA for an environmental impact report that will identify a hardened ramp structure as the preferred alternative for the project (available at https://www.unitedwater.org/wp-content/uploads/2023/09/Notice-of-Preparation-for-EIR_September-2023.pdf). In a joint stipulation filed with the court in July 2023, the plaintiffs and UWCD jointly proposed an order for the court to sign that would require UWCD to submit complete regulatory applications in February 2024 and submit 90% engineered design plans in June 2024 (*Wishtoyo Found., et al. v. United Water Conservation Dist.*, Case No. 2:16-CV-03869-DOC-PLA, Dkt. No. 590 (July 18, 2023)).

Tributaries that intersect the Santa Clara River above the Vern Freeman Diversion Dam historically provided most of the suitable Southern SH/RT spawning and rearing habitat in the watershed. Santa Paula Creek, a tributary to the Santa Clara River, contains high quality suitable *O. mykiss* spawning and rearing habitat. The Harvey Diversion Dam is located on the lower reaches of Santa Paula Creek. While this diversion originally provided fish passage, strong flows rendered the facility irreparable in 2005 (Stoecker and Kelley 2005). More recently, the Harvey Diversion Fish Passage Remediation Project has the goal of restoring fish passage at the facility to reestablish connection to the upstream watershed on Santa Paula and Sesar creeks (California Trout 2018).

Sespe and Piru creeks are the largest tributaries of the Santa Clara River and support higher *O. mykiss* numbers than Santa Paula Creek (Stoecker and Kelley 2005). Sespe Creek contains over 198 km of habitat historically accessible to steelhead and sustains the highest relative abundance of wild *O. mykiss*. It is thought that Sespe Creek offers the highest potential for steelhead recovery because it lacks mainstem migration barriers (Stillwater Sciences 2019). However, Sespe Creek is known to dry in years with low precipitation, leading to a loss of connectivity with the Santa Clara River (Puckett and Villa 1985; Stoecker and Kelley 2005). A recent survey found high abundances of aquatic invasive species throughout most reaches of Sespe Creek downstream of its confluence with Howard Creek, which transports high abundances of invasive species from the Rose Valley Lakes (Stillwater Sciences 2019).

The Piru Creek watershed includes the Santa Felicia and Pyramid Dams. Both dams block access to upstream historical habitat on the Santa Clara River. Reservoir and dam operations also lead to unnatural and diminished flow regimes in the watershed (Moore 1980). Prior to the

construction of both dams, adult steelhead were reported to migrate up into Buck and Snowy creeks (Stoecker and Kelley 2005). Piru Creek does not provide spawning and rearing habitat to Southern SH/RT (Moore 1980); however, Aqua Blanca and Fish creeks contain suitable habitat and currently support adfluvial *O. mykiss* populations, which could be important in the future for restoring an anadromous run in this tributary (Stoecker and Kelley 2005).

Various Santa Clara tributaries, including those mentioned above, were stocked in the 1930s through 1950s with hatchery *O. mykiss* as well as those rescued from the Santa Ynez River in 1944 (Titus et al. 2010). Some minor tributaries of the Santa Clara River were also stocked but have no historical records of *O. mykiss* presence. These tributaries include Hopper Canyon, Tom, Pole, and Willard creeks (Titus et al. 2010).

4.3.2 Conception Coast Biogeographic Population Group

Many small coastal watersheds that are relatively uniform in geographic features comprise the Conception Coast BPG, which spans about 50 miles of the southern California coast (NMFS 2012a; Figure 8). Streams in this BPG run north to south and have steep slopes in the upper portions of their watersheds where there is perennial flow. Precipitation can be much higher in the upper watersheds and can lead to “flashy” flows due to the steep stream gradients (NMFS 2012a). Both the Carpinteria Creek and Gaviota Creek watersheds have been the focus of habitat restoration in recent years, as both provide high-quality spawning and rearing habitat for Southern SH/RT and have high recovery potential (NMFS 2012a).

4.3.2.1 Gaviota Creek

Gaviota Creek is about six miles in length, connecting with the Pacific Ocean just south of Las Cruces, California. Steelhead were documented in Gaviota Creek in the 1930s in the winter (Becker and Reining 2008) and multiple ages of *O. mykiss* were observed in the 1990s and early 2000s (Becker and Reining 2008). Steelhead runs in Gaviota Creek, which were historically present in most years, were likely small (Becker and Reining 2008). Livestock grazing is responsible for reductions in suitable habitat for Southern SH/RT in the watershed (Becker and Reining 2008). In recent years, periodic bankside observations conducted by the Department have observed a range of zero to a few hundred *O. mykiss* and no adult steelhead in Gaviota Creek (K. Evans, CDFW, unpublished data).

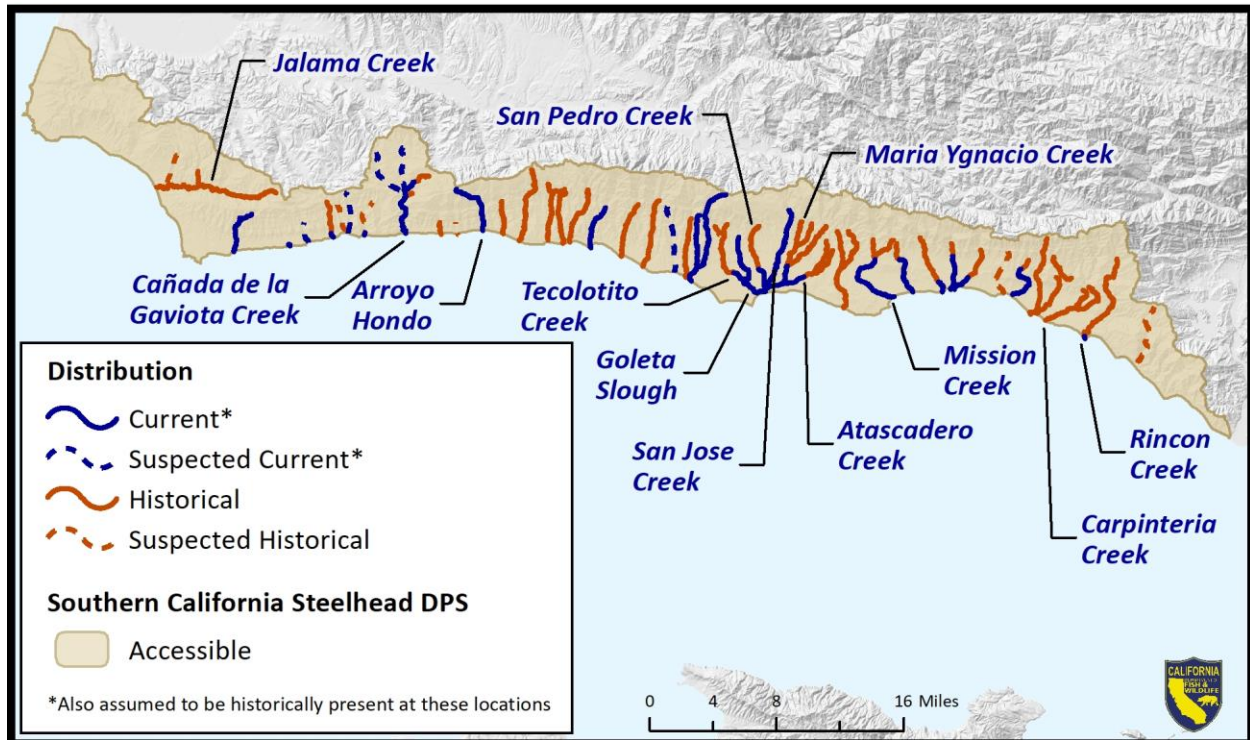


Figure 8. Map of the Conception Coast BPG depicting known and suspected current and historical distribution.

4.3.2.2 Carpinteria Creek

Carpinteria Creek is approximately 6.5 miles long and connects with the Pacific Ocean near Carpinteria, California. Southern SH/RT were observed in the watershed in 1942 (Stoecker et al. 2002) and the stream was understood to have a historical steelhead run (Becker and Reining 2008). Different life stages of *O. mykiss* were seen in the mid-1990s (Becker and Reining 2008), and many were seen in the upper watershed (Becker and Reining 2008) which is known to have suitable habitat (Becker and Reining 2008). A few *O. mykiss* of varying sizes were found in the lower watershed in 2008 (Becker and Reining 2008). In recent years, monitoring conducted by the Department from 2016-2022 have observed few if any individuals of either life-history forms (K. Evans, CDFW, unpublished data).

4.3.2.3 Other Creeks

There are many other creeks flowing into the Pacific Ocean, some of which may have supported Southern SH/RT historically (e.g., Jalama Creek), some where there have been recent observations, and others where *O. mykiss* has not been seen at all. These coastal creeks are typically no longer than 10 stream miles. In addition to Gaviota and Carpinteria creeks, other suitable streams with more recent sightings of Southern SH/RT include Arroyo Hondo Creek and

Rincon Creek (Becker and Reining 2008). Arroyo Hondo Creek contains the least number and severity of threats for Southern SH/RT in the Conception Coast BPG (NMFS 2012a).

4.3.3 Santa Monica Mountains Biogeographic Population Group

There are five watersheds in the Santa Monica Mountains BPG, the majority of which are small with geography resembling that of watersheds in the Conception Coast BPG (NMFS 2012a; Figure 9). Except for Malibu Creek, the headwaters of the streams occur prior to passing through the Santa Monica mountains. Malibu Creek is the largest watershed in the BPG (NMFS 2012a) but is similar to Topanga Creek in stream length (Becker and Reining 2008). There are two substantial anthropogenic migration barriers on Malibu Creek, Rindge Dam and Malibu Lake Dam. Rindge Dam is located a few miles upstream from the mouth and prevents access to nearly all historical Southern SH/RT habitat. The remaining three streams include Big Sycamore Canyon Creek, Arroyo Sequit, and Las Flores Canyon Creek (NMFS 2012a).

4.3.3.1 Malibu Creek

The Malibu Creek watershed encompasses about 105 square miles including 8.5 miles of stream that outflows into the Pacific Ocean at Malibu Lagoon State Beach in Santa Monica Bay (Becker and Reining 2008). Rindge Dam was constructed in 1924 about three miles upstream from the mouth (Becker and Reining 2008; Titus et al. 2010). Before the dam was built, steelhead were able to access spawning habitat in Las Virgenes and Cold creeks (Titus et al. 2010). In 1947, a steelhead run was observed when the sandbar at the mouth was manually opened. In the 1970s, steelhead were observed migrating upstream up to Rindge Dam (Becker and Reining 2008). In 1980, a Department employee counted 61 steelhead immediately downstream of Rindge Dam (Titus et al. 2010). Multiple life stages of *O. mykiss* were observed during a study conducted in the winter and spring of 1986. A total of 158 fish was reported though only one was an adult steelhead. Later in 1986 and in 1987, a handful of adult *O. mykiss* were found below Rindge Dam and a few adult *O. mykiss* were seen just below the dam in 1992 (Titus et al. 2010). The quality of spawning and rearing habitat is the best just below Rindge Dam (Titus et al. 2010), which explains the greater use of that area by juvenile *O. mykiss* (Titus et al. 2010). Stocking of hatchery Rainbow Trout occurred in 1984 at Malibu Creek State Park with additional stockings likely occurring frequently (Titus et al. 2010).

In addition to Rindge Dam and other migration barriers blocking access to historical habitat, the natural flow regime and water quality of Malibu Creek has been modified by operations of the Tapia Water Reclamation Facility (approximately 5 miles upstream from the ocean). Treated water releases from the facility sustain flows in Malibu Creek throughout the year (Titus et al. 2010). Currently, a new recycled wastewater treatment facility is being proposed that would treat effluent from the Tapia Water Reclamation Facility with the purpose of re-distributing the

water to the service area rather than releasing it back to Malibu Creek (Las Virgenes-Triunfo Joint Powers Authority 2022). The implementation of this project could lead to less streamflow in Malibu Creek as a result of the repurposing of discharged recycled water that would have previously been released to Malibu Creek.

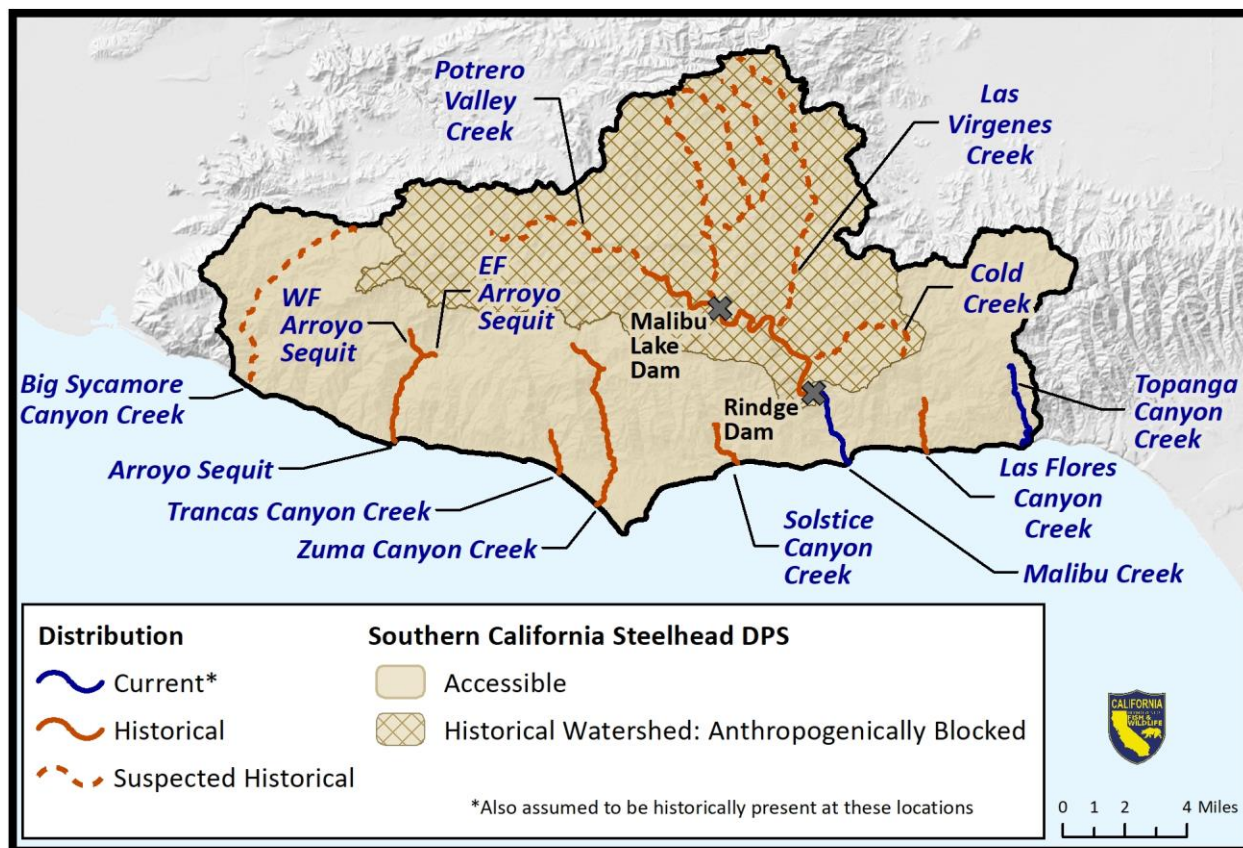


Figure 9. Map of the Santa Monica Mountains BPG depicting known and suspected current and historical distribution. Abbreviations: EF = East Fork, WF = West Fork.

In more recent years, *O. mykiss* have been seen in Malibu Creek below Rindge Dam (Becker and Reining 2008). A die off of about 250 *O. mykiss* occurred in the creek in 2006 after yellowing of the fish was noticed during snorkel surveys (Becker and Reining 2008). Recent drought conditions starting in 2012 have led to reduced abundances of *O. mykiss* in Malibu Creek based on similar observations on Topanga Creek (Dagit et al. 2017)

4.3.3.2 Topanga Creek

Topanga Creek empties into the ocean at Topanga Beach and contains similar stream mileage to Malibu Creek but contains less accessible habitat for Southern SH/RT (Becker and Reining 2008). Some steelhead can access Topanga Creek in years when there is sufficient precipitation (Becker and Reining 2008), and *O. mykiss* of various sizes were observed in the watershed in

1979 (Becker and Reining 2008). Juvenile *O. mykiss* were observed by Department staff in Topanga Creek again in 1982 (Becker and Reining 2008). Unlike in Malibu Creek, the upstream impassable migration barrier for Southern SH/RT is a natural barrier in Topanga Creek (Camm Swift, Emeritus, Section of Fishes, Natural History Museum of Los Angeles County, personal communication).

The Southern SH/RT population in Topanga Creek was recently monitored from 2001-2007, revealing consistent use by spawning steelhead adults and successful smolt production (Becker and Reining 2008). Bell et al. (2011b) characterized the Topanga population as a satellite population that is supported by other populations in the Southern SH/RT range but provides minimal production to other streams. As a satellite population, Topanga Creek *O. mykiss* support the metapopulation in southern California but are more vulnerable to extirpation (Bell et al. 2011b). The effects of the most recent prolonged drought on Southern SH/RT have been severe. Significant reductions for all life-stages were observed from 2012-2016, leading to reductions of the population from 358 individuals in 2008 to less than 50 individuals in 2016 (Dagit et al. 2017).

4.3.3.3 Other Creeks

Big Sycamore Canyon Creek was surveyed in 1989-1990 but no steelhead were observed (Becker and Reining 2008). NMFS (2005) designated the population as extirpated after another survey in 2002.

Arroyo Sequit Creek was reported to have a small historical steelhead run. Steelhead were seen in a 1989-1990 survey of the stream and again in a 1993 survey. From 2000-2007 steelhead were reported utilizing Arroyo Sequit Creek (Becker and Reining 2008).

Overall, from 2005-2019, monitoring in Arroyo Sequit Creek done by the Resource Conservation District of the Santa Monica Mountains (RCDSMM) has observed few *O. mykiss*, primarily due to two instream barriers that were eventually removed in 2016. Two adult observations occurred after the removal of barriers in 2017 (Dagit et al. 2019). There is also limited documentation of steelhead in the West and East forks of Arroyo Sequit Creek (Becker and Reining 2008). Las Flores Canyon Creek is reported to have suitable steelhead habitat but there is no evidence of historical or present use by steelhead (Becker and Reining 2008; Titus et al. 2010).

4.3.4 Mojave Rim Biogeographic Population Group

There are three relatively large watersheds that make up the Mojave Rim BPG (NMFS 2012a; Figure 10). These watersheds include the San Gabriel, Santa Ana, and Los Angeles rivers. The

headwaters of these streams are in the San Gabriel and San Bernardino mountains, which experience greater seasonal precipitation than is seen in the neighboring BPGs. Lower watershed areas span the flat coastal plain of the Los Angeles River, which historically contained widespread springs and marshes (Mendenhall 1907). Over time the mouths of these rivers have drifted to different areas along the coast. Currently, the river mouths are each less than 20 miles apart (NMFS 2012a).

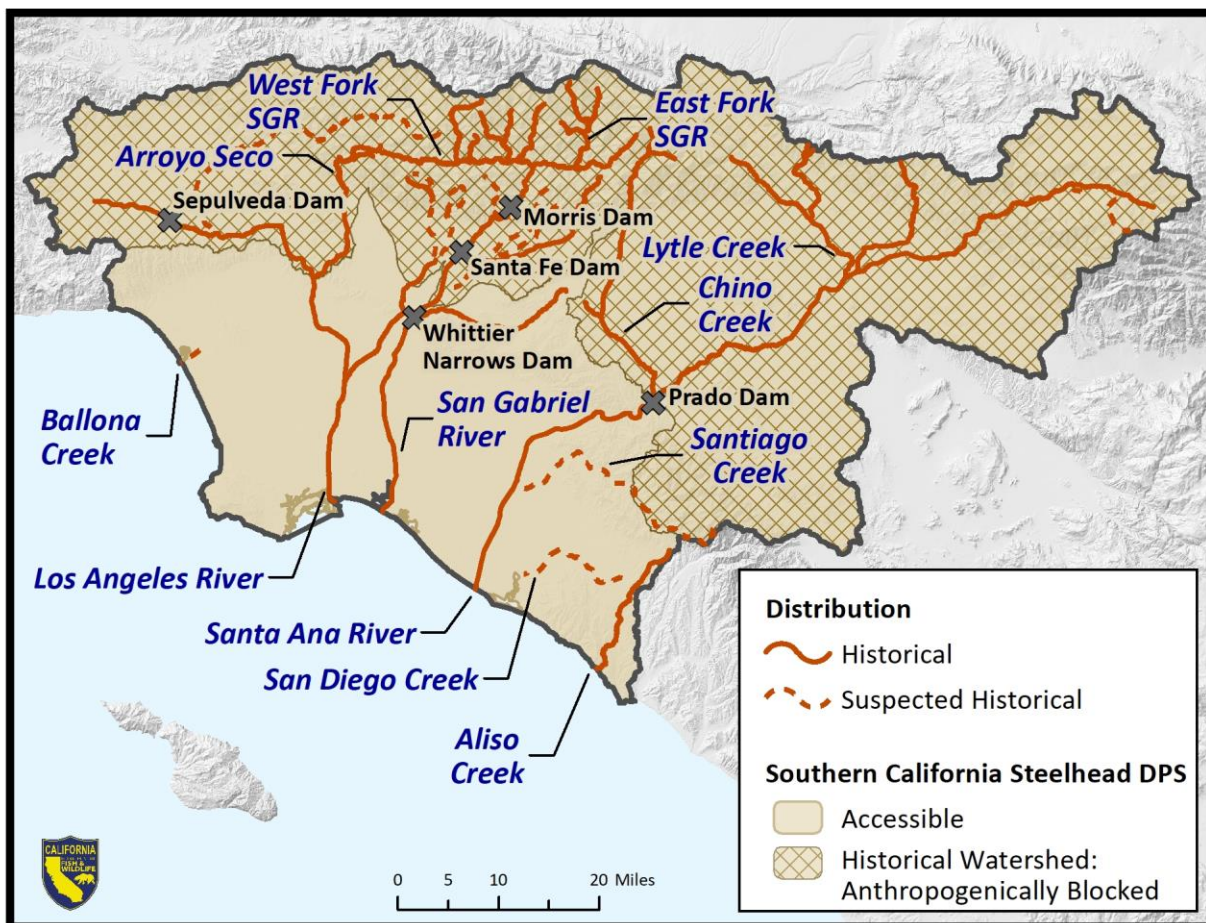


Figure 10. Map of the Mojave Rim BPG depicting known and suspected current and historical distribution. Abbreviations: SGR= San Gabriel River.

4.3.4.1 San Gabriel River

The San Gabriel River encompasses more than 58 stream miles but about half of it is channelized below Santa Fe Dam. Morris Dam and Santa Fe Dam were both constructed in the 1930s (Becker and Reining 2008) and are considered complete barriers to fish migration. Rainbow trout were seen by Department staff in the 1930s, but the river was also stocked during that time (Becker and Reining 2008). Stocking below Morris Dam also occurred on Little Dalton Creek in 1945 (Titus et al. 2010). Rainbow Trout fishing was good from the late 1930s to

late 1940s according to various Department stream surveys and in 1951, Department staff noted that natural production was average (Becker and Reining 2008). Fish Canyon Creek and Robert's Canyon Creek, which are mainstem tributaries downstream of Morris Dam, were observed by Department surveyors to have *O. mykiss* in the 1940s, 1950s, and 1973 (Titus et al. 2010).

Southern SH/RT historically occurred in a few tributaries of the San Gabriel River such as San Jose Creek. Many tributaries to the San Gabriel River have been channelized and contain fish passage barriers. Most were stocked for recreational angling in the 1930s and 1940s (Becker and Reining 2008). Southern SH/RT remain in tributaries above the two barrier dams and are known to presently inhabit the East Fork. The ancestry of these fish is unclear and may have genetic influence from stocking *O. mykiss* from other watersheds (Nielsen 1999). There is also a remnant historical population of Rainbow Trout just below Morris Dam that appears to self-propagate (Becker and Reining 2008).

4.3.4.2 Santa Ana River

The Santa Ana River is the largest river within southern California at almost 100 miles long (Becker and Reining 2008). Prado Dam, which is located approximately 30 miles upstream of the river outlet, was constructed in 1941 (O.C. Public Works, n.d.). The lower 24 miles of channelized river below the dam outflows to the Pacific Ocean in Huntington Beach (Becker and Reining 2008). Rainbow Trout were first observed and captured in the upper Santa Ana River drainage in the 1850s (Boughton et al. 2006). Rainbow Trout were also observed in the mountainous upper watershed during the 1930s, coinciding with when stocking occurred (Becker and Reining 2008). A steelhead run was historically present in the lower river (Becker and Reining 2008); however, in 1951 and 1955, no *O. mykiss* were observed in any stream reaches below Prado Dam during Department surveys (Titus et al. 2010). Various water uses have highly altered flows in the Santa Ana River and low numbers of fish in the lower river are attributed to limited water releases from Prado Dam (Titus et al. 2010). Southern SH/RT are thought to be extirpated from the Santa Ana River (Nehlsen et al. 1991), but resident *O. mykiss* remain in the upper watershed above natural and manmade impassable barriers (Boughton et al. 2005).

Southern SH/RT were historically present in Santiago Creek below Prado Dam. Many tributaries upstream of where the dam was built were stocked with *O. mykiss* in the 1930s and fish have been observed reproducing naturally in the decades that followed (Becker and Reining 2008).

4.3.4.3 Los Angeles River

The Los Angeles River is approximately 52 miles long and flows to the Pacific Ocean in Long Beach. Like the San Gabriel River, the Los Angeles River is completely channelized with much of the lower mainstem channel paved with concrete for flood control purposes (Becker and Reining 2008; Titus et al. 2010). Southern SH/RT are assumed to have been present in the watershed but there have been no actual observations to confirm this assumption (Titus et al. 2010). Major tributaries to the Los Angeles River were stocked in the 1930s or 1940s (Becker and Reining 2008; Titus et al. 2010) but some of these tributaries were later channelized and no longer support *O. mykiss*. Due to the highly modified nature of the river basin, Southern SH/RT cannot utilize the mainstem Los Angeles River for spawning or rearing (Titus et al. 2010) and are considered extirpated (Nehlsen et al. 1991). However, resident *O. mykiss* have been observed in the major tributaries of the Los Angeles River, including Arroyo Seco and Big Tujunga Creeks (Becker and Reining 2008). Fish passage by native Southern SH/RT on Arroyo Seco is obstructed by Devil's Gate Dam. Recently, Department-led fish rescues have transplanted Southern SH/RT from the West Fork San Gabriel River and Bear Creek to Arroyo Seco as a result of the Bobcat Fire (Pareti 2020).

4.3.5 Santa Catalina Gulf Coast Biogeographic Population Group

Multiple medium sized watersheds comprise the Santa Catalina Gulf Coast BPG (Figure 11). Most have their headwaters in the Santa Ana or Peninsular Mountain ranges and flow south over coastal terraces (NMFS 2012a). Many watersheds in the BPG have intermittent flow and are seasonally dry due to limited precipitation and groundwater depletion (D. Boughton, NOAA, personal communication). Some smaller drainages within the BPG might occasionally support steelhead. Streams in this BPG have substantial tributary mileage in the upper watershed areas due to the fragmented landscape in the region (NMFS 2012a).

4.3.5.1 San Juan Creek

San Juan Creek is 22-mile stream located in Orange and Riverside Counties. Arroyo Trabuco Creek is a major tributary to San Juan Creek with approximately the same stream length (Becker and Reining 2008). Steelhead were observed in the creek in 1939 (Swift et al. 1993) and in the 1940s as well as in 1968 and 1974 (Becker and Reining 2008). Trout stocking to support fishing in San Juan Creek occurred year-round in 1981 (Becker and Reining 2008) and possibly in other years. San Juan Creek contains suitable habitat for *O. mykiss*, which have been observed in some but not all years in recent decades (Becker and Reining 2008).

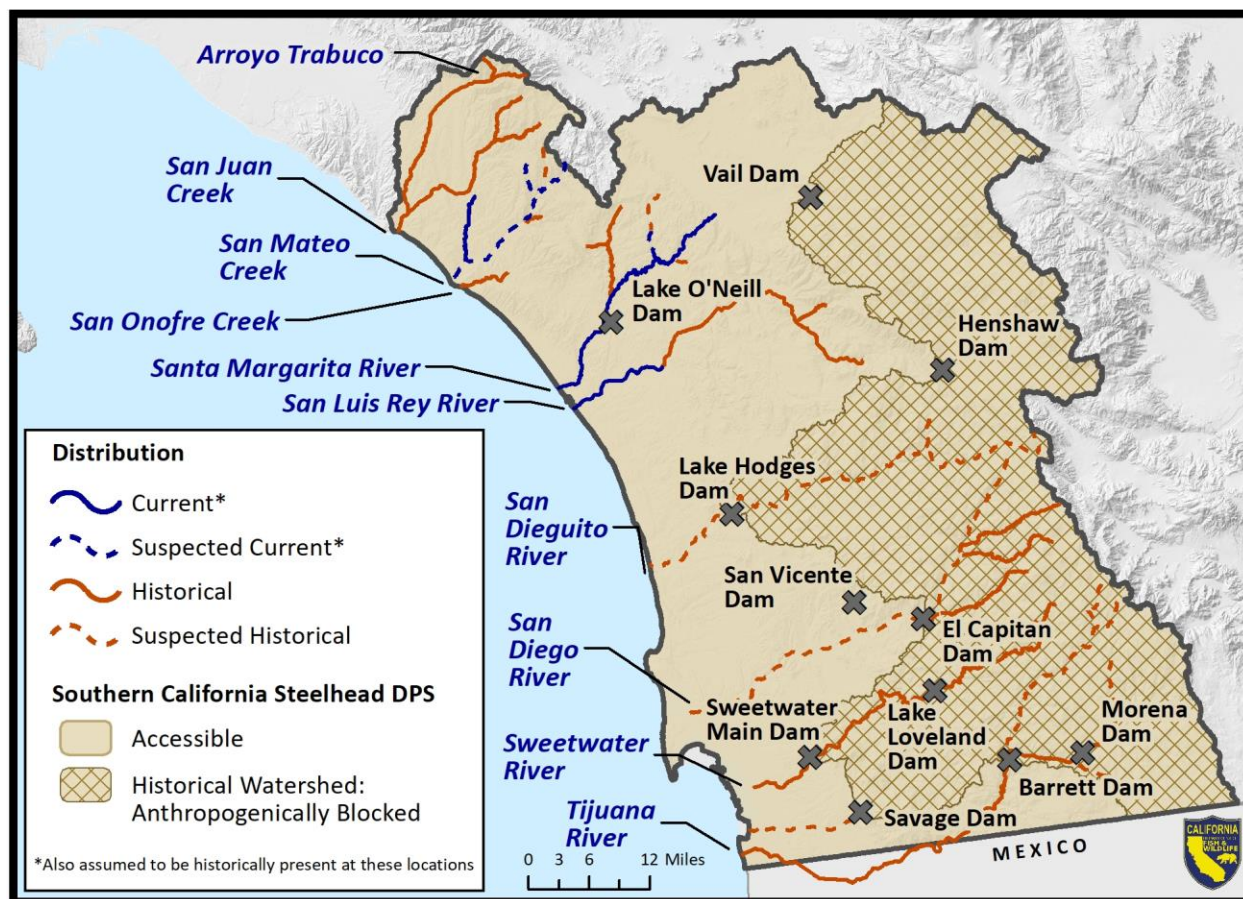


Figure 11. Map of the Santa Catalina Gulf Coast BPG depicting known and suspected current and historical distribution.

Arroyo Trabuco was a historical Southern SH/RT stream; however, there is now a complete barrier to fish migration about 2.4 miles from the confluence with San Juan Creek. Regardless, the stream still appears to contain suitable habitat and steelhead were still thought to be present in 2004 below the barrier (Becker and Reining 2008). Recently, efforts to remediate fish passage at two total barriers to migration on Trabuco Creek are in progress. Completion of this project would provide access to 15 miles of upstream spawning and rearing habitat.

4.3.5.2 San Mateo Creek

San Mateo Creek, which has a similar stream length as San Juan creek, supported a historical steelhead run (Titus et al. 2010). In the early 1900s, anglers were successful in catching Southern SH/RT of greater sizes than in other regional watersheds (Titus et al. 2010). In 1939, juvenile Southern SH/RT were observed and rescued in the thousands from isolated reaches and transferred to the estuary lagoon (Titus et al. 2010). Stocking of the creek began in 1945 (Becker and Reining 2008). Anadromous and resident Southern SH/RT were thought to persist

in 1950 (Becker and Reining 2008), though after that year, Southern SH/RT encounters declined (Titus et al. 2010). In 1999, *O. mykiss* sampled by the Department were surmised to be offspring from anadromous Southern SH/RT because of the lack of a resident population (Becker and Reining 2008). Habitat quality in the watershed has been degraded by anthropogenic activities and intermittent streamflow has posed migration issues for Southern SH/RT (Titus et al. 2010). Steelhead were thought to be extirpated from San Mateo Creek (Nehlsen et al. 1991) until more recent monitoring by Hovey (2004) documented a small resident *O. mykiss* population in Devil Canyon Creek, a major tributary to San Mateo Creek. Currently, the San Diego Regional Water Quality Control Board is considering using a draft invasive species Total Maximum Daily Load (TMDL) and plan to certify that actions of other entities will correct impairments to the creek caused by invasive species (Loflen 2022).

4.3.5.3 San Onofre Creek

San Onofre Creek consists of 13 miles of stream in Orange County. Personal observations of annual steelhead runs in the creek prior to 1946 suggest it was a historical Southern SH/RT stream (Becker and Reining 2008). Fletcher Creek, a tributary to San Onofre Creek, was considered a steelhead rearing area in 1950 and *O. mykiss* were observed by Department staff during a survey in 1979 (Titus et al. 2010). By the 2000s, San Onofre Creek was observed to be dry (Boughton et al. 2005), though reaches in the upper watershed may still offer suitable *O. mykiss* habitat (Becker and Reining 2008).

4.3.5.4 Santa Margarita River

The Santa Margarita River is almost 30 miles long, but a diversion weir located approximately ten miles upstream within the boundaries of Camp Pendleton likely acts as a complete barrier to upstream fish migration (Becker and Reining 2008; Titus et al. 2010). This diversion eliminates surface flow during most of the year (Titus et al. 2010). Adult and juvenile steelhead were observed in the river in the 1930s and 1940s and steelhead were thought to migrate upstream to the town of Fallbrook when flows allowed (Becker and Reining 2008). DeLuz Creek, a tributary to the Santa Margarita River, also historically supported steelhead (Becker and Reining 2008). Stocking of *O. mykiss* in the Santa Margarita watershed began in 1941 (Becker and Reining 2008) and occurred most recently in 1984 (Titus et al. 2010). Currently, the reaches downstream of O'Neill Lake do not support Southern SH/RT spawning (Titus et al. 2010) and they are thought to be extirpated (Nehlsen et al. 1991). As part of the Santa Margarita River Conjunctive Use Project, the existing O'Neill weir diversion will be replaced with an inflatable structure that will allow fish passage during most flow events (FPUD 2016). Further upstream, efforts are also underway to replace a fish passage barrier at the Sandia Creek Drive bridge to provide passage to 12 miles of upstream rearing and spawning habitat (Dudek 2021).

4.3.5.5 San Luis Rey River

The San Luis Rey River is a large river in northern San Diego County that runs approximately 69 stream miles from its river mouth near Oceanside, California. Lake Henshaw Dam, which was built in 1924, reduces the downstream flow of the river and blocks steelhead access to the uppermost portion of the drainage (Becker and Reining 2008; Titus et al. 2010). According to Native Americans and other observers of *O. mykiss* in the late 1800s, there was a historical run of steelhead that was able to reach areas above where the dam was constructed (Becker and Reining 2008). Stocking of Rainbow Trout occurred sometime prior to 1946 (Becker and Reining 2008). Although resident Rainbow Trout remain in tributaries of the upper watershed like Pauma Creek and the West Fork San Luis Rey River (Becker and Reining 2008), native Southern SH/RT are extirpated from the lower reaches of the San Luis Rey River (Nehlsen et al. 1991; Becker and Reining 2008).

4.3.5.6 San Dieguito River

The San Dieguito River is a large river in San Diego County that runs for 23 stream miles before entering into the Pacific Ocean north of the City of San Diego. Hodges Dam, which was constructed 12 miles upstream from the mouth in 1918, serves as a complete barrier to anadromy (Becker and Reining 2008). A journal article by Hubbs (1946) mentioned anglers catching possible steelhead in the estuary (Titus et al. 2010). Rainbow trout have been stocked below the dam (Titus et al. 2010); however, those downstream reaches no longer support *O. mykiss* (Becker and Reining 2008). Prior to the construction of the Sutherland Lake dam on Santa Ysabel Creek, a major tributary of the San Dieguito River, Department staff saw *O. mykiss* in a creek upstream of the eventual dam site, though there had been stocking efforts in that creek (Becker and Reining 2008). Black Canyon Creek, another smaller tributary to the San Dieguito River, was also stocked for rainbow trout fishing (Becker and Reining 2008).

4.3.5.7 San Diego River

The San Diego River has a stream length of 52 miles but El Capitan Dam, built in 1934, blocks about 22 miles of historical Southern SH/RT habitat (Becker and Reining 2008). Additionally, channelization of downstream reaches has eliminated suitable habitat below the dam (Titus et al. 2010). Anglers may have caught steelhead historically (Titus et al. 2010) but the population is now thought to be extinct (Nehlsen et al. 1991). Upper watershed tributaries above the dam were stocked in the 1930s and earlier and may still support *O. mykiss* (Becker and Reining 2008; Titus et al. 2010).

4.3.5.8 Sweetwater River

The Sweetwater River is a large river in San Diego County that runs for 55 miles before emptying into San Diego Bay southeast of the City of San Diego. The Sweetwater Reservoir, formed by the construction of the Sweetwater Dam in 1888, serves as a total barrier to anadromy (Becker and Reining 2008; Titus et al. 2010). Although *O. mykiss* were present historically and may still be found in the upper watershed, there are no mentions of a historical anadromous steelhead run in the Sweetwater River (Becker and Reining 2008; Titus et al. 2010). In years leading up to 1946, Cold Stream, a small tributary to Sweetwater River, was stocked with Rainbow Trout and these fish may have continued to naturally reproduce for some time (Becker and Reining 2008).

4.3.5.9 Otay River

The Otay River enters the south end of San Diego Bay near the U.S.-Mexico Border. There are no known historical or current records of Southern SH/RT existing in the Otay River. Fish passage is obstructed by the dam that forms Lower Otay Lake, though there may be *O. mykiss* residing in upper reaches above the reservoir (Titus et al. 2010).

4.3.5.10 Tijuana River

The Tijuana River is the southernmost stream within the Southern SH/RT range and extends for 26 miles from the intersection of Cottonwood Creek (Becker and Reining 2008). Other than one account of a few steelhead seen in 1927 by Department law enforcement, there has been no other documentation of historical use of the mainstem river (Titus et al. 2010). Steelhead were present in Cottonwood Creek in the mid-1930s, which was stocked with *O. mykiss* at that time, but Southern SH/RT are no longer able to pass multiple dams within the creek (Titus et al. 2010). If a steelhead run did exist in the Tijuana watershed, it is now assumed to be extirpated (Titus et al. 2010).

4.4 Abundance and Trends

To provide the best scientific information in our evaluation of Southern SH/RT as required by Fish and Game Code Section 2074.6, we analyzed its status and trends with annual abundance data compiled from a variety of sources (see Section 4.2 for Sources of Information).

Southern SH/RT, as defined in the Petition, include both anadromous and resident forms below complete migration barriers. To account for both life-history forms in our review, our analyses in Sections 4.4-4.8 examine data on anadromous adult Southern SH/RT (Adult SH) separately from data on *O. mykiss* not identified as anadromous adult Southern SH/RT (Other *O. mykiss*),

as most existing monitoring efforts produce datasets that use these two categories. This is because it is possible to distinguish anadromous adult Southern SH/RT in rivers and streams due to their larger size (fork length >400mm), greater girth, and steel-gray appearance, but it is otherwise difficult to conclude which life history an individual *O. mykiss* that does not have the identifying characteristics of an adult fish has expressed or will express. (Dagit et al. 2020; Moyle et al. 2017).

The analysis presented below is structured on the five BPGs with an emphasis on Core 1 and Core 2 populations within each BPG (NMFS 2012a; Boughton et al. 2007). The BPGs are a federal delineation based on a suite of environmental conditions (e.g., hydrology, local climate, geography) and watershed characteristics (i.e., large inland or short coastal streams). Core 1 and 2 populations occupy watersheds that exhibit the physical and hydrological conditions necessary to sustain self-sufficient viable populations of Southern SH/RT (NMFS 2012a). Datasets were reviewed to ensure that they were collected from monitoring conducted below the upper limit to anadromy in each watershed to remain consistent with the geographic scope of the listing unit proposed in the Petition. Where sufficient data were available for a given population, we present and discuss abundance and long-term population trend estimates for each BPG. The Department was unable to analyze core watersheds in the Mojave Rim and Santa Catalina Gulf Coast BPGs in detail due to data limitations. In these instances, as well as in other cases where data was limiting or unavailable, we provide a qualitative discussion, such as a viability assessment, based on the sources identified in Section 4.2 (Boughton et al. 2022a).

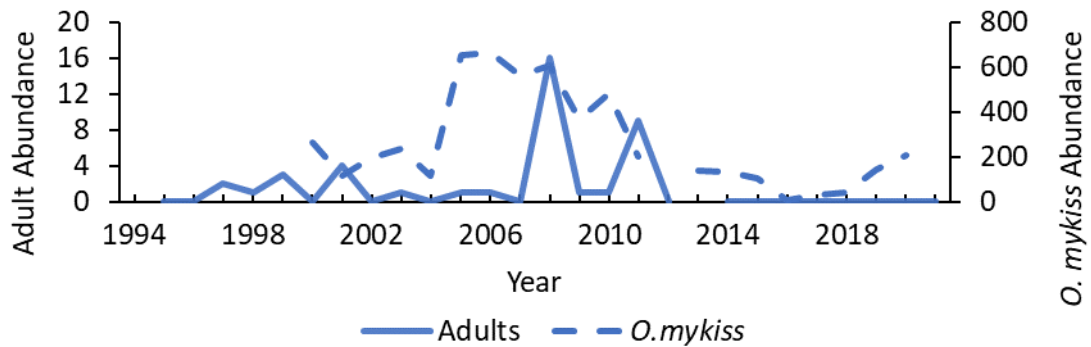
4.4.1 Time Series of Abundance

Southern SH/RT populations in the Monte Arido Highlands BGP have the longest running time-series dating back to the 1990s for the Santa Ynez and Santa Clara rivers (COMB 2022; Booth 2016) and the early 2000s for the Ventura River (CMWD 2005-2021; Dagit et al. 2020) (Figure 12). However, no organized monitoring efforts have been conducted on the Santa Maria River since steelhead were federally listed in 1997. Therefore, no further analysis of the Santa Maria Southern SH/RT populations are conducted in this chapter.

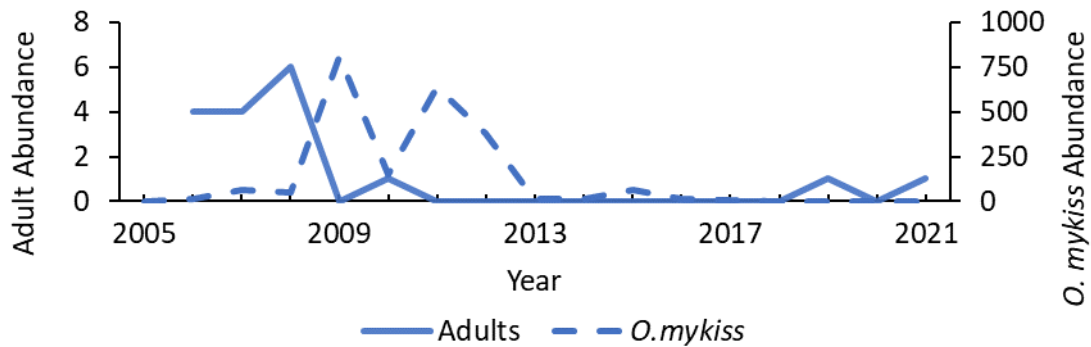
More recently, monitoring has been intermittently conducted on Carpinteria, Mission, and Arroyo Hondo in the Conception Coast BPG by the Department (Boughton et al. 2022a). Malibu, Topanga, and Arroyo Sequit creeks in the Santa Monica Mountains BPG have been actively monitored since the early 2000s (Dagit et al. 2019) (Figure 13). No recent or historical monitoring has been conducted in either the Mojave Rim or Santa Catalina Gulf Coast BPGs.

4.4.1.1 Monte Arido Highlands BPG

A. Santa Ynez River



B. Ventura River



C. Santa Clara River

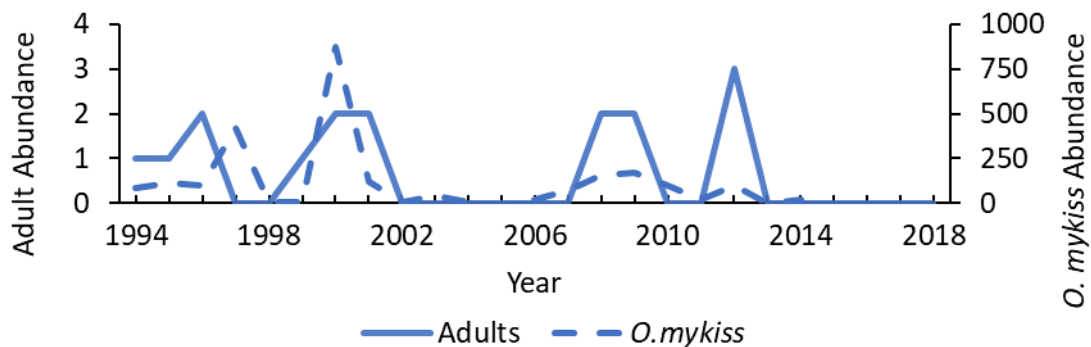


Figure 12. Adult steelhead (Adults) and other *O. mykiss* (*O. mykiss*) abundances for the Monte Arido Highlands BPG. A) Santa Ynez River; no data 2013. Biological Opinion Incidental Take provisions have been required since 2014. B) Ventura River. C) Santa Clara River. Adult abundance is on the left -axis with the solid blue line and *O. mykiss* abundance is on the right axis with the dashed blue line. Note different scales on the Y-axis.

4.4.1.2 Conception Coast BPG

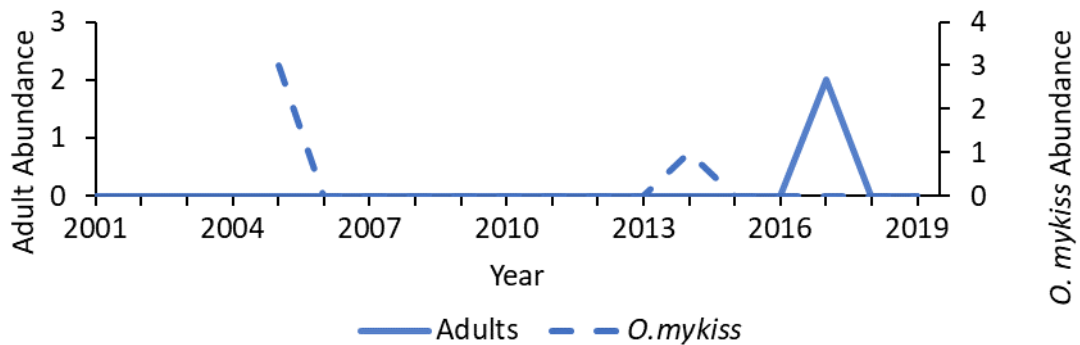
Very few monitoring activities have occurred throughout the Conception Coast BPG, and most of the work that has occurred in more recent years was conducted by the Department. We were unable to develop a full-time series of Southern SH/RT abundance for Conception Coast populations.

Although past monitoring is limited in this BPG, Dagit et. al (2020) documented a total of 42 adult steelhead opportunistic observations from 2000-2018. Two adults were observed in Arroyo Hondo Creek in 2017 and 10 adults were documented in the Goleta Slough Complex with the most recent observation occurring in 2017. For the entirety of Conception Coast BPG, 64% (n=27) of all adult observations occurred in Mission Creek, primarily from 1998-2008. However, from 2018-2022, Department redd and snorkel surveys documented zero adult steelhead in Mission Creek (K. Evans, CDFW, unpublished data). Three adults were observed opportunistically in Carpinteria Creek in 2008 (Dagit et al. 2020); however, from 2008-2019, zero adult steelhead were observed based on recent monitoring conducted by the Department (Boughton et al. 2022a).

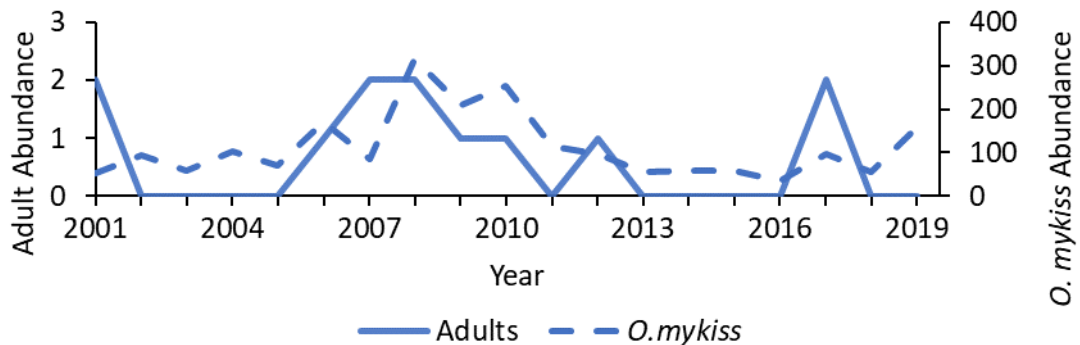
There is also limited data for *O. mykiss* in the Conception Coast BPG. No *O. mykiss* have been documented in Carpinteria Creek since 2016. In Mission Creek, no *O. mykiss* were observed from bankside surveys during the 2018-2019 spawning season (Carmody et al. 2019). In recent years, the largest number of *O. mykiss* observations in this BPG have occurred on Arroyo Hondo Creek, indicating that despite being a small watershed, the creek contains suitable habitat that is relatively undisturbed due to its inclusion in a natural reserve system (NMFS 2012a). Snorkel surveys have documented a total of 2,363 *O. mykiss* in Arroyo Hondo Creek from 2017-2019 (Carmody et al. 2019), while bankside *O. mykiss* observations have documented a total of 12,090 *O. mykiss* from 2015-2022 (K. Evans, CDFW, unpublished data).

4.4.1.3 Santa Monica Mountains BPG

A. Arroyo Sequit Creek



B. Topanga Creek



C. Malibu Creek

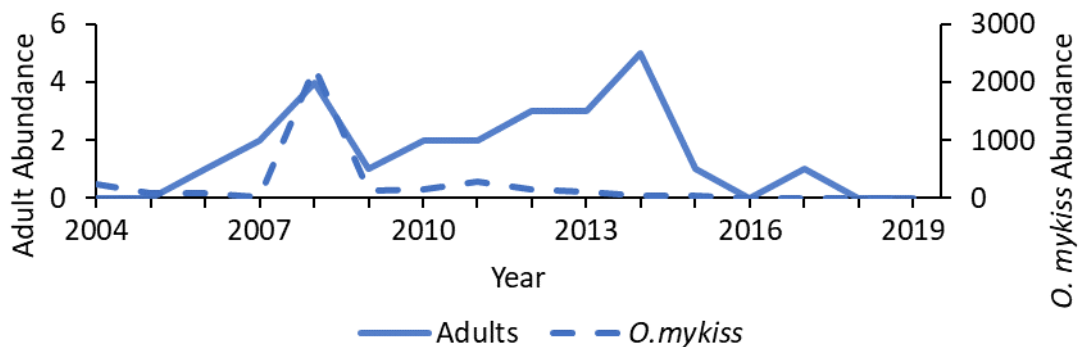


Figure 13. Adult steelhead (Adults) and other *O. mykiss* (*O. mykiss*) abundances for the Santa Monica Mountains BPG. A) Arroyo Sequit Creek. B) Topanga Creek. C) Malibu Creek. Adult abundance is indicated on the left -axis and delineated by the solid blue line and *O. mykiss* abundance is indicated on the right axis and delineated by the dashed blue line. Note different scales on the Y-axis.

4.4.1.4 Mojave Rim BPG

Abundance data is generally not available for this BPG; therefore, we were unable to create a full-time series of Southern SH/RT abundances for the San Gabriel River, Santa Ana River, and Los Angeles River watersheds.

A total of 3 adult steelhead were observed opportunistically in the Mojave Rim BPG from 2000-2018. Two observations occurred on Ballona Creek in 2007, and one observation occurred on the San Gabriel River in 2016 (Dagit et al. 2020). It is generally accepted that all over-summering, rearing, and spawning habitat occurring upstream is no longer accessible to Southern SH/RT due to the presence of extensive physical and velocity related passage barriers located within the lower reaches of each of the three major rivers; therefore, steelhead are not expected to be present in the lower reaches of these watersheds (NMFS 2012a).

4.4.1.5 Santa Catalina Gulf Coast BPG

We were unable to construct a full-time series of Southern SH/RT abundance for these populations because no data series were available to analyze the Santa Catalina Gulf Coast BPG. A total of 15 adult steelhead have been observed in the Santa Catalina Gulf Coast BPG from 2001-2018. Ten of these steelhead observations occurred on either San Juan or San Mateo creeks, and the remainder of observations were distributed throughout the Santa Margarita and San Luis Rey rivers and Los Penasquitos Creek (Dagit et al. 2020).

4.4.2 Geometric Mean Abundance

We calculated the geometric mean of abundance for Southern SH/RT populations (N_a) with at least 3-4 generations of data for three time periods. The long-term calculation represents the total available time series. The medium-term calculation represents 12 years or three generations of data, while the short-term calculation is for the most recent 5 years of data. Missing data are noted in the following tables and there was no effort to interpolate or otherwise fill in missing data. Furthermore, we did not substitute values for years in which zero individuals were observed; instead, these values were omitted from the calculation in order to obtain an informative result.

The geometric mean is a useful metric for evaluating species' status because it calculates the central tendency of abundance while minimizing the effect of outliers in the data. Furthermore, the geometric mean is thought to more effectively characterize time series data of abundance based on counts than the arithmetic mean (Good et al. 2005; Spence et al. 2008). We did not calculate arithmetic mean because of its tendency to be overly sensitive to outlier data to a few

large counts and can result in the incorrect depiction of central tendency. A range of minimum and maximum abundances were also calculated to provide scale.

Using methods from Spence et al. (2008), we defined the geometric mean of Southern SH/RT abundance as:

$$Na_{(geom)} = (\prod Na(i))^{1/n}$$

where $Na(i)$ is the total number of adult steelhead in year i , and n is the number of years of data available.

4.4.2.1 Monte Arido Highlands BPG

Maximum abundance of adult steelhead in the Monte Arido Highlands BPG has remained consistently low since the mid-1990s and early 2000s (Table 2a-2c). For each population examined, maximum counts from the most recent 5-year period are less than either the medium or long-term time frames. For all three watersheds, years in which zero adults were observed have occurred more frequently than years in which at least one fish was observed.

The highest average abundance in this BPG was during the 12-year time frame (2010-2021) on the Santa Ynez River. Both the Santa Clara and Santa Ynez rivers have higher 12-year averages compared to the long-term average. Overall, all three populations have lower 5-year averages when compared to the long-term average and geometric mean abundances remain low across all time frames (Table 3).

Table 2a. Minimum and maximum adult steelhead abundance for the Santa Ynez River over three-time frames: 1995 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year). No data for 2013. Biological Opinion Incidental Take provisions have been required since 2014.

Abundance	Minimum	Maximum
Long-term	0	16
12-year	0	9
5-year	0	0

Table 2b. Minimum and maximum adult steelhead abundance for the Ventura River over three-time frames: 2006 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year).

Abundance	Minimum	Maximum
Long-term	0	6
12-year	0	1
5-year	0	1

Table 2c. Minimum and maximum adult steelhead abundance for the Santa Clara River over three-time frames: 1994 to 2018 (long-term), 2007 to 2018 (12-year), and 2014 to 2018 (5-year).

Abundance	Minimum	Maximum
Long-term	0	3
12-year	0	3
5-year	0	0

Table 3. Long-term, medium-term, and short-term geometric mean abundance of adult steelhead in the Monte Arido Highlands BPG.

Population	Years	Long-term Mean	Years	12-year mean	Years	5-year mean
Santa Ynez River ¹	1995-2021	2.1	2010-2021	3.0	2017-2021	0.0
Ventura River	2006-2021	2.1	2010-2021	1.0	2017-2021	1.0
Santa Clara River	1994-2018	1.7	2007-2018	2.3	2014-2018	0

¹ No data long-term 2013; Biological Opinion Incidental Take provisions have been required since 2014.

Maximum abundances of *O. mykiss* for all populations in the Monte Arido BPG are considerably less when comparing the 5-year time frame to the long-term time frame (Table 4a-4c). On the Ventura River, a maximum of 807 *O. mykiss* were observed during the long-term time frame compared to just nine individuals being observed during the most recent 5-year time frame. Minimum abundances range from zero to five *O. mykiss* for all three time-periods and populations. All three *O. mykiss* populations have lower 5-year averages compared to the 12-year and long-term time frames (Table 5). The Santa Ynez River has the highest average abundance of the three populations for each time frame. Overall, mean abundances of *O. mykiss* in this BPG have declined to low numbers, especially in the last five years.

Table 4a. Minimum and maximum *O. mykiss* (Other *O. mykiss*) abundance for the Santa Ynez River over three-time frames: 2001 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year). No data for 2013. Biological Opinion Incidental Take provisions have been required since 2014.

Abundance	Minimum	Maximum
Long-term	5	665
12-year	5	484
5-year	5	205

Table 4b. Minimum and maximum *O. mykiss* abundance (Other *O. mykiss*) for the Ventura River over three-time frames: 2005 to 2021 (long-term), 2010 to 2021 (12-year), and 2017 to 2021 (5-year).

Abundance	Minimum	Maximum
Long-term	0	807
12-year	0	640
5-year	0	9

Table 4c. Minimum and maximum other *O. mykiss* abundance for the Santa Clara River over three-time frames: 1994 to 2014 (long-term), 2003 to 2014 (12-year), and 2010 to 2014 (5-year). No data for 2005.

Abundance	Minimum	Maximum
Long-term	1	876
12-year	1	170
5-year	1	100

Table 5. Long-term, medium-term, and short-term geometric mean abundance of *O. mykiss* (Other *O. mykiss*) in the Monte Arido Highlands BPG.

Population	Years	Long-term Mean	Years	12-year mean	Years	5-year mean
Santa Ynez River ¹	2001-2021	166.4	2010-2021	100.5	2017-2021	43.7
Ventura River	2005-2021	44.7	2010-2021	34.5	2017-2021	3.0
Santa Clara River ²	1994-2014	39.5	2003-2014	30.5	2010-2014	21

¹ No data long-term 2013; Biological Opinion Incidental Take provisions have been required since 2014.

² No data long-term 2005

4.4.2.2 Conception Coast BPG

We were unable to calculate geometric mean abundance estimates for the Conception Coast BPG aside from the Arroyo Hondo Creek *O. mykiss* population due to the lack of long-term data. Based on bankside *O. mykiss* observations as part of spawner redd surveys, the geometric mean abundance was 581 individuals from 2015-2022, the maximum abundance of 8,614 individuals was observed in 2021, and the minimum abundance of zero individuals was observed in 2022 (K. Evans, CDFW, unpublished data).

4.4.2.3 Santa Monica Mountains BPG

Maximum abundance counts of adult steelhead in the Santa Monica Mountains BPG have remained consistently low since the early 2000s (Table 6a-6c). A total of two adult steelhead were observed in Arroyo Sequit Creek in 2017, coinciding with the removal of all instream barriers on the creek below the Mulholland culvert in 2016; however, no adult steelhead have been observed in this creek since 2017. The maximum abundance of adult steelhead in Topanga and Malibu creeks has not been greater than five individuals for any given year during all time periods. For adult steelhead populations in both Topanga and Malibu creeks, the 5-year average is lower than the long-term average (Table 7). Overall, average abundances of adult steelhead for all three populations remain low across all time frames.

Table 6a. Minimum and maximum adult steelhead abundance for Arroyo Sequit Creek over three-time frames: 2005 to 2018 (long-term), 2007 to 2018 (12-year), and 2014 to 2018 (5-year).

Abundance	Minimum	Maximum
Long-term	0	2
12-year	0	2
5-year	0	2

Table 6b. Minimum and maximum adult steelhead abundance for Malibu Creek over three-time frames: 2004 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year).

Abundance	Minimum	Maximum
Long-term	0	5
12-year	0	5
5-year	0	1

Table 6c. Minimum and maximum adult steelhead abundance for Topanga Creek over three-time frames: 2001 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year).

Abundance	Minimum	Maximum
Long-term	0	2
12-year	0	2
5-year	0	2

Table 7. Long-term, medium-term, and short-term geometric mean abundance of adult steelhead in the Santa Monica Mountains BPG.

Population	Years	Long-term mean	Years	12-year mean	Years	5-year mean
Arroyo Sequit Creek ¹	2005-2019	NA	2008-2019	NA	2015-2019	NA
Topanga Creek	2001-2019	1.4	2008-2019	1.3	2015-2019	1
Malibu Creek	2004-2019	1.9	2008-2019	2.1	2015-2019	1

¹ Insufficient data to produce meaningful results.

For all populations in this BPG, maximum abundances of *O. mykiss* for the 5-year time frame are considerably lower compared to the long-term time frame (Table 8a-8c). Since 2005, a total of four *O. mykiss* were observed in Arroyo Sequit Creek with most years recording zero observations (Table 8a). For the Malibu Creek population, a maximum abundance of 2,245 *O. mykiss* was observed from 2004-2019 compared to just 32 individuals during the 5-year time frame (Table 8b). Topanga Creek appears to support a small but consistent population of *O. mykiss* with a long-term maximum and minimum abundance of 316 and 34 individuals, respectively (Table 8c). Topanga Creek *O. mykiss* have also declined in abundance over the three time periods, but this difference is less pronounced than the decline observed for the Malibu Creek population (Table 9).

Table 8a. Minimum and maximum *O. mykiss* (Other *O. mykiss*) abundance for Arroyo Sequit Creek over three-time frames: 2005 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year).

Abundance	Minimum	Maximum
Long-term	0	3
12-year	0	1
5-year	0	0

Table 8b. Minimum and maximum *O. mykiss* (Other *O. mykiss*) abundance for Malibu Creek over three-time frames: 2004 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year).

Abundance	Minimum	Maximum
Long-term	0	2,245
12-year	0	2,245
5-year	0	32

Table 8c. Minimum and maximum *O. mykiss* (Other *O. mykiss*) abundance for Topanga Creek over three-time frames: 2001 to 2019 (long-term), 2008 to 2019 (12-year), and 2015 to 2019 (5-year).

Abundance	Minimum	Maximum
Long-term	34	316
12-year	34	316
5-year	34	160

Table 9. Long-term, medium-term, and short-term geometric mean abundance of *O. mykiss* (Other *O. mykiss*) in the Santa Monica Mountains BPG. Data used are the sum of the average number of *O. mykiss* observed per month.

Population	Years	Long-term geometric Mean	Years	12-year geometric mean	Years	5-year geometric mean
Arroyo Sequit Creek ¹	2005-2019	NA	2008-2019	NA	2015-2019	NA
Malibu Creek	2004-2019	55.9	2008-2019	52.6	2015-2019	6.1
Topanga Creek	2001-2019	94.2	2008-2019	100.1	2015-2019	70

¹ Insufficient data to produce meaningful results.

4.4.2.4 Mojave Rim and Santa Catalina Gulf Coast BPG

We were unable to calculate geometric mean abundance estimates for either the Mojave Rim or Santa Catalina Gulf Coast BPG due to the lack of long-term data. See Sections 4.3.4, 4.4.1.4, 3.3.5 and 3.4.1.5 for more information on adult steelhead and *O. mykiss* distribution and abundances in these two BPG.

4.4.3 Trend Analysis

Trends were calculated as the slope (β_1) of the regression of log-transformed abundance against years. A value of one was added to the number of Southern SH/RT before the log-transformation to address any zero values if they were present in the dataset [i.e., $\ln(N_a + 1)$]. Using methods from Good et al. (2005), the linear regression can be expressed as:

$$\ln(N_a + 1) = \beta_0 + \beta_1 X + \epsilon$$

Where N_a is annual adult steelhead abundance, β_0 is the intercept, β_1 is the slope of the equation, and ϵ represents the random error term. Population trend, T , for the specified time series was expressed as the exponentiated slope from the regression above:

$$\exp(\beta_1)$$

with 95% confidence intervals calculated as:

$$\exp(\beta_1) \pm t_{0.05(2),df} s_{b1}$$

where b_1 is the estimate of the true slope, β_1 , $t_{0.05(2),df}$ is the two-sided t-value for a confidence level of 0.95, df is equal to $n-2$, n is the number of data points in the time series, and s_{b1} is the standard error of the estimate of the slope, b_1 (Good et al. 2005). We converted the slope to percent annual change (Busby et al. 1996), calculated as:

$$100 * (\exp(\beta_1) - 1)$$

Negative trend values indicate declining abundances over time, whereas positive values indicate growth of the population. Slopes significantly different from zero ($P < 0.05$) were noted.

4.4.3.1 Monte Arido Highlands BPG

We calculated adult steelhead and *O. mykiss* population trends for the Santa Ynez, Ventura, and Santa Clara rivers; however, due to lack of monitoring data we were unable to calculate trends for the Santa Maria River adult steelhead and *O. mykiss* populations (Tables 10 and 11). All three adult steelhead populations have declining trends in abundance for their respective data series and the decline in the Ventura River population is statistically significant ($p=0.03$). Our trend estimates are consistent with other recently reported trend estimates for the Monte Arido Highlands BPG (Boughton et al. 2022a). Similarly, all three *O. mykiss* populations have declining trends in abundance with significant declines observed on the Santa Ynez ($p=0.03$) and Ventura ($p=0.05$) rivers (Table 11).

Table 10. Trends in adult steelhead abundance using slope of ln-transformed time series counts for three Monte Arido Highland BPG populations. Missing years of data were eliminated and not interpolated in any way. Bolded trend values were found to be significant ($p < 0.05$).

Population	Years	Trend (%/year) ¹	Lower 95% CI	Upper 95% CI
Santa Ynez River ¹	1995-2021	-2.24	-6.12	1.59
Ventura River	2006-2021	-7.54	-13.77	-0.86
Santa Clara River	1994-2018	-2.29	-4.99	0.49

¹ No data 2013, Biological Opinion Incidental Take provisions have been required since 2014.

Table 11. Trends in *O. mykiss* (Other *O. mykiss*) abundance using slope of ln-transformed time series counts for three Monte Arido Highland BPG populations. Missing years of data were eliminated and not interpolated in any way. Bolded trend values were found to be significant ($p < 0.05$).

Population	Years	Trend (%/year) ¹	Lower 95% CI	Upper 95% CI
Santa Ynez River ¹	1995-2021	-8.81	-15.98	-1.03
Ventura River	2006-2021	-19.39	-34.89	-0.20
Santa Clara River ²	1994-2018	-6.09	-18.03	7.58

¹ No data 2013, Biological Opinion Incidental Take provisions have been required since 2014.

² No data 2005

4.4.3.2 Santa Monica Mountains BPG

Both Topanga and Malibu Creek populations have a declining but non-significant trend in adult abundance (Table 12). The trend estimates reported here are consistent with recently reported trend estimates for Topanga and Malibu creeks (Boughton et al. 2022a).

The Malibu Creek *O. mykiss* population has experienced a statistically significant ($p = 0.002$) average declining trend in abundance of approximately 26% per year from 2004-2019 (Table 13). The average trend in adult *O. mykiss* abundance for the Topanga Creek population also suggests a decline from 2001-2019; however, the trend is not statistically significant.

Table 12. Trends in adult steelhead abundance using slope of ln-transformed time series counts for the Santa Monica Mountains BPG populations. Missing years of data were not included. Bolded trend values were found to be significant ($p < 0.05$).

Population	Years	Trend (%/year)	Lower 95% CI	Upper 95% CI
Arroyo Sequit ¹	2001-2019	NA	NA	NA
Topanga Creek	2001-2019	-1.70	-5.76	2.54
Malibu Creek	2004-2019	-1.41	-8.49	6.22

¹ Insufficient data to produce meaningful results.

Table 13. Trends in *O. mykiss* (Other *O. mykiss*) abundance using slope of ln-transformed time series counts for the Santa Monica Mountains BPG populations. Missing years of data were not included. Bolded trend values were found to be significant ($p < 0.05$).

Population	Years	Trend (%/year)	Lower 95% CI	Upper 95% CI
Arroyo Sequit ¹	2005-2019	NA	NA	NA
Malibu Creek	2004-2019	-25.56	-37.19	-11.79
Topanga Creek	2001-2019	-1.24	-6.44	4.25

¹ Insufficient data to produce meaningful results.

4.4.3.3 Conception Coast, Mojave Rim, and Santa Catalina Gulf Coast BPGs

We were unable to calculate trends for populations of Southern SH/RT in the Conception Coast, Mojave Rim, and Santa Catalina Gulf Coast BPGs due to lack of available data, with the exception of Arroyo Hondo Creek *O. mykiss*. The analysis of the Arroyo Hondo Creek *O. mykiss* population counts from seven years of bankside observations conducted during winter redd surveys indicate a declining trend in *O. mykiss* abundance, but the trend is not statistically significant ($p = 0.71$).

Many watersheds in the Mojave Rim and Santa Catalina Gulf Coast BPGs likely supported intermittent Southern SH/RT populations characterized by repeated local extinctions and recolonization events in dry and wet years, respectively (NMFS 2012a). The sporadic and intermittent nature of these populations preclude the ability to effectively analyze trends in abundance. Furthermore, many adult steelhead populations occurring south of the Santa Monica Mountains are considered severely reduced and, in many instances, extirpated (Boughton et al. 2005).

4.5 Productivity

Productivity or population growth rate provides important information on how well a population is “performing” in the habitat it occupies throughout its life cycle. Productivity is a key indicator of a population’s viability in terms of its long-term trends in abundance and the ability for it to recover after short-term disturbances (Boughton et al. 2022b). Productivity and abundance are closely linked metrics as a population’s growth rate should be sufficient to maintain its abundance above viable levels (McElhany et al. 2000).

A population’s cohort replacement rate (CRR) is defined as the rate at which each subsequent cohort or generation replaces the previous one (NOAA 2006). Data for adult steelhead in southern California contain too many years of zero observations to effectively calculate a CRR; therefore, we did not attempt to estimate this ratio. We calculated the CRR for *O. mykiss*

populations in the Santa Ynez, Ventura, and Santa Clara rivers, as well as Malibu and Topanga creeks to account for the possibility of some individuals from these populations contributing to the anadromous life-history form. These watersheds were also selected because there was sufficient data (i.e., years with nonzero data) to produce CRR estimates.

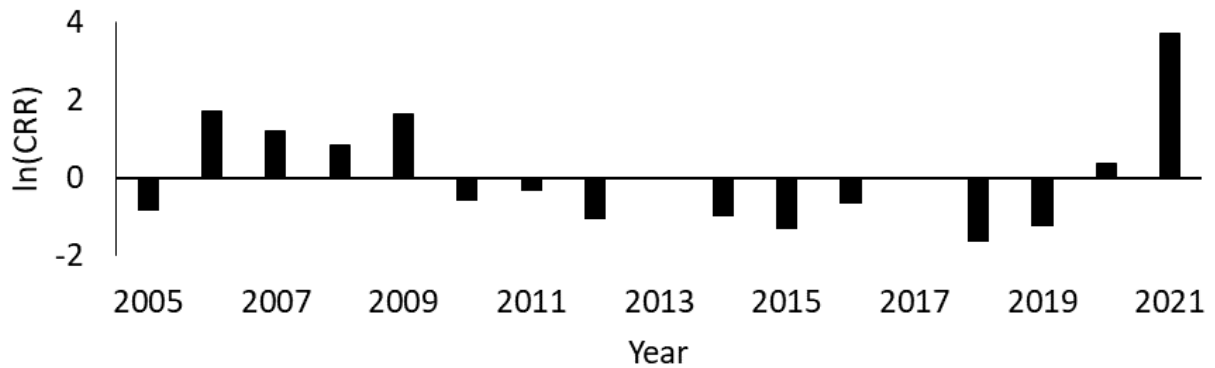
The CRR is defined as:

$$CRR = \ln (N_{t+4}/N_t)$$

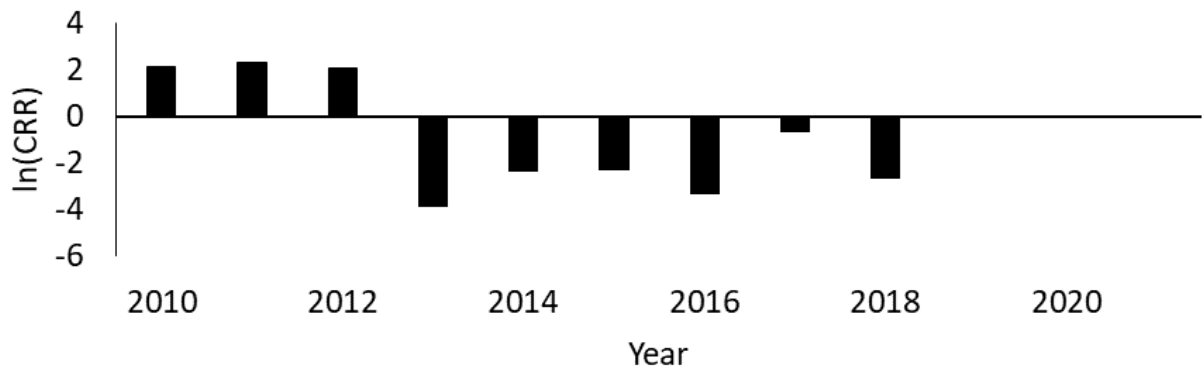
Natural log transformed CRRs greater than zero indicate that the cohort increased in size that year in relation to the brood year three years earlier, whereas a CRR less than zero indicates that the cohort decreased in size. This analysis assumes a generation time of four years, which has been determined to be reasonable based off our best understanding of the Pacific steelhead fluvial-anadromous life-history (NMFS 2012a; Shapovalov and Taft 1954). However, it is important to note that not all Southern SH/RT will return and spawn at age 4, and there is likely considerable variation in age structure (1-4 years) within individual populations (Boughton et al. 2022b).

Over the entire time series, CRR values for the Santa Ynez, Ventura, and Santa Clara River *O. mykiss* populations were more negative than positive (Figure 13). Negative CRRs most frequently occurred from 2013-2018, which coincide with the most recent extreme drought period and associated drought-related low flow conditions. The Santa Ynez River population may be rebounding, as indicated by a high CRR in 2021. Topanga Creek had more positive CRRs than negative, however, 89% of the years with positive values occurred prior to 2012. The CRRs on Topanga Creek are consistent with a recent study that found a significant decline of the abundance of all life stages of *O. mykiss* due to the 2012-2017 drought (Dagit et al. 2017). Population growth rates on Malibu Creek appear to be declining as CRR values have been negative since 2012.

A.



B.



C.

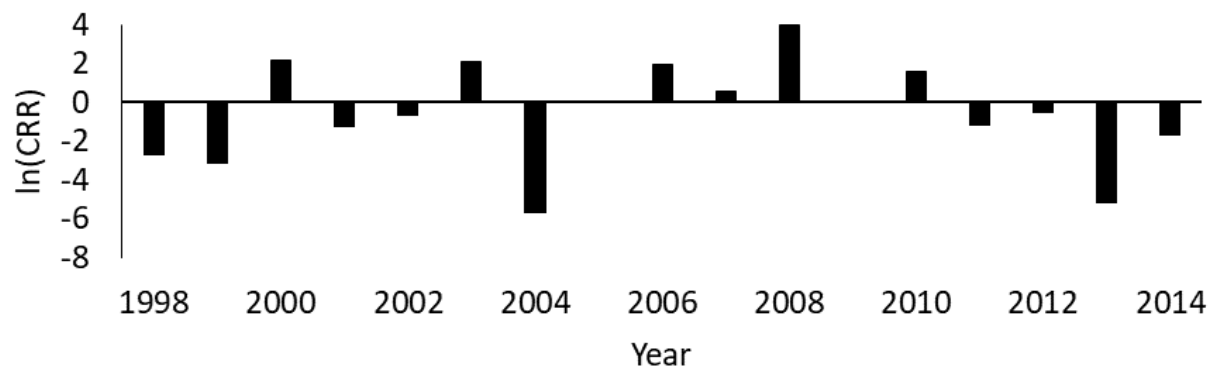
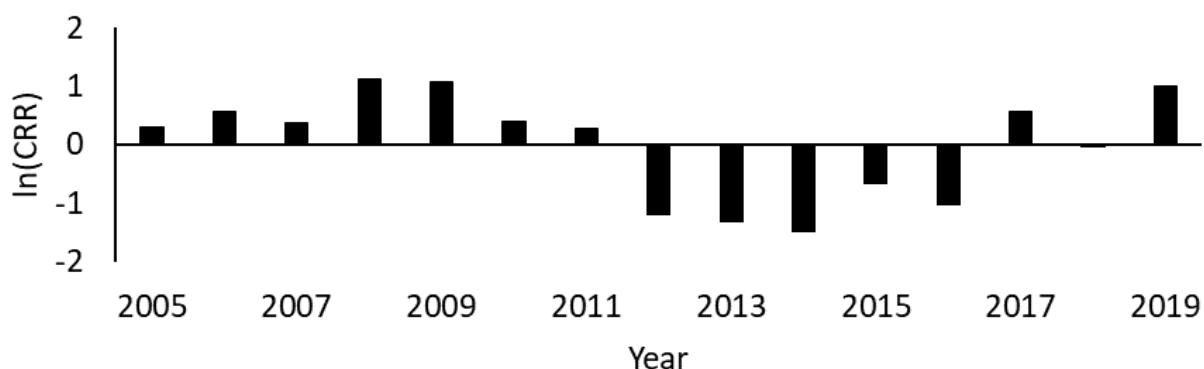


Figure 14a. Ln-Cohort Replacement Rates for *O. mykiss* (Other *O. mykiss*) populations, A) Santa Ynez River, B) Ventura River, and C) Santa Clara River; Biological Opinion Incidental Take provisions have been required since 2014. Gaps are a result of missing years of data. Note different scales on the Y-axis.

D.



E.

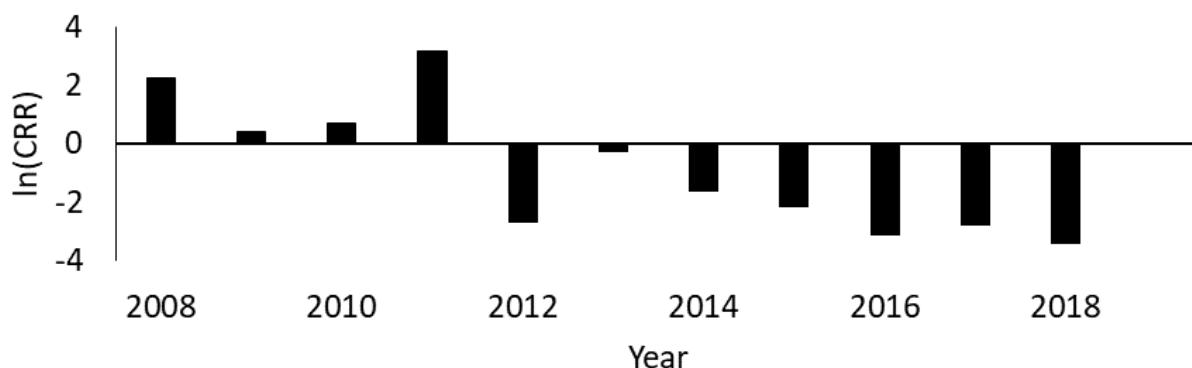


Figure 14b. Ln-Cohort Replacement Rates for *O. mykiss* (Other *O. mykiss*) populations, D) Topanga Creek, and E) Malibu Creek. Gaps are a result of missing years of data. Note different scales on the Y-axis.

4.6 Population Spatial Structure

Population spatial structure refers to the spatial distribution of individuals in the population and the processes that generate that distribution. Population spatial structure is a function of habitat quality, spatial configuration, and dispersal rates of individuals within different habitat types. Spatial structure reflects the extent to which a population's abundance is distributed among available or potentially available habitats at any life stage. All else being equal, a population with low abundance is likely to be less evenly distributed within and among watersheds and is more likely to experience extinction from catastrophic events. Furthermore, populations with low abundance have a reduced potential to recolonize extirpated populations.

Numerous discrete and spatially dispersed but connected populations are required to achieve long-term persistence of Southern SH/RT (NMFS 2012a). Though we cannot specifically classify the spatial structure necessary to maintain Southern SH/RT viability with certainty, examining

similarities and differences between their historical and current spatial distribution can provide a better understanding of their present extinction risk. Southern SH/RT historically occupied at least 46 watersheds in southern California, but currently, only 37-43% of these watersheds are thought to still be occupied (NMFS 2012a). This finding not only highlights the severe contraction of the distribution and abundance of Southern SH/RT in their range, but also indicates that they are prone to range-wide extinction due to several factors such as low population growth rate, loss of genetic diversity, and the limited number of sparsely distributed individuals that may be necessary to recolonize extirpated neighboring populations.

The truncated Southern SH/RT spatial structure observed today can be attributed to the presence of numerous dams, artificial barriers, other instream structures, and groundwater extraction that have long impeded migration and access to high quality upstream habitat throughout southern California (NMFS 2012a). Dams and other barriers not only restrict access to upstream spawning and rearing habitat, but also prevent important ecological and genetic interactions with *O. mykiss* from occurring both upstream and downstream of the total barrier. Isolated *O. mykiss* populations containing ancestry of native Southern SH/RT continue to persist above barriers in approximately 77% of watersheds where the anadromous component has been lost below the barrier (Nielsen et al. 1997; Boughton et al. 2005; Clemento et al. 2009). The impact of dams and other artificial barriers is especially notable on the large rivers and small coastal streams in the northern portion of Southern SH/RT's range. For example, Cachuma, Gibraltar, and Juncal dams on the Santa Ynez River block access to at least 70% of historical spawning and rearing habitat within the watershed. Matilija and Casitas dams located on Matilija and Coyote creeks, respectively, restrict access to 90% of the available spawning habitat in Ventura River watershed. Similarly, Santa Felicia and Pyramid dams on Piru Creek block access to all upstream spawning habitat on this major tributary of the Santa Clara River. On Malibu Creek, the Rindge Dam and Malibu Lake dam blocks access to over 90% of historical anadromous spawning and rearing habitat within the watershed (NMFS 2012a).

Historically, the lower and middle reaches of streams in southern California were used as both migration corridors to higher quality upstream habitat and juvenile rearing habitat in stream reaches that maintained perennial surface flows (Moore 1980). Today, these reaches are the only remaining accessible spawning habitat for Southern SH/RT and are characterized by high urban densities, channelization, impaired stream flows, instream diversions, groundwater extraction, and habitat that generally favors non-native fishes (NMFS 2012a). Furthermore, habitat loss and fragmentation has led to the loss of habitat diversity (i.e., riparian cover, instream habitat structure), which has prevented fish from utilizing these once connected and intact habitats.

The current distribution of Southern SH/RT across its range is inadequate for their long-term persistence and viability (NMFS 2012a). The majority of watersheds in southern California contain dams and artificial barriers that restrict access to high quality upstream spawning and rearing habitat. Barriers to migration isolate and prevent ecological interactions with upstream native *O. mykiss* that would otherwise have the potential to be anadromous. Population level impacts include increased susceptibility to local extirpation due to natural demographic and environmental variation and the loss of genetic and life-history diversity (NMFS 2012a). Range-wide, the historically widespread Southern SH/RT are now sparsely distributed across the landscape with significant reductions in abundance. The degraded spatial structure of Southern SH/RT threatens the viability of the population because extinction rates of individual sub-basin populations are likely much higher than the rate of the formation of new populations from recolonization (McElhany et al. 2000). This is especially relevant for populations occurring in watersheds south of the Santa Monica Mountains; originally, these watersheds supported infrequent Southern SH/RT populations that were likely characterized by repeated local extinction and recolonization events by either neighboring watersheds or from resident populations in upstream drought refugia in dry and wet cycles.

4.7 Diversity

Diversity refers to the phenotypic (e.g., life-history diversity) and genetic characteristics of a population. Life-history diversity allows populations to utilize a wide array of habitats and confers resilience against short-term spatial-temporal variation in the environment. Genetic diversity affects a population's ability to persist during long-term changes in the environment due to both natural and anthropogenic influences. The variation in the life history characteristics in any given population are typically the result of its genetic diversity interacting with environmental conditions. Populations lacking genetic diversity may not have as many genetic "options" to generate new or modified life history types in the face of changing environmental conditions, since natural selection may favor new or different genetic variants. As such, a genetically depauperate population that may be well adapted to the current steady state could be maladapted to new environmental conditions. The combination of both diversity types in a natural environment provides populations with the ability to adapt to long-term changes and be more resilient to these changes over both short- and long-term time scales (McElhany et al. 2000).

Our analysis in Section 4.4 demonstrates declines in *O. mykiss* populations across much of its southern California coast range and preserving Southern SH/RT life-history strategies and adaptations is a critical component for the recovery of the Southern California Steelhead DPS (NMFS 2012a). Ideally, all three Southern SH/RT life-history types (i.e., fluvial-anadromous, freshwater-resident, lagoon-anadromous) would be expressed within a single population, or

the population would harbor the underlying genetic variation to express those life-history types when environmental conditions allow. The freshwater-resident life-history type is still present in many populations of Southern SH/RT; however, this form frequently occurs in the isolated upper reaches of the watershed where opportunities for gene flow with anadromous fish are prevented by barriers to migration. Bond (2006) demonstrated accelerated growth rates of juvenile *O. mykiss* expressing the lagoon-anadromous life-history form. Larger size at ocean entry is thought to enhance marine survival and improve adult returns (Bond 2006); however, it is unlikely that this life-history form is currently viable, because approximately 75% of estuarine habitat in southern California has been lost, and the remaining intact habitats are constrained by agricultural and urban development, highways, and railroads, and threatened by sea level rise and invasive species (NMFS 2012a). The artificial breaching of lagoons also poses a significant threat to the lagoon-anadromous life-history form as a recent study observed considerable mortality of Southern SH/RT directly after artificial breaching (Swift et al. 2018). As presented in Section 4.4, the anadromous form of Southern SH/RT still occurs in very low abundances in a limited portion of their historical range. The preservation of this life-history component will require substantial habitat restoration and modifications or removal of the numerous artificial barriers that currently restrict access to upstream high-quality spawning habitat (NMFS 2012a).

Several recent studies highlight the important role that genetic factors have in determining the life-history expression of coastal steelhead. Pearse et al. (2014) identified two *Omy5* haplotypes linked to the anadromous (“A”) and resident (“R”) life-history forms whereby “AA” and “AR” genotype are more likely to be anadromous than the “RR” genotype (Pearse et al. 2019). Rundio et al. (2021) found that age 1+ juveniles with “RR” and “AR” genotypes experienced higher growth rates than fish with the “AA” genotype, and that overall condition was slightly higher in future resident fish than in future smolts, particularly among resident males. The divergence of the “A” and “R” haplotypes in Southern SH/RT populations is influenced by the presence of numerous artificial barriers in southern California, which act as a strong selection pressure against the “A” haplotype in above-barrier populations. For example, on the Santa Clara River, the Vern Freeman Diversion Dam and other instream diversions have limited upstream fish passage to spawning and rearing habitat on its tributaries, Sespe and Santa Paula creeks (NMFS 2012a). Populations of *O. mykiss* from both tributaries were found to display moderately high frequencies of the “R” haplotype (Pearse et al. 2019). Relative frequencies of the “R” and “A” haplotypes can also be altered in populations that have become introgressed with other strains of Rainbow Trout that may have much different haplotype frequencies.

The recognition of the “A” and “R” haplotypes provide insight on the genetic integrity and viability of Southern SH/RT. The frequency of the anadromous haplotype may substantially decline during periods of adverse conditions due to the low predicted survival of migrating

smolts (i.e., “AA” and “AR” individuals). Likewise, “RR” and “AR” residents may be favored during adverse conditions, which could eventually lead to declines of the “A” haplotype over time and the gradual loss of the “AA” genotype from the population. Without considerable restoration of habitat connectivity through the removal of artificial barriers, the “A” haplotype in “AR” individuals in isolated populations above barriers is expected to be slowly lost over time (Apgar et al. 2017). While “AR” smolts may produce “AA” individuals when favorable migration conditions continue and retain the “A” haplotype in resident populations, it is unclear that the resident component can reliably produce anadromous fish after prolonged periods of unfavorable conditions in the long term (Boughton et al. 2022a). Furthermore, climate change projections for Southern SH/RT range predict an intensification of typical climate patterns such as more intense cyclic storms, drought, and extreme heat (NMFS 2012a). These projections suggest that Southern SH/RT will likely experience more frequent periods of adverse conditions and continued selection pressure against the anadromous life-history form.

4.8 Conclusions

This section summarizes the abundance, trends, and productivity analyses. Because quantitative analyses were not conducted for population spatial structure and diversity, we do not provide conclusions for these metrics as the qualitative discussions in Sections 4.6 and 4.7 provide sufficient detail and information.

4.8.1 Abundance and Trends

The data evaluated indicate an overall long-term declining trend of Southern SH/RT with critically low range-wide abundances. In the past decade, adult abundance counts have not been greater than ten for any watershed examined, and most streams have observed no adult returns during this time period. For the Monte Arido Highlands BPG, which is thought to be a potential source population for smaller coastal watersheds such as the Conception Coast BPG, only a single adult has been observed returning in the past five years. For each of the three populations analyzed, the data for this BPG shows a long-term declining trend in adult abundance. The steepest decline occurred in the Ventura River population, for which a statistically significant -7.54% per year was observed.

The data evaluated for the Santa Monica Mountains BPG indicate that these watersheds support small but consistent runs of adult steelhead ranging from zero to five individuals per year. However, like other salmonid-supporting streams in the Southern SH/RT range, few adults have been observed in the past five years, and it is unlikely that these streams historically supported large runs of Southern SH/RT due to their small size. The data also show declining but not statistically significant trends in adult abundance for Malibu and Topanga creeks. The Department's South Coast Region staff have not observed any *O. mykiss* in Malibu Creek since

before the Woosley fire in 2018, which suggests that Southern SH/RT have been effectively extirpated below Rindge Dam (D. St. George, CDFW, personal communication). A combined total of five adults have been observed for the Conception Coast, Mojave Rim, and Santa Catalina Gulf Coast BPGs since 2017 (Dagit et al. 2020). Our finding of generally declining trends in the abundance of adult steelhead is consistent with the results of a recent viability assessment for the southern California Coast Domain produced by Boughton et al. (2022a).

O. mykiss trends also demonstrate measurable declines in overall abundance. Maximum abundance and long-term averages of *O. mykiss* have declined in all three Monte Arido Highland populations. Similarly, all populations in this BPG show declining trends in *O. mykiss* abundance with statistically significant declines of -8.81% and -19.39% per year on the Santa Ynez and Ventura rivers, and a non-statistically significant decline of -6.09% on the Santa Clara River. Within the Santa Monica Mountains BPG, both Malibu and Topanga creek *O. mykiss* populations have experienced a long-term decline. The *O. mykiss* population in Topanga Creek appears to be more viable than Malibu Creek as our results indicate only a small long-term decline. Our results indicate a trend of -25.56% per year on Malibu Creek, which is the steepest average annual decline for any of the Southern SH/RT populations that we analyzed.

The most recent prolonged drought from 2012-2017 correlates with significant reductions of all life-history forms and stages of Southern SH/RT. Drought conditions are associated with the loss of suitable spawning and rearing habitat, insufficient instream flows required for migration, diminished water quality, reductions in available food supply, and increases in direct mortality due to predation and stranding (Dagit et al. 2017). Our analyses show a relatively consistent range-wide pattern of higher abundances prior to 2012, followed by consecutive years of lower abundances starting at the onset of the drought. It appears that few populations have rebounded from the drought as current abundance estimates remain low relative to pre-drought conditions. The recovery of Southern SH/RT will likely depend on the successful recruitment of downstream migrants from upstream resident populations in refugia habitats. However, virtually all refugia populations are currently above impassable barriers. Furthermore, many southern California watersheds do not contain upstream drought refugia. In these instances, recolonization from source populations in other watersheds is likely the only mechanism for these populations to rebound (Boughton et al. 2022a).

Boughton et al. (2007) established a precautionary run size criteria for the southern California Coast Domain of 4,150 spawners per year to provide a 95% chance of persistence of the watershed's population over the next 100 years. While this goal may not be feasible for many of the smaller coastal watersheds in southern California, NMFS (2012) speculated that this target may be more feasible for the larger watersheds (i.e., Monte Arido Highland BPG). Even if we applied a lower criterion of 834 spawners (Boughton et al. 2022a), the results of our

analyses demonstrate that no population is near the criteria necessary to provide resilience from extinction.

It is important to highlight limitations of our analyses. First, our analysis may underestimate the true abundance of adult steelhead because data analyzed for this effort are usually collected during periods of high stream flows and turbidity, making monitoring difficult to conduct (Dagit et al. 2020). Second, the data used in this effort are derived from various single-basin monitoring efforts, each of which utilize different survey designs and approaches. Thus, we were required to interpret the data as reported, while recognizing the potential limitations in making inter-watershed comparisons in instances where the data were from various monitoring efforts that did not necessarily meet standards established by the Department's California Coastal Monitoring Program (CMP). Third, the lack of any monitoring of most watersheds occurring south of the Santa Monica Mountains inhibited our ability to make definitive and comprehensive range wide conclusions on Southern SH/RT abundance and trends. However, it is likely that abundance estimates for many watersheds in the southern portion of the range are so low that obtaining accurate estimates would remain difficult even with increased monitoring.

4.8.2 Productivity

The results of our CRR analysis for *O. mykiss* on the Santa Ynez, Ventura, and Santa Clara rivers show more years of negative than positive CRR values. Negative CRR values were observed during the 2012-2017 drought period for all populations. However, the most recent 2021 estimate for the Santa Ynez population was positive, which may suggest a rebounding population. CRR values for Topanga Creek were more positive than negative; however, most positive values occurred prior to the onset of 2012 drought conditions. In recent years, Malibu Creek CRR values have been negative, particularly during the 2012-2017 drought period.

While the CRR values for *O. mykiss* do not necessarily reflect true spawner to spawner ratios due to the high likelihood that many observed fish were not actually part of the spawning cohort during that year, our results demonstrate that *O. mykiss* populations occurring below the barrier to anadromy in these watersheds do not appear to be viable because abundances are too low to sustain positive population growth rate on a yearly basis. This result is especially concerning given that the long-term resilience of the anadromous component of Southern SH/RT likely depends on the production of anadromous juveniles from the freshwater-resident life-history form.

5. HABITAT THAT MAY BE ESSENTIAL TO THE CONTINUED EXISTENCE OF SOUTHERN SH/RT

5.1 Migration

Southern SH/RT migration into freshwater is linked with seasonal winter and spring high flows that establish connectivity between the ocean and freshwater spawning areas (NMFS 2012a). Adult steelhead require water depths of at least 18 cm depth for upstream movement; however, 21 cm is considered to be more suitable for upstream passage of all possible sizes of individual fish, because it allows sufficient clearance so that contact with the streambed is minimized (Bjornn and Reiser 1991; SWRCB 2014). Low dissolved oxygen (<5 mg/L) and high turbidity can deter migrating salmonids such as steelhead (Bjornn and Reiser 1991). Delayed migration may also occur when stream temperatures are too high or low (Bjornn and Reiser 1991). Disease outbreaks can occur as a result of extreme high temperatures (Bjornn and Reiser 1991; Spence et al. 1996). Salmonids usually migrate when water temperatures are below 14°C (Spence et al. 1996); however, salmonids can adapt to higher thermal limits when slowly exposed to increased water temperatures over time (Threader and Houston 1983).

Instream structure, like waterfalls, sandbars, and debris jams can act as impediments to upstream fish migration. Steelhead are able to jump a maximum of 3.4 m (Spence et al. 1996) and typically, pool depth must be at least 25% greater than barrier height to achieve the required swimming velocity to pass the barrier (Spence et al. 1996). Pool shape can also influence if a barrier is passable by steelhead. For example, water flow over a steep waterfall into a plunge pool may increase jump height capacity due to upward thrust created by the hydrodynamics within the pool (Bjornn and Reiser 1991). Physical structures such as large woody debris and boulders within streams can offer flow and temperature refuge for resting fish during migration to upstream spawning areas (Spence et al. 1996). Wood structures, overhanging banks, and riparian flora can provide cover to steelhead for protection from terrestrial and avian predators. Deep pools provide important holding habitats for migrating adult salmonids (Chubb 1997).

5.2 Spawning

Habitat attributes necessary for successful spawning include cover, appropriate substrate, cool stream temperatures, and adequate streamflow (Reiser and Bjornn 1979). Salmonids select spawning sites in pool-riffle transitional areas where downwelling or upwelling currents occur that create loose gravel with minimal sediment and litter (Bjornn and Reiser 1991). Rainbow Trout can spawn in a relatively wide range of temperatures, from 2 – 22°C, but may respond to abrupt temperature declines with decreased spawning activity and production (Reiser and Bjornn 1979). Steelhead and Rainbow Trout require gravel substrate of 0.5 – 10.2 cm in diameter to construct their redds and a high proportion of the redd substrate must be

comprised of smaller-sized gravel within this range (Reiser and Bjornn 1979). Cover habitat, which offers protection from predation, can include overhanging banks, riparian or aquatic vegetation, large and small woody debris, rocks, boulders, and other instream features. Having access to cover close to a redd is advantageous for Southern SH/RT and may influence spawning site selection (Reiser and Bjornn 1979). Minimum water depth must be sufficient to cover the spawning fish and, depending on individual fish size, is likely to range from 6-35cm (Bjornn and Reiser 1991).

Steelhead and Rainbow Trout have been documented to spawn in water velocities ranging from 21-117 cm/s (Reiser and Bjornn 1979; Bovee and Milhous 1978). Under moderate water velocities, increasing streamflow leads to a greater amount of covered gravel substrate for spawning; however, if water velocities and associated stream flows are too high, the additional suitable spawning habitat becomes unusable for salmonids and stream spawning capacity declines (Reiser and Bjornn 1979; Bjornn and Reiser 1991). Total suitable spawning area within a stream is dependent on the density and size of spawning fish, water depth and velocity, and amount of appropriately sized gravel substrate available (Bjornn and Reiser 1991). These factors combined drive habitat suitability for steelhead and other salmonids (Bjornn and Reiser 1991).

5.3 Instream Residency

Temperature, dissolved oxygen, salinity, water flow, and water depth are all factors that determine stream habitat suitability for *O. mykiss*. Water temperature is especially critical for survival in southern California, as stream temperature can vary drastically within the span of a single day, sometimes peaking at over 30°C during summer months (Sloat and Osterback 2013). For Southern SH/RT, changes in behavior occur above 25°C, such as decreased feeding or movement into refugia (Ebersole et al. 2001; Sloat and Osterback 2013) and the estimated mortality threshold is 31.5°C (Sloat and Osterback 2013), which is marginally higher than that of more northern steelhead populations (Rodnick et al. 2004; Werner et al. 2005). This increased temperature tolerance indicates that Southern SH/RT may have acclimated to higher temperature conditions; however, it does not necessarily suggest that they have undergone local adaptation with genetic underpinnings (Sloat and Osterback 2013). Dissolved oxygen levels should generally be at or above 5 mg/L for Southern SH/RT survival (Reiser and Bjornn 1979; Bjornn and Reiser 1991; Moyle et al. 2017) but concentrations greater than 7 mg/L are ideal (Moyle et al. 2017). In cooler temperatures, Rainbow Trout can survive in minimal dissolved oxygen levels of 1.5-2.0 mg/L (Moyle 2002).

Adult Rainbow Trout preferentially select habitat in deeper water and can be found in runs or pools close to swift water (Moyle 2002). In such habitats, fish can move into fast water habitat

for feeding and then return to hold and rest in slower water (Moyle 2002). Tobias (2006) found that Southern SH/RT in Topanga Creek exhibited a preference for pools over other habitat types. Trench pools were strongly favored and mid-channel pools and step pools were also selected; however, fish avoided plunge pools, corner pools, and lateral scour pools as well as riffles and cascades. Glides and step runs were neither avoided nor strongly selected.

Resident Rainbow Trout prey on aquatic and terrestrial invertebrates that drift by, both in the water column or on the surface, as well as benthic invertebrates and sometimes smaller fishes (Moyle 2002). Larger stream-dwelling salmonids (>270 mm) often exhibit an ontogenetic niche shift, moving away from consuming invertebrates and depending more on piscivory to achieve efficient growth (Keeley and Grant 2001). Size of invertebrate and fish prey increased with body length (Keeley and Grant 2001). Stomach contents of *O. mykiss* in Topanga Creek revealed that aquatic and terrestrial insects, other invertebrates, and fish comprised most of their diet during fall and spring. Consumption of introduced Arroyo Chub (*Gila arcuati*) by Topanga Creek *O. mykiss* suggests that chub may be an important component of their diet in this stream, particularly during the late fall when aquatic macroinvertebrates may be less available (Krug et al. 2012; Swift et al. 1993).

5.4 Egg and Larval Development and Fry Emergence

Many environmental factors influence salmonid embryo incubation success, including dissolved oxygen, temperature, substrate size and porosity, and extra-gravel and inter-gravel hydrodynamics (Bjornn and Reiser 1991). Inter-gravel dissolved oxygen is particularly important to egg development and insufficient oxygen can lead to high mortality. Dissolved oxygen requirements increase as embryos grow and peaks just prior to hatching (Quinn 2018). Intra-gravel oxygen allows for embryo respiration, and oxygen concentrations of 8 mg/l or more contribute to high survival of steelhead embryos (Reiser and Bjornn 1979).

Water velocity is correlated with the amount of dissolved oxygen available to incubating eggs, and lower water velocity leads to higher embryo mortality (Bjornn and Reiser 1991). Reduced flows can also cause redd dewatering, which may result in egg mortality if there is no subsurface flow (Reiser and White 1983). The settling of fine sediment within gravels used to construct redds can prevent the interstitial flow of water and oxygen, and thus smother and kill embryos and post-hatch alevins (Bjornn and Reiser 1991). Finer sediment particles such as ash from wildfires or dust, are most effective at filling interstitial spaces within the redd substrate and can be a contributor to egg asphyxiation and recruitment failure (Beschta and Jackson 1979; Chapman 1988; Bjornn and Reiser 1991).

In addition to negative impacts from sediment deposition, unsuitable temperatures can have negative effects on embryonic development and survival (Bjornn and Reiser 1991). Higher

temperatures are correlated with faster embryonic growth and development (Kwain 1975; Bjornn and Reiser 1991); however, if temperatures exceed upper suitability thresholds, mortality increases (Kwain 1975; Rombough 1988; Melendez and Mueller 2021). The ideal temperature range for incubation is 7-10°C (Kwain 1975) and incubation temperatures surpassing 15°C can result in considerable embryo mortality (Kwain 1975; Rombough 1988). Faster development and early hatching resulting from elevated temperatures can manifest in substantial reductions in body mass and length of newly hatched alevin (Melendez and Mueller 2021). These environmentally driven developmental changes could have negative implications for predation response and survival (Hale 1996; Porter and Bailey 2007). Alternatively, extremely cold water can induce mortality (Reiser and Bjornn 1979), although water temperatures that are below steelhead tolerances are likely a rare occurrence in southern California streams. Fry emerge in late spring or early summer and incubation time is dependent on water temperature (Moyle et al. 2017; Quinn 2018). Cold water temperatures, or those above 21.1°C, can decrease survival of emerging fry by restricting their ability to obtain oxygen from the water (McEwan and Jackson 1996).

5.5 Rearing and Emigration

Suitable rearing habitats for juvenile *O. mykiss* require adequate water temperature, flow velocity, water depth, dissolved oxygen concentrations, and availability of prey items. Juveniles generally occupy cool, clear, higher velocity riffles which provide cover from predators (Moyle 2002). Rearing juveniles require habitat with sufficient food production such as riffles with gravel substrate (Reiser and Bjornn 1979). Juvenile *O. mykiss* in southern California have been found to rear in both perennial and intermittent streams (Boughton et al. 2009). Intermittent streams are common in the southern California region and can in some cases benefit native fishes and other aquatic organisms that have evolved within these conditions. By seasonally fragmenting watersheds and disconnecting populations of introduced warm-water tolerant species, intermittent stream desiccation can reduce potential predation and competition from invasives. However, these same conditions can also negatively affect steelhead survival through loss of wetted habitat or degraded water quality conditions, prevent adult spawning migrations or juvenile/smolt emigration, and otherwise isolate subpopulations (Boughton et al. 2009).

Preferred water temperatures for juvenile *O. mykiss* range between 15 and 18°C (Moyle 2002), although they can tolerate temperatures up to 29°C if dissolved oxygen concentrations are high and there is an abundant food supply (Dressler et al. 2023; Sloat and Osterback 2013). Southern SH/RT have been observed functioning in stream temperatures outside of the preferred range up to the mid to high twenties (Dressler et al. 2023; Moyle et al. 2017; SYRTAC 2000). For example, the Santa Ynez River was determined to be thermally suitable, albeit thermally stressful, for Southern SH/RT in both normal and warm years, with thermal suitability

characterized as a maximum daily temperature below 29°C and a mean daily temperature below 25°C (Boughton et al. 2015). Temporary or intermittent exposure to temperatures above the upper tolerance limit for salmonids can be tolerated in some populations (Dressler et al. 2023; Johnstone and Rahel 2003), whereas chronic or long-term exposure to high temperatures is typically lethal (Dickerson and Vinyard 1999; Johnstone and Rahel 2003). Additionally, feeding behavior and activity level are generally reduced when fish are temporarily exposed to warmer temperatures that cause thermal stress (Johnstone and Rahel 2003). However, Spina (2007) found that in Topanga Creek, there were no available daytime thermal refugia available for juvenile *O. mykiss*, yet they were able to tolerate temperatures up to 24.5°C without changes in behavior or activity level. These findings may indicate that Southern SH/RT are acclimated to higher daily stream temperatures than more northern *O. mykiss* populations. Juvenile salmonids acclimated to higher water temperatures, such as those in many Southern SH/RT streams, can sustain higher maximum thermal tolerances than those acclimated at lower temperatures (Lohr et al. 1996).

Metabolic demand increases with higher environmental temperatures. Warmer waters can result in faster growth rates where the forage base is abundant or may slow if food is scarce (Noakes et al 1983.; Brett 1971). Thus, freshwater growth is strongly dependent on primary productivity and food accessibility within the stream (NMFS 2012a). In Topanga Creek, juvenile Southern SH/RT had high growth rates during the summer despite temperatures that frequently surpassed known high temperature tolerances (Bell et al. 2011a).

Thermal refugia are especially important for summer rearing, when Southern SH/RT juveniles must find stream reaches that are sufficiently cool (NMFS 2012a). In southern California streams, higher altitude can provide thermal refuge as well as near-coastal areas that benefit from the ocean acting as a temperature sink (NMFS 2012a). Riparian cover is also important for moderating stream temperatures, as exposed or non-shaded streams are generally warmer than those shaded by riparian canopy (Li et al. 1994). These types of shaded, cool-water stream habitats are most frequently found in headwater reaches within the range of Southern SH/RT (NMFS 2012a).

In Sespe Creek, juvenile Southern SH/RT were observed to occupy the coolest areas of pools during daytime hours in summer months (Matthews and Berg 1997). Fish were consistently found congregating in a seep area that provided cool groundwater during the hottest times of day. The juvenile Southern SH/RT appeared to experience a trade-off between dissolved oxygen and water temperature but chose cooler temperatures, deeper within the temperature stratified pools, over higher levels of dissolved oxygen which were closer to the stream surface. In the spring, *O. mykiss* have been found to emigrate downstream into lower mainstem areas when tributaries may become warmer and/or drier (Spina et al. 2005). As flows increase in the

fall and winter, fish may move upstream into tributary habitat to overwinter (Bramblett et al. 2002); however, this behavior has not been confirmed for Southern SH/RT (Spina et al. 2005).

Cover is also an important habitat component for juvenile Southern SH/RT survival, particularly during the winter months. Riparian cover, such as canopy and undercut banks, as well as instream cover like large woody debris (LWD) and deep pools, are important in providing shelter to rearing salmonids (Bjornn and Reiser 1991). Cover quality and availability have been correlated with local instream fish abundance for multiple salmonid species (Bjornn and Reiser 1991). In the mainstem Ventura River, juvenile Southern SH/RT densities were found to be positively correlated with velocity and cover (Allen 2015 p. 133). In western Oregon and Washington streams, juvenile steelhead were found in higher densities in reaches treated with LWD during the winter (Roni and Quinn 2001). Pool formation and enhancement can result from presence of live hardwood or LWD in a stream (Thompson et al. 2008). Instream tree roots can produce scour in high flow conditions leading to long-lasting pools. Trees in the stream channel can also anchor dead LWD and create wood jams. Jams constructed around standing trees are more durable and will last longer in watersheds dominated by hardwood species (Thompson et al. 2008).

Certain substrate types can also provide cover habitat for rearing salmonids. Larger substrate offers interstitial spaces for fish to avoid visual detection from predators. Boulders may be particularly important features in southern California streams, due to the paucity of LWD in these watersheds (Boughton et al. 2009; Tsai 2015). Boulders can assist in the formation of pools and create habitat complexity, which increases habitat suitability for Southern SH/RT (Roni et al. 2006; Tsai 2015). The presence of boulders in streams can also have a significant positive effect on *O. mykiss* survival and abundance due to their role in providing hiding areas and refuge from winter storms and associated flows (Tsai 2015). In contrast, areas with increased stream substrate embeddedness (more compacted stream bottoms) have been associated with lower juvenile salmonid densities (Bjornn and Reiser 1991).

Some Southern SH/RT will remain in freshwater through their life cycle, while those expressing the anadromous life history strategy will begin migrating downstream towards the ocean after two to three years of rearing in freshwater (NMFS 2012a). It is common in southern California for seasonal lagoons to be formed during the summer due to decreased stream flows and the natural accumulation of a sand berm at the point where the stream meets the ocean. Some juveniles take advantage of rearing in the warmer lagoon environment to achieve greater size prior to entering the ocean, which allows them a greater chance of survival (Bond et al. 2008; Hayes et al. 2008).

In Scott Creek (central California), during years when a seasonal lagoon formed, growth rates were 2-6 times greater for steelhead rearing in the estuary-lagoon than those in the cooler, less productive upstream habitat (Hayes et al. 2008). Juvenile *O. mykiss* in central California streams have been observed to exhibit a lagoon-anadromous, or “smolting” twice, life history strategy. These life history variants travel downstream to the closed estuary to rear during the summer, then migrate back upstream into more suitable conditions when the estuary starts to become less hospitable (Hayes et al. 2011; Huber and Carlson 2020). Juvenile *O. mykiss* also preferentially seek out areas with higher water quality when confined within a seasonally closed estuary (Matsubu et al. 2017). However, estuaries in poor condition, including lagoons with poor water quality, may lead to mortality of rearing juveniles if they do not have access to suitable habitat upstream. Seasonal lagoons in southern California typically do not reconnect to the ocean until the first rainfall occurs in the fall or winter (Booth 2020). Juvenile *O. mykiss* benefit from pulse flows initiated by storms and successful emigration is largely dependent on storm flow events matching the timing of *O. mykiss* smolt outmigration (Booth 2020). Smolts in southern California streams, such as the Santa Clara River are largely unable to take advantage of lagoon rearing and its associated benefits due to poor water quality in the estuary and dry reaches upstream (Booth 2020).

5.6 Ocean Growth

Little information exists specific to ocean growth of anadromous Southern SH/RT, but data from other west coast steelhead populations can provide some insight into habitat requirements of this life stage. Steelhead exhibit early ocean migratory behavior that is thought to maximize bioenergetic efficiency (Atcheson et al. 2012). In contrast to other Pacific salmon species, which typically remain relatively close to shore and feed in coastal waters along the continental shelf during their first summer at sea, steelhead quickly leave these productive coastal habitats for the open ocean (Atcheson et al. 2012; Daly et al. 2014). Many California steelhead juveniles spend only a few months feeding in the California Current Ecosystem (CCE) before they migrate northwest to cooler waters offshore (Daly et al. 2014). In the open ocean, steelhead maximize their energy intake by consuming high-energy prey items like fish and squid at moderate rates rather than consuming lower-energy food resources at high rates (Atcheson et al. 2012). Fish and squid make up a substantial portion of the juvenile steelhead diet for those rearing in the Gulf of Alaska, which serves as an important rearing location for west coast steelhead (Atcheson et al. 2012).

While feeding and growing in the ocean, steelhead typically occupy waters within the temperature range of 6-14°C (Hayes et al. 2016; Quinn 2018). Steelhead exhibit strong thermal avoidance, remaining within a narrow range of suitable sea surface temperatures (SSTs) during their ocean foraging and migrations, generally within 20 meters of the surface (Burgner et al.

1992 in Atcheson et al. 2012; Nielsen et al. 2010). Deviations outside of their thermal tolerance have negative consequences for growth and survival in the ocean (Atcheson et al. 2012) and generally poor ocean conditions can negatively affect survival especially during early ocean residence (Kendall et al. 2017). For example, warm SSTs were associated with lower post-smolt survival of Keogh River steelhead off the coast of Alaska (Friedland et al. 2014). In recent years, the CCE experienced a severe marine heatwave (Di Lorenzo and Mantua 2016), which impacted species abundance and distribution at multiple trophic levels, including the prey base for Pacific salmon (Daly et al. 2017; Peterson et al. 2017). During years with anomalously warm ocean conditions, young Chinook Salmon were observed to be much thinner, and their survival rates were depressed compared to years with cooler ocean temperatures, likely resulting from this shift in availability of prey species (Daly and Brodeur 2015; Daly et al. 2017).

Steelhead average a travel distance in the ocean of 2,013 km but have been tracked traveling up to 5,106 km (Quinn 2018). Steelhead are not typically captured in commercial fisheries possibly resulting from their swift movement offshore, and most catches of steelhead in research trawls are in the upper 30 meters of the water column (Moyle et al. 2017; Quinn 2018).

6. FACTORS AFFECTING THE ABILITY TO SURVIVE AND REPRODUCE

6.1 Changes in Ocean Conditions

The long-term relationship between ocean conditions, food web structure, and Southern SH/RT productivity is not well understood; however, these relationships have been examined for steelhead populations in the Pacific Northwest. While the Pacific Northwest coastal rivers are distant from the coastal rivers of southern California in terms of both geography and ecology, these findings still improve our understanding of the relationship between ocean temperatures and the dietary composition and morphology of west coast steelhead populations. Comparisons may also offer insights into similar mechanisms that may potentially influence Southern SH/RT ocean diet compositions. Thalmann et al. (2020) detected significant differences in the prey items consumed by juvenile steelhead during warm ocean years compared to average or cold ocean years. They also found significant interannual variability in stomach fullness, with significantly lower than average stomach fullness associated with warm ocean years. Steelhead sampled during warmer years were thinner, on average, than those sampled during cooler years. In 2015 and 2016, when ocean conditions were anomalously warm, there was limited availability of cold-water prey species with higher energetic and lipid content. Although some level of plasticity was demonstrated in the juvenile steelhead diet, consumption of lower-quality prey items likely led to reduced growth and poorer body condition during those years (Thalmann et al. 2020).

In the North Pacific, the 2013–2020 period was characterized by exceptionally high sea surface temperatures coupled with widespread declines and low abundances for many west coast salmon and steelhead populations (Boughton et al. 2022a). For example, the abundance of southern Chinook salmon and steelhead populations reached very low counts between 2014 and 2019, leading to the designation of many stocks as overfished (PFMC 2020). Increased sea temperatures and associated impacts have resulted in a significant biological response at all trophic levels, from primary producers to marine mammals and birds.

6.2 Effects of Climate Change

The climate of the United States is strongly connected to the changing global climate (USGCRP 2017), and temperatures are projected to continue to rise another 2°F (1.11°C) to 4°F (2.22°C) in most areas of the United States over the next few decades (Melillo et al. 2014). The waters of the United States are projected to lose between 4 and 20% of their capacity to support cold water-dependent fish by the year 2030 and as much as 60% by 2100 due to climate change and its impacts (Eaton and Scheller 1996). The greatest loss of this important aquatic habitat capacity is projected for California, owing to its naturally warm and dry summer climate (O’Neal 2002; Preston 2006; Mote et al. 2018). The recent multidecadal (2000–2021) “megadrought” in the southwestern U.S., including California, has been the driest 22-year period over the past 1,000 years in this region (OEHHA 2022). Severe drought was documented across much of the southwest during this period, with record-breaking low soil moisture, extended heat waves, reduced precipitation, and intensifying weather extremes (Garfin et al. 2013; OEHHA 2022; Williams et al. 2022). These conditions are expected to continue or increase in the region (Gershunov et al. 2013), with predicted outcomes dependent upon the level and extent of human efforts to address and offset CO₂-driven climate change impacts, both within the United States and across the globe (Overpeck et al. 2013; NMFS 2016; USGCRP 2017; OEHHA 2022).

Since 1895, California has warmed more than both the North American and global temperature averages (NOAA 2021; OEHHA 2022). As such, the state is considered one of the most “climate-challenged” areas in North America (Bedsworth et al. 2018), facing increasingly extreme weather patterns and comparatively rapid shifts in regional climate- and local weather-based averages and trends (e.g., Overpeck et al. 2013; Pierce et al. 2018). California’s temperatures have paralleled global trends in terms of increasing at an even faster rate since the 1980s (Figure 15; OEHHA 2022). The past decade has been especially warm; eight of the ten warmest years on record for California occurred between 2012 and 2022 (OEHHA 2022). In general, the portions of California with lower latitudes and elevations will be subject to the greatest increase in duration and intensity of higher air and water temperatures due to climate change (Wade et al. 2013). Thus, the southwestern part of California, which includes the range of Southern SH/RT, will likely face disproportionate climate change-related impacts when compared to

other regions of the state. Southern SH/RT are, therefore, likely to face more severe and challenging conditions than their northern salmonid relatives.

The broad-scale climatic factors that appear to primarily shape the habitat suitability and population distribution of Southern SH/RT are summer air temperatures, annual precipitation, and severity of winter storms (NMFS 2012a). These factors and their influences on the landscape are predicted to intensify under long-term, synergistically driven conditions brought about by climate change. They are also expected to exacerbate existing stressors for Southern SH/RT and other cold water-dependent native aquatic organisms in stream and river systems in southern California (NMFS 2012b). In a comprehensive rating of California native fish species, Moyle et al. (2013) determined southern California steelhead to be “critically vulnerable” to climate change and likely to go extinct by 2100 without strong conservation measures. This was reaffirmed by an analysis conducted by Moyle et al. (2017).

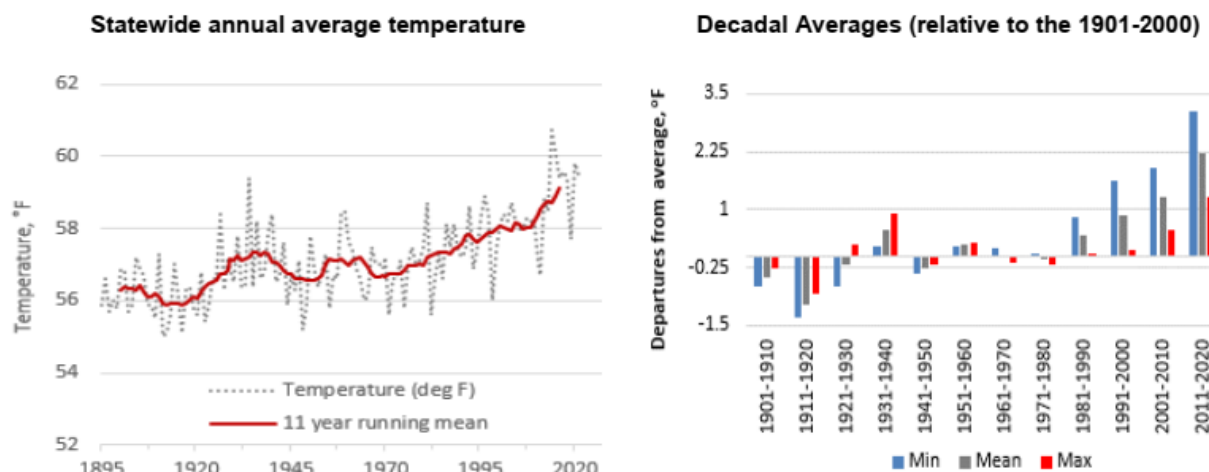


Figure 15. Temperature trend (left) and departure from average (right) graphs for California, from about 1900-2020 (source: OEHHA 2022).

6.2.1 Rising Temperatures

Extreme heat events in California have become more frequent, dating back to the 1950s; however, they have become especially pronounced in the past decade (OEHHA 2022). Heat waves, defined as two or more consecutive heat events (which are characterized by temperatures at or above the highest 5% of historical values), have also become more frequent during this period (OEHHA 2022). For context, over the past 70 years, extreme heat events increased at a rate of about 1 to 3 events per decade at 10 of a set of 14 statewide long-term monitoring sites across California (OEHHA 2022). Further, at several monitoring sites, daytime heat waves increased to as many as 6 events per year, and nighttime heat waves similarly increased to as many as 10 events per year (OEHHA 2022). Long-term regional climate

observations for southern California also follow this pattern of long-term, steady temperature increases. Based on analyses of California South Coast National Oceanic and Atmospheric Administration (NOAA) Climate Division temperature records from 1896–2015, He and Gautam (2016) found significant upward trends in annual average, maximum, and minimum temperatures, with an increase of about 0.29°F (0.16°C) per decade. Likewise, every month of the year has experienced significant positive trends in monthly average, maximum, and minimum temperatures, across the same 100-year period (Hall et al. 2018).

Importantly, nighttime temperatures in California, which are reflected as minimum daily temperatures, have increased by almost three times more than daytime temperatures since 2012 (OEHHA 2022). Gershunov et al. (2009) showed that heat waves over California and Nevada are increasing in frequency and intensity while simultaneously changing in character and becoming more humid. This shift toward humid heat waves in the southwestern U.S. is primarily expressed through disproportionate increases in nighttime air temperatures (Garfin et al. 2013). These changes started in the 1980s and appear to have accelerated since the early 2000s (Garfin et al. 2013). Nighttime warming has been more pronounced in the summer and fall, increasing by about 3.5°F (1.94°C) over the last century, and southern California has warmed faster than Northern California (OEHHA 2022). These long-term regional changes will have disproportionate impacts on aquatic habitats due to elevated atmospheric humidity levels and diminished nighttime cooling effects on southern California waterways (Garfin et al. 2013).

In fact, water temperatures in many streams across California have risen for some time and are continuing to do so (Kaushal et al. 2010). Stream temperatures across the state have increased by an average of approximately 0.9–1.8°F (0.5–1.0°C) in the past 20+ years (e.g., Bartholow 2005 in Moyle et al. 2013). While such increases may seem small, they can push already marginal waters over thresholds for supporting cold water-dependent fishes (Moyle et al. 2015; Sloat and Osterback 2013). Summer water temperatures already frequently exceed 68°F (20°C) in many California streams and are expected to keep increasing under all climate change scenarios (Hayhoe et al. 2004; Cayan et al. 2008 in Moyle et al. 2015). Organisms that are adapted to California’s traditional nighttime cooling influence on their habitats, including Southern SH/RT, are less prone to recover from extreme and extended periods of excessive daytime heat, particularly when humidity and temperatures remain high at night (Garfin et al. 2013; OEHHA 2022).

6.2.2 Drought

Overall, California has been getting warmer and drier since 1895; as part of this long-term climatic shift, droughts are becoming more frequent, extended, and severe in their impacts (OEHHA 2022). As noted, 2000–2021 was the driest 22-year period in the last millennium in the

southwestern United States, including California (Williams et al. 2022). The 2012–2016 drought was one of the warmest and driest on record in California, negatively affecting both aquatic and terrestrial environments across the state (Figure 16; CDFW 2018a). Notable statewide aquatic habitat impacts from this and other prolonged droughts include seasonal shifts in stream hydrographs to earlier peaks with extended summer and fall low flow periods, contraction and desiccation of typically perennial aquatic habitats (Figure 18), poor water quality, elevated water temperatures, changes in migratory cues, spawn timing, and other fish behaviors, stranding, and both direct and indirect mortality of fish, along with estuary and lagoon habitat degradation, among other ecological impacts (CDFW 2018a; Bedsworth et al. 2018).

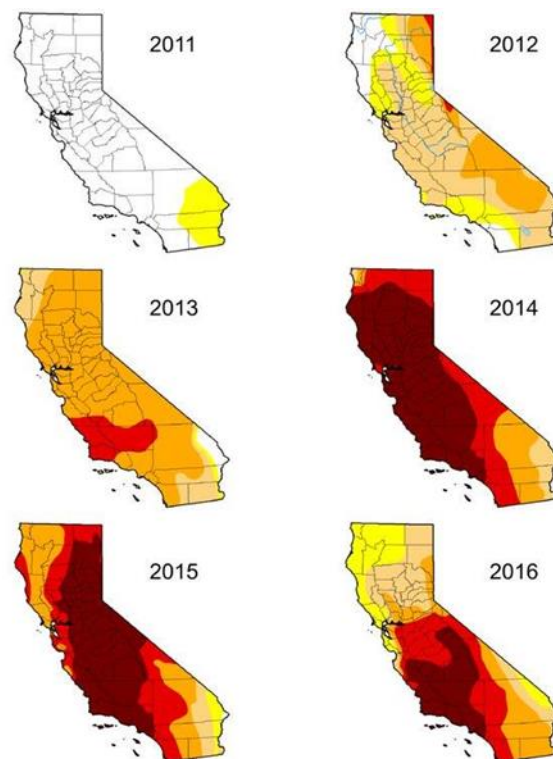


Figure 16. The distribution and progression of drought conditions in California from 2011 to 2016, depicting the level of drought at the beginning of each Water Year (October 1). White indicates no drought conditions, whereas yellow to dark red indicates increasing drought conditions, including duration and intensity (CDFW 2018a, based on U.S. Drought Monitor).

No part of the state has been more impacted by drought than southern California, with significant reductions in precipitation compared to long-term averages, along with record high temperatures, exceptionally dry soils, and low regional snowpack in surrounding mountain ranges in the past decade (Hall et al. 2018). Southern California is naturally arid and already prone to periods of extremely dry conditions (MacDonald 2007; Woodhouse et al. 2010), so increasing drought conditions have amplified many existing ecological stressors while also

creating new ones. As an example, during normal water years, many streams in California's south-coastal region maintain perennial flows in their headwaters but become intermittent or dry in lower portions of their watersheds, especially in areas of concentrated urbanization or agriculture. The 2012–2016 drought dramatically exacerbated these conditions, leading to widespread stream drying in this region, even outside of areas that typically experience annual desiccation (Figure 17; CDFW 2018a). Not surprisingly, CDFW (2018a) noted that the two most common causes of fish kills in southern California during the 2012–2016 drought were stream drying and reduced dissolved oxygen levels (impaired water quality).

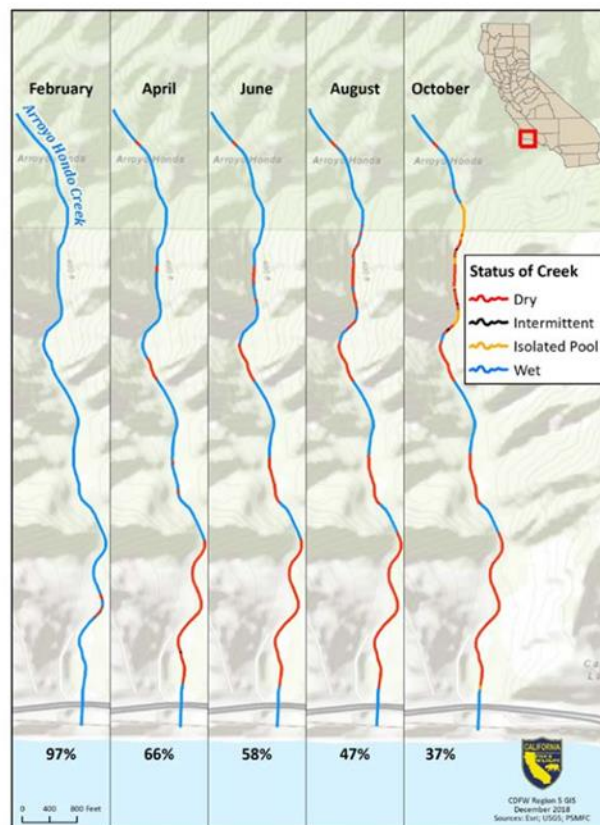


Figure 17. Example southern California stream (Arroyo Hondo Creek, Santa Barbara County), showing seasonal desiccation across 60% of its study area wetted length during February–October 2015 (source: CDFW 2018a). 2015 was a notably bad drought year in California, but the large extent of stream drying in this creek may be an indicator of future climate change-driven conditions in this and other southern California regional streams.

Further desiccation of Southern SH/RT habitats is expected due to climate change, leading to reduced natural spawning, rearing, and migratory habitats for already small and fragmented Southern SH/RT populations. This undesirable future state includes the increasing probability that low-precipitation years continue to align and coincide with warm years, further amplifying

the risk of future severe droughts and low snowpack in California, especially in southern latitudes (Difenbaugh et al. 2015; Berg and Hall 2017; Williams et al. 2015).

In their five-year status reviews, NMFS (2016; 2023) concluded that ongoing “hot drought” conditions, among other negative factors, likely reduced salmonid survival across DPSs and ESUs for listed steelhead and salmon in California, including Southern SH/RT. It is likely that these same Southern SH/RT populations, already impacted and diminished in abundance and distribution, will face more frequent and severe drought periods in the future, along with more intense and destructive (albeit less frequent) winter storms, under all predicted scenarios. Both stressors, in combination, will further negatively affect the remaining suitable habitats for Southern SH/RT in California.

6.2.3 Reduced Snowpack

As air temperatures have warmed, more precipitation has been falling as rain instead of snow at high elevations in the western United States, where widespread snowpack declines of 15–30% have been documented since the 1950s (Mote et al. 2018; Siirila-Woodburn et al. 2021). Since 1950, California’s statewide snow-water content has been highly variable, ranging from more than 200% of the average in 1952, 1969, and 1983 to 5% in 2015 in the midst of the 2012–2016 drought (OEHHA 2022). The past decade included years that were among the lowest (2013, 2014, 2015, and 2022) and the highest (2011, 2017, 2019) on record for snowpack (OEHHA 2022). These patterns demonstrate increasing variability in the amount of overall precipitation the state receives, the frequency and intensity of storm systems, and the amount of precipitation received as rainfall versus snowfall. Annual snowpack in the Peninsular Ranges of southern California (e.g., Santa Ana Mountains, San Jacinto Mountains, and Laguna Mountains) is expected to continue to diminish, so future stream flows in the range of Southern SH/RT will be increasingly driven by rainfall events (Mote et al. 2018).

Snowmelt attenuates stream flows in basins that usually receive annual snowpack at higher elevations. An increase in the ratio of rain to snow and rain-on-snow events will result in more peak flows during winter and early spring, along with an increasing frequency of high flow events and damaging flooding. With earlier seasonal peak hydrographs, many southern California streams will experience diminished spring pulses and protracted periods of low flows through the summer and fall seasons (Moyle et al. 2015). These conditions will translate into warmer water temperatures at most elevations, reflecting both increases in air temperatures and reduced base flows (Moyle et al. 2017). Future shifts from snow to rain may also negatively impact overwintering rearing habitat for juvenile Southern SH/RT and reduce the availability of cold-water holding habitats as refuges in rivers and streams during the summer and fall months (Williams et al. 2016). Such abiotic shifts will affect the physical habitat availability and

suitability for Southern SH/RT and are also anticipated to change species interactions, generally favoring introduced species with broader environmental tolerances (Moyle et al. 2013).

6.2.4 Increasing Hydrologic Variability – Reduced Stream Flows to Catastrophic Flooding

Climate change is likely to increase the impacts of El Niño and La Niña events, which are predicted to become more frequent and intense by the end of the century (OEHHA 2022). Increasingly dramatic swings between extreme dry years (or series of years) and extreme wet years are already occurring in California and are expected to escalate under various climate change scenarios (Swain et al. 2018; Hall et al. 2018). California’s recent rapid shifts from drought periods (2012-2016, 2020-2022) to heavy precipitation and flooding (winter 2016-2017, winter 2022-23) exemplify “precipitation whiplash” and its potential for widespread natural habitat and human infrastructure damage and destruction (OEHHA 2022). California’s river and stream systems will bear the brunt of these impacts since they are the natural conduits for water conveyance on the state’s landscape.

Such precipitation variability and intensity in California is now increasingly influenced by “atmospheric rivers,” or long, narrow bands of precipitation originating over ocean bodies from the tropics to the poles that transport large amounts of water vapor (USGCRP 2017; Hall et al. 2018). During the winter months, heavy precipitation associated with landfalling atmospheric rivers can produce widespread flooding in most of the southwestern U.S. states (Garfin et al. 2013). California is especially vulnerable to this source of destructive flooding because of its proximity to the Pacific Ocean, where atmospheric rivers are generated (USGCRP 2017). As a result of these changes, southern California stream flows will almost certainly become more variable and “flashy” on an annual basis. Predictions include likely extreme fluctuations in precipitation, with intermittent heavy winters producing high stream flows, coastal impacts, and extensive flooding during otherwise prolonged periods of drought, with low to no flows in many streams. Changes in seasonal flow regimes (especially flooding and low flow events) may also affect salmonid behavior. Expected behavioral responses include shifts in the seasonal timing of important life history events such as adult migration, spawning, fry emergence, and juvenile migration (NMFS 2016). The outmigration of juvenile steelhead from headwater tributaries to mainstem rivers and their estuaries may be disrupted by changes in the seasonality or extremity of stream hydrographs (NMFS 2016; Figure 18). Flood events can also disrupt incubation and rearing habitats due to increased bed mobility (Fahey 2006). Conversely, low flow periods with elevated water temperatures and impaired water quality can cause direct mortality to steelhead across wide portions of southern California’s mountain desert streams (CDFW 2018a). Stream drying can also further isolate and restrict subpopulations, potentially leading to genetic drift, interfering with gene flow and genetic mixing at the larger population/ESU level, and potentially further reducing overall fitness.

6.2.5 Sea Level Rise

Along California's coast, mean sea levels have increased over the past century by about 8 inches (203 mm) at monitoring sites in San Francisco and La Jolla (OEHHA 2022). For the southern California coast, roughly 1-2 feet (0.3 m – 0.6 m) of sea level rise is projected by the mid-century, and the most extreme projections indicate 8–10 feet (2.4 m – 3.0 m) of sea level rise by the end of the century (Hall et al. 2018). Sea level rise is predicted to further alter the ecological functions and dynamics of estuaries and near-shore environments. Rising sea levels may impact estuary hydrodynamics with increased saltwater intrusion, potentially increasing salinity levels in estuaries and shifting the saltwater/freshwater interface upstream (Glick et al. 2007). Loss or degradation of already scarce estuary habitats in southern California's coastal areas due to sea level rise may negatively affect Southern SH/RT survival and productivity, since estuaries and lagoons serve as important nursery habitats for juvenile steelhead (Moyle et al. 2017). Alternatively, sea level rise may potentially increase the amount of available estuary habitat by inundating previously dry areas or creating additional brackish, tidal marsh, or lagoon habitats, which serve as important rearing habitats for juvenile salmonids (NMFS 2016). Overall, however, predictions indicate substantial reductions in southern California's coastal lagoon and estuary habitats, which may reduce steelhead smolt survival and numbers of outmigrants to the ocean, further constraining populations of Southern SH/RT (Moyle et al. 2017).

6.2.6 Ocean Acidification

Ocean acidification occurs when excess carbon dioxide (CO₂) is absorbed from the atmosphere, acidifying or lowering the pH of sea water (CDFW 2021b). Ocean acidification is becoming evident along California's central coast, where increases in CO₂ and acidity levels in seawater have been measured since 2010 (OEHHA 2022). Coupled with warming ocean waters and reduced dissolved oxygen levels, ocean acidification poses a serious threat to global marine ecosystems (OEHHA 2022). If left unchecked, ocean acidification could dramatically alter the Pacific Ocean's marine food webs and reduce the forage base for California's salmonids. Forage fish, which are a primary prey source for steelhead in the ocean (LeBrasseur 1966; Quinn 2018), may suffer declines in abundance due to reduced biomass of copepods and other small crustaceans resulting from ocean acidification (Busch et al. 2014). Ocean acidification makes it harder for the shells of ecologically and economically important species, including krill, oysters, mussels, and crabs, to form and potentially causes them to dissolve. Reduced seawater pH has also been shown to adversely affect olfactory discrimination in marine fish (Munday et al. 2009), which could result in impaired homing of Southern SH/RT to their natal streams.

6.2.7 Wildfires

Wildfires are a natural and fundamental part of California's ecological history in many parts of the state. Wildfires are an essential ecological process for the periodic renewal of chaparral vegetation communities (Sugihara et al. 2006), which dominate much of the south-coastal part of California. Historical fires were, therefore, important episodic ecological events with generally lower intensity impacts, at smaller geographic scales, and generally positive long-term outcomes for fish habitats (Boughton et al. 2007).

Euro-American influences and activities on the western landscapes of the U.S., coupled with climate change, have made modern western fires more frequent, severe, and catastrophic in nature (e.g., Gresswell 1999; Noss et al. 2006; and Moyle et al. 2017). Future frequency and size of wildfires in the range of Southern SH/RT is expected to increase, driven by rising atmospheric temperatures and prolonged droughts associated with climate change (NMFS 2012a, OEHHA 2022). Potter (2017) examined satellite data for the 20 largest fires that have burned since 1984 in the central and southern coastal portions of California and found that climate and weather conditions at times of ignition were significant controllers of the size and complexity of high-burn severity fire areas. Since 1950, half of California's largest wildfires (10 of 20) occurred between 2020 and 2021 (OEHHA 2022). One study predicted a nearly 70% increase in the area burned in southern California by the mid-21st century, due to warmer and drier climatic conditions (Jin et al. 2015). This study also evaluated southern California's wildfires in terms of their impacts in the presence or absence of regionally prominent Santa Ana winds. This research found that non-Santa Ana fires which occur mostly in June through August affected higher-elevation forests, while Santa Ana-driven fires which occur mostly from September through December spread three times faster and occurred closer to urban areas (Jin et al. 2015). Recent examples of devastating Santa Ana wind-driven fires include the destructive Thomas Fire (approximately 282,000 acres) in Ventura and Santa Barbara counties (December 2017) and the Woolsey Fire (approximately 97,000 acres) in Los Angeles and Ventura counties (November 2018), both of which were also influenced by preceding record-breaking heatwaves and extremely dry fall conditions (Hulley et al. 2020).

Projected increases in precipitation extremes will lead to increased potential for floods, mudslides, and debris flows (Hall et al. 2018). Wildfires and subsequent debris torrents in southern California were demonstrated to have destroyed Southern SH/RT habitats in 2004, 2006, and 2008 (Moyle et al. 2015). More recent events, including mass wasting and debris flows, such as those in Santa Barbara County in early 2018, resulted from heavy rains preceded by wildfires (Livingston et al. 2018). High-intensity wildfires can accelerate the delivery of sediments to streams (Boughton et al. 2007) by stripping the land of vegetative cover and eliminating stabilizing root structure, thereby degrading spawning habitats for salmonids and

other fishes. Increased soil friability greatly increases rates of fine soil mobilization, erosion, transport, and deposition into watercourses affected by fire due to the elimination of vegetation, the input of large amounts of dry ash and charcoal, the lack of soil shading, and the associated increased solar warming and drying of soils (NMFS 2012a). These fine materials often become so dry after a fire that they become hydrophobic, making it much easier for runoff water to mobilize and transport. Fine sediments delivered to streams in large amounts have been shown to cover and smother coarser-grained spawning gravels, which are required for salmonid spawning success (Moyle et al. 2015). Large-scale sediment mobilization events can also change the channel characteristics of streams, destroy instream and riparian vegetation, and possibly cause direct or indirect mortality to multiple life history stages of Southern SH/RT, while also facilitating the rapid spread of non-native plant and animal species. High flows and floods in fire scars can also scour redds, depending on their seasonal timing, possibly nearly eliminating a Southern SH/RT subpopulation's cohort post-spawn if gravels are mobilized and eggs or juveniles are washed downstream.

6.3 Disease

Numerous diseases caused by bacteria, protozoa, viruses, and parasitic organisms can infect Southern SH/RT in both juvenile and adult life stages. These diseases include bacterial kidney disease (BKD), *Ceratomyxosis*, *Columnaris*, *Furunculosis*, infectious hematopoietic necrosis virus, redmouth and black spot disease, Erythrocytic Inclusion Body Syndrome, and whirling disease (NMFS 2012a). Water quality and chemistry, along with warm stream temperatures, influence infection rates. As water temperatures rise and fish become thermally stressed, lower host resistance aligns with higher pathogen growth rates due to shorter generation times and can lead to a sharp increase in infection rates and associated mortality (Belchik et al. 2004; Stocking and Bartholomew 2004; Crozier et al. 2008). There is little current information available to evaluate the potential impacts of these kinds of infections on Southern SH/RT populations.

6.4 Hatcheries

Extensive stocking of hatchery-origin *O. mykiss* has occurred throughout the southern California region to support recreational fisheries, but no efforts have specifically targeted the conservation and supplementation of Southern SH/RT. Historical stocking records dating back to the 1930s occasionally reference the stocking of "steelhead"; however, it appears that these references represent nomenclature being used interchangeably rather than identification of fish from native migratory populations. Hatchery-origin *O. mykiss* were stocked widely for recreational fisheries up until the late 1990s. Stocking was ceased in the anadromous waters of

southern California as a protective conservation measure starting in 1999 (J. O'Brien, CDFW, personal communication).

While restricted stocking of *O. mykiss* has continued in the region above barriers to anadromy, potential remains for the inadvertent introduction of hatchery stocks into anadromous waters due to downstream movement or during reservoir spill events. To mitigate the risk of hatchery-origin fish interbreeding with wild fish, the Department shifted to stocking only triploid hatchery-origin *O. mykiss* in waters above anadromous barriers following the adoption of the Hatchery and Stocking Program Environmental Impact Report (EIR) in 2010 (Jones and Stokes 2010). Triploid *O. mykiss* have been used across the western United States to reduce the risks of introgression and hybridization associated with stocking programs that support recreational fisheries. The application of heat- or pressure-induced “triploidizing” on salmonid eggs, including *O. mykiss*, has a proven 91-100% sterilization rate, often at the upper end of that range (Kozfkay et al. 2011). Using triploid hatchery-origin *O. mykiss* for recreational fisheries has mitigated some of the inherent risk of potential hybridization and introgression with native and wild stocks, although some risks to Southern SH/RT may still exist. Competition and predation from hatchery stocks remain of concern since the degree to which triploid *O. mykiss* may compete with or prey upon native *O. mykiss* is not well understood.

Hatchery-origin *O. mykiss* have been tagged prior to stocking into select regional reservoirs to attempt to evaluate if and the extent to which they may be escaping these impoundments and entering anadromous waters below dams. No reservoir spills have occurred across the region since tagging began due to the predominance of drought conditions, except for during the winter and spring of 2023. To date, downstream monitoring has not been conducted since the inception of the tagging study (J. O'Brien, CDFW, personal communication). Due to climate change impacts and the decreased frequency with which many southern California reservoirs are filling or overflowing, it is expected that threats from interactions between hatchery-stocked *O. mykiss* and remaining native stocks of Southern SH/RT will be considerably reduced in the future. However, the large number of atmospheric rivers that impacted much of California during the recent winter of 2022–2023, causing some southern California reservoirs to fill and overflow, is a reminder that such events remain possible.

While exclusively triploid hatchery-origin *O. mykiss* are stocked above barriers to anadromy in southern California, historical regional stocking practices of non-triploid fish have led to introgression, or hybridization with hatchery stocks, in some Southern SH/RT populations. Levels of introgression appear to vary across the landscape, differing between populations and watersheds. Some populations retain high levels of native southern California steelhead ancestry, while others are highly introgressed and exhibit high levels of hatchery-origin genetics (primarily Central Valley *O. mykiss* genetics), while some are in between, with genetic

signatures from both native and hatchery origins (Clemento et al. 2008; NMFS 2016; Jacobson et al. 2014). See Section 6.7 in this Status Review for more information.

6.5 Predation

6.5.1 Predation in Freshwater Environments

California's salmonids have evolved under selective pressure from a variety of natural predators, including many species of fish, birds, and mammals; however, a growing number of non-native aquatic species have also become established within the range of Southern SH/RT (Busby et al. 1996; NMFS 2016; Stillwater Sciences 2019; Dagit et al. 2019; COMB 2022). Established populations of non-native fishes, amphibians, and invertebrates, combined with anthropogenic habitat alterations that often favor non-native species, have led to increased impacts from predation, competition, and other stressors on Southern SH/RT across much of its range (NMFS 1996b). Stream habitat alteration can also directly affect predation rates by reducing available cover for prey species, creating flow and velocity regimes that favor non-native predators, and creating obstructions to passage that can lead to migration delays and increased exposure to predators (Moyle et al. 2013; Dagit et al. 2017). Further, stream habitat alterations can influence water temperatures, often increasing them, which may then lead to higher metabolic rates for piscivorous fishes and increased predation pressure (Michel et al. 2020). In addition to physical habitat alterations, chemical habitat alterations in the form of contaminants known to alter fish behavior and reduce avoidance or cover-seeking activities are also likely to increase predation rates, particularly from avian predators (Grossman 2016).

Established populations of non-native catfish and centrarchids occur in the lower reaches of many watersheds throughout the range of Southern SH/RT, leading to widespread predation risk (NMFS 2016; Stillwater Sciences 2019; Dagit et al. 2019; COMB 2022). Grossman (2016) found that non-native Channel Catfish (*Ictalurus punctatus*) may be a primary predator of Central Valley steelhead in the San Joaquin River, suggesting they may pose the same level of risk to Southern SH/RT. Non-native centrarchids have been demonstrated to negatively impact salmonid populations through direct predation on rearing juveniles and resident adult *O. mykiss* (Dill and Cordone 1997; Marks et al. 2010; NMFS 2012a; Bonar et al. 2005).

Abundant populations of non-native fish have been documented in many southern California coastal watersheds, including Malibu Creek, lower Arroyo Trabuco, Santa Margarita, and San Luis Rey rivers. These species include largemouth and redeye bass, green sunfish, mosquito fish, and black bullhead (C. Swift, Emeritus, Section of Fishes, Natural History Museum of Los Angeles County, personal communication; O'Brien et al. 2022).

In addition to piscivorous fishes, non-native invertebrates and amphibians have also been introduced and spread across the Southern SH/RT range. American bullfrogs (*Lithobates catesbeianus*) have become widely established and can prey upon rearing juvenile steelhead (COMB 2022; Cucherousset and Olden 2011; Dagit et al. 2019; Stillwater Sciences 2019). Non-native Red Swamp Crayfish (*Procambarus clarkia*) populations have also increased in some Southern SH/RT waters (Garcia et al. 2015; Dagit et al. 2019). Direct observations of YOY Southern SH/RT being attacked by crayfish in shallow riffle-run habitat suggest that predation poses a threat to the survival of juvenile steelhead (Dagit et al. 2019).

6.5.2 Predation in Marine Environments

Marine predation influences on Southern SH/RT are not well documented or understood. Primary predators of salmonids in the marine environment are pinnipeds, such as harbor seals (*Phoca vitulina*) and California sea lions (*Zalophus californianus*) (Cooper and Johnson 1992; Spence et al. 1996). Although fish are a major dietary component of marine pinnipeds, their predation on Southern SH/RT may be minimal at present, given the very low relative abundances of Southern SH/RT.

6.6 Competition

Competition is the interaction between individuals of the same or different species that compete for a limited supply of a common resource (Holomuzki et al. 2010). The extent to which competition impacts the distribution, abundance, and productivity of Southern SH/RT populations is not well understood. Pacific steelhead typically compete with other salmonid species like Coho and Chinook salmon in freshwater; however, unlike northern populations of steelhead that typically co-occur with other salmonid species, Southern SH/RT are the only salmonids that occur in their range. While inter-specific competition with other salmonids is unlikely to occur, intraspecific competition among Southern SH/RT may be prevalent in southern California watersheds, especially those that are highly degraded. Poor and degrading habitat conditions can contribute to increased competition, which, in turn, can adversely affect fish during the juvenile life-history stage and lead to reduced recruitment and reproductive performance over the entire life cycle (Chilcote et al. 2011; Tatara et al. 2012). Limited habitat space, coupled with high juvenile densities, is associated with reduced growth, premature emigration, increased competition for food, decreased feeding territory sizes, and increased mortality (Kostow 2009).

Juvenile steelhead are habitat generalists, occupying a variety of microhabitat types in streams depending on the size and age of individuals (Spina et al. 2005). Non-native fish species can competitively restrict the spatial distribution of juvenile steelhead to suboptimal habitats such as shallower, higher-velocity riffles, where the energetic cost to forage is higher (Rosenfeld and

Boss 2001). Non-native fish species may also exclude juvenile steelhead from areas of suitable habitat. For example, recent watershed-wide surveys in Sespe Creek, a large and unregulated tributary to the Santa Clara River, documented the absence of Southern SH/RT in several stream reaches with suitable steelhead habitat (i.e., cool water with deep pools) that were dominated by multiple species of non-native juvenile fishes (Stillwater Sciences 2019). According to Krug et al. (2012), Arroyo Chub may also compete with Southern SH/RT juveniles for food resources. Like juvenile steelhead, Arroyo Chub are opportunistic feeders and consume benthic and drift invertebrates, sometimes switching preferences depending on food abundance. Southern SH/RT and Arroyo Chub are frequently part of the same native southern California fish assemblages and generally habitat partition, with juvenile steelhead mostly feeding on drift invertebrates while chub have a more benthic diet. However, periods of diet overlap may lead to strong interspecific competition between the two species. While other native fishes may impose some level of competitive threat to Southern SH/RT, it remains likely that non-native competitors pose the greater threat, especially with these species continued expansion and proliferation (O'Brien and Barabe 2022).

6.7 Genetic Diversity

West coast steelhead have considerable genetic diversity, both within and across populations, including variation in traits linked to anadromy, morphology, fecundity, spawning, and run timing, as well as age at smolting and maturation (McElhany et al. 2000). While some traits are entirely genetically based, the expression of most traits usually varies, due to a combination of both genetic and environmental factors. Species with high genetic diversity typically occupy a wider range of habitats than those with lower diversity and are more resilient to both short-and long-term spatial-temporal fluctuations in the environment such as ecological disturbances (i.e., wildfires, floods, and landslides) and human-caused impacts. Generally, populations need to be large enough to maintain long-term genetic diversity and avoid genetic problems, such as loss of variation, inbreeding depression, bottlenecks, and the accumulation of deleterious mutations, all of which occur more frequently in smaller populations.

A range-wide genetic analysis demonstrated that populations in the southernmost portions of the Southern SH/RT range are dominated by hatchery ancestry, indicating genetic introgression of native lineages with hatchery strains (Jacobson et al. 2014; Abadia-Cardoso et al. 2016). Most of these hybridized wild populations occur above barriers in the upper reaches of the Los Angeles, San Gabriel, Santa Ana, San Juan, San Diego, and Sweetwater rivers. It is unclear whether introgression will decrease the viability of these southern populations, since the introduction of small amounts of novel genetic material, even from hatchery stocks, can lead to increased diversity and the phenomenon known as “hybrid vigor,” conferring adaptive resilience to changing environments and the negative impacts of inbreeding. This study also

confirmed that the northernmost populations of Southern SH/RT, including all watersheds in the Monte Arido Highlands BPG, contain native steelhead ancestry and generally higher genetic diversity than more southern populations (Clemento et al. 2009; Abadia-Cardoso et al. 2016).

As with other salmonids, natural straying and the resultant gene flow between populations maintain the genetic diversity of Southern SH/RT. A recent study, which examined the otoliths of seven adult steelhead from a small basin on the Big Sur coast of California, revealed that all adults were strays, coming from at least six different source populations, including neighboring ones on the Big Sur coast as well as distant populations such as the Klamath River (Donohoe et al. 2021). As is the case for many coastal steelhead populations, the genetic diversity of Southern SH/RT has been compromised by human impacts on their habitats, such as the blocking of migration corridors by artificial dams and widespread reductions in streamflow, at least partially due to locally and regionally intensive water diversions for municipal, agricultural, and other human consumptive uses (NMFS 2012a).

Measures of genetic diversity, such as heterozygosity and allelic richness, indicate that Southern SH/RT populations have lower diversity than northern coastal populations. Within the range of Southern SH/RT, the northernmost populations in the Santa Maria, Santa Ynez, Ventura, and Santa Clara rivers have higher genetic diversity than the southernmost populations (Abadia-Cardoso et al. 2016). Previous genetic studies have revealed that populations occurring downstream of modern artificial barriers are genetically more similar to above-barrier populations in the same basin than they are to populations below barriers in neighboring basins (Clemento et al. 2009). While above- and below-barrier populations within the same drainage are usually each other's closest relatives, they appear divergent in respect to the frequencies of the anadromous (A) and resident (R) haplotypes found in each subpopulation (see Section 4.7). The A haplotype is more common below dams, while the R haplotype is found more frequently above dams. This evidence of selection against the anadromous genotype is likely a product of artificial dams or other barriers blocking anadromous adults from returning to these upstream areas to reproduce and provide A haplotype genetic influx to the above-barrier population (Pearse et al. 2014; Pearse et al. 2019). Apgar et al. (2017) found that the frequency of the A haplotype is strongly associated with several factors, including the extent of migration barriers present, barrier type (complete, partial, artificial, or natural), barrier age (recent or longstanding), and migration distance. Genetic diversity in above-barrier populations is an important repository of genetic material, serving a similar function as conservation hatcheries do in other parts of the Southern SH/RT range (D. Boughton, NOAA, personal communication; NMFS 2012a)

Because migratory phenotypes are primarily genetically based, variation in the reproductive success of anadromous and resident individuals can influence the tendency of populations to

produce anadromous offspring, corresponding to changes in the frequency of the A haplotype. Moreover, environmental factors, such as intra-and inter-annual climate variation, food availability, and water temperature, also influence the expression of anadromy in Southern SH/RT populations (Satterthwaite et al. 2009; Ohms et al. 2014; Kendall et al. 2015). Furthermore, climate change projections for Southern SH/RT range predict an intensification of climate patterns, such as more intense cyclic storms, droughts, and extreme heat (NMFS 2012a). These projections suggest that Southern SH/RT will likely experience more frequent periods of adverse conditions and continued selection pressure against the anadromous life-history form.

6.8 Habitat Conditions

The decline of Southern SH/RT can be attributed to a wide variety of human activities, including, but not limited to, urbanization, agriculture, and water development. These activities have degraded range-wide aquatic habitat conditions, particularly in the lower and middle reaches of most watersheds in the Southern SH/RT range (NMFS 2012a). Southern California is home to over 20 million people and 1.8 million acres of metropolitan, urban, and suburban areas (DWR 2021) which has resulted in highly urbanized watersheds that are impacted by surface and groundwater diversions and associated agricultural, residential, and industrial uses. Major rim dams, instream diversion dams, and other water conveyance infrastructure have significantly reduced or eliminated access to the majority of historical upstream rearing and spawning habitat for southern steelhead. While some of these human activities have been reduced, eliminated, or mitigated, the cumulative impacts of these activities remain throughout most of the Southern SH/RT range, particularly in larger systems such as the Santa Maria, Santa Ynez, Ventura, Santa Clara, Los Angeles, San Gabriel, Santa Ana, and Santa Margarita watersheds, as well as in smaller coastal systems such as Malibu Creek.

6.8.1 Roads

High human population densities in southern California have led to the development of an extensive network of transportation corridors throughout the range of Southern SH/RT. The extensive road and highway networks across much of the Southern SH/RT range, especially in areas proximate to rivers and streams, are attributed to increases in a number of negative habitat impacts. Among these are: non-point pollution (e.g., oil, grease, and copper from braking systems); sedimentation; channel incision due to bankside erosion; substrate embeddedness; floodplain encroachment and loss of floodplain connectivity; loss of channel heterogeneity (e.g., filling of pool habitats); and higher frequencies of flood flows (NMFS 2012a). Additionally, extensive road and highway networks require many road crossings (e.g.,

culverts and bridges) that are often improperly designed for the volitional passage of aquatic organisms (CalTrans 2007; NMFS 2012a).

NMFS (2012) assessed the impacts of roads and transportation corridors on Southern SH/RT using roads per square mile of watershed and the density of roads within 300 feet of streams per square mile of watershed as metrics. The results of their analysis demonstrated that roads and associated passage barriers have the highest impact on rivers and streams in the Santa Monica Mountains and Conception Coast BPG regions: 60% of watersheds in the Conception Coast BPG ranked “very high” or “high” in severity for roads as a stressor, while 100% of the watersheds that drain the Santa Monica Mountains received the same ranking. Highway 101 and the Union Pacific Railroad cross the mainstem of each watershed along the Conception Coast BPG region (as well as the Monte Arido Highlands BPG region) near their river mouths. At each major transportation crossing, culverts were constructed to allow stream flows to pass through to the Pacific Ocean, but they were not necessarily engineered to allow upstream fish passage. For example, the Highway 101 culvert on Rincon Creek serves as a total barrier to upstream migration, preventing Southern SH/RT from reaching any of its historical habitats upstream of the barrier. Road development, bridges, and other transportation corridors are also partly responsible for the significant (70-90%) reduction of estuarine habitat across all BPGs (Hunt and Associates 2008).

The Mojave Rim and Santa Catalina Gulf Coast BPG regions are home to the highest urban densities across the Southern SH/RT range, and both BPGs are impacted by high road densities. For example, in the Santa Catalina Gulf Coast BPG region, the Rancho Viejo Bridge, Interstate-5 Bridge array, and the Metrolink drop structure are all recognized as total fish passage barriers on Arroyo Trabuco Creek, a tributary to San Juan Creek. On the Santa Margarita River, an outdated box culvert at the Sandia Creek Bridge serves as a significant fish passage barrier on the river (Dudek 2001). Recently, efforts have been undertaken to repair and modify these barriers to provide upstream steelhead passage and again allow access to many miles of historical habitat in these watersheds (see Chapter 6: Influence of Existing Management Efforts).

6.8.2 Dams, Diversions, and Artificial Barriers

A number of anthropogenic impacts, including water diversions, dams, and other artificial barriers, influence stream flows in most Southern SH/RT-supporting watersheds. Municipal and agricultural beneficial uses comprise the majority of water demand in the South Coast region (Mount and Hanak 2019). Surface water diversions can lead to reduced downstream flows, as well as changes to the natural flow regime (e.g., magnitude, timing, and duration of flow events), stream hydrodynamics (e.g., velocity, water depth), and degradation of both habitat

quality and quantity needed to support Southern SH/RT (NMFS 2012a; Yarnell et al. 2015). Changes to the natural flow regime can result in elevated downstream water temperatures, reduced water quality, shifts in fish community composition and structure, increased travel times for migrating fish, increased susceptibility of native aquatic organisms to predation, and reduced gravel recruitment from upstream areas of watersheds to the lower reaches of rivers (NMFS 1996b; Axness and Clarkin 2013; Kondolf 1997). Dams physically separate fish populations into upstream and downstream components, leading to population and habitat fragmentation, along with potential changes to population spatial and genetic structure over time (NMFS 2012a). Large dams often trap upstream sediments, which naturally would be transported downstream and deposited, augmenting substrates and improving spawning habitats for salmonids and other fish. It is common for rivers and streams with large dams to exhibit more scouring and streambed degradation downstream of the impoundment (Kondolf 1997; Yarnell et al. 2015). Stream flow reductions also interfere with the downstream transport and influx of freshwater to estuaries. The consequences of reduced inflows to estuaries include wetland and edge habitat loss, changes to the amount and location(s) of suitable habitat for aquatic organisms and accelerated coastal erosion (Nixon et al. 2004).

Many types of artificial stream barriers exist throughout the range of Southern SH/RT, including dams, concrete channels for flood control, gravel and borrow pits, roads and utility crossings, fish passage facilities, and other non-structural features such as velocity barriers. In the South Coast hydrologic region, a total of 164 known total migration barriers were identified as part of a larger effort to inventory fish passage barriers across California's coastal watersheds (California Coastal Conservancy 2004). Of the 164 total barriers, 11 were identified as requiring modification or removal to improve fish passage. Dams were identified as the most numerous barrier type, followed by stream crossings and non-structural barriers. The Santa Maria River, San Antonio Creek, Cuyama River, Santa Ynez River, and Santa Barbara coastal watersheds, which all belong to the Central Coast hydrologic region, also contain hundreds of known barriers scattered throughout the area, with the highest number found along the Santa Barbara coastal area (California Coastal Conservancy 2004).

Artificial barriers act as physical impediments but may also contribute to, or enhance, non-structural barriers to steelhead spawning migrations. For example, the three major watersheds of the Los Angeles basin have channelized concrete aqueducts in their lower reaches, with some extending from their mouths upstream for miles. As a result, adult Southern SH/RT can no longer access the lower reaches of these three major regional rivers (Titus et al. 2010). Furthermore, if Southern SH/RT were to successfully enter into the channelized reaches of these rivers, migration success would be limited because individuals would encounter non-structural velocity barriers that would require greater swimming speeds than could be sustained (Castro-Santos 2004). Other non-structural barriers may exist in the form of low

flows, disconnected wetted habitat, and poor or lethal water quality in these largely metropolitan lower river aqueduct reaches.

Most of the large rivers in the Monte Arido Highlands BPG region contain multiple large, impassable dams. Twitchell Dam on the Cuyama River is primarily managed for groundwater recharge in the Santa Maria Valley. Operations of Twitchell Dam limit downstream surface flows into the mainstem Santa Maria River (NMFS 2012a). Cachuma, Gibraltar, and Juncal dams on the mainstem Santa Ynez River prevent upstream migratory access to approximately 70% of historical spawning and rearing habitat in the watershed (NMFS 2012a). In the Ventura River watershed, Matilija and Casitas dams on Matilija Creek and Coyote Creek, respectively, block access to 90% of historical Southern SH/RT spawning and rearing habitat. However, the recent Matilija Dam Ecosystem Restoration Project is aimed at restoring over 20 miles of perennial Southern SH/RT habitat in the Matilija Creek watershed through the removal of Matilija Dam. Santa Felicia Dam and Pyramid Dam on Piru Creek, as well as Castaic Dam on Castaic Creek, block access to historical habitat in the tributaries of the mainstream Santa Clara River. Several of these large dams are operated along with smaller downstream diversion dams: primarily the Robles Diversion Dam on the Ventura River and the Vern Freeman Diversion Dam on the Santa Clara River. The Robles Diversion Dam diverts water from the upper Ventura River into storage at Lake Casitas, while the Vern Freeman Diversion diverts water for groundwater recharge purposes in the Santa Clara Valley.

Two major dams impair habitat connectivity and hydrologic function in the Malibu Creek watershed: Rindge Dam and Malibu Lake Dam. Both dams have created favorable habitat conditions for non-native species, including crayfish, snails, fish, and bullfrogs. As a result, invasive aquatic species have been documented in high abundance in Malibu Creek (NMFS 2012a). Rindge Dam is located only 2 miles upstream of the mouth and is no longer functional, so it is targeted for future removal. The removal of this dam alone would allow Southern SH/RT access to 18 miles of high-quality spawning and rearing habitat in the Malibu Creek watershed.

Dams are ranked “high” or “very high” as a threat in 88% of the component watersheds that comprise the Mojave Rim BPG region (NMFS 2012a). There are also at least 20 jurisdictional-sized dams (i.e., a dam under the regulatory powers of the State of California) within each of the three major watersheds of the Los Angeles basin, owned by federal, state, local, and/or private entities and operated for multiple purposes, including: irrigation, flood control, storm water management, and recreation. The principal impoundments in the San Gabriel River watershed are Whittier Narrows, Santa Fe, Morris, San Gabriel, and Cogswell dams. Sepulveda Dam on the Los Angeles River is operated as a flood control structure approximately 8 miles downstream from the river’s source. Big Tujunga Dam on Big Tujunga Creek, a tributary to the Los Angeles River, is also operated as a flood control structure. Prado Dam on the Santa Ana

River is also primarily operated as a flood risk management project. These dams alter the physical, hydrological, and habitat characteristics of the lower and middle reaches of the mainstem rivers in this BPG. They also create favorable habitat for non-native species such as crayfish, largemouth bass, and bullfrogs, which have all been documented in the Los Angeles, San Gabriel, and Santa Ana rivers. Periodic removal of sediments accumulated behind dams on the San Gabriel River also degrades downstream riparian and instream habitat conditions (Hunt and Associates 2008).

In the Santa Catalina Gulf Coast BPG, dams also ranked “high” or “very high” as a threat in 90% of constituent watersheds. At least 20 major dams and diversions without fish passage facilities occur throughout the BPG’s distribution. Prominent dams in this BPG include Agua Tibia, Henshaw, and Eagles Nest dams in the San Luis Rey watershed; and the O’Neill Diversion and Vail dams in the Santa Margarita River watershed. Dams in this BPG are generally not operated with fish passage as a consideration in flow release schedules, and many of these facilities lack fish passage provisions (NMFS 2012a).

Groundwater extraction for agricultural, industrial, municipal, and private use from coastal aquifers has increased with population growth in southern California since the mid-1850s (Hanson et al. 2009). Currently, around 1.57 million acre-feet of groundwater are used on an annual basis in southern California to meet both urban and agricultural water demands (DWR 2021). Groundwater is an important input for surface flows during the summer low flow period in many southern California watersheds (Hanson et al. 2009). Groundwater contributions can help sustain suitable over-summering Southern SH/RT juvenile rearing habitat in both mainstem and tributary habitats (Tobias 2006). Unsustainable groundwater water diversions have led to the depletion of several large aquifers in the region (NMFS 2012a). Offsite pumping can impact the surface-water to groundwater interactions by intercepting water that would have otherwise discharged to a stream or by lowering the water table, causing a reduction of baseflow derived from groundwater during the summer low flow period. While some riparian species can tolerate reduced groundwater contributions to streams, for many other species, such as Southern SH/RT, adequate surface water depth, velocity, and water quality characteristics must be maintained in order to survive (Tobias 2006). The combination of surface water diversions and groundwater extractions can lead to the complete drying of streams, which can lead to the stranding of Southern SH/RT in isolated pools and direct mortality. On average, 57% of watersheds across the five BPGs ranked “high” or “very high” for groundwater extraction as a threat (NMFS 2012a).

Recently, the Sustainable Groundwater Management Act priority process identified several groundwater basins across the South Coast hydrologic region as either critically over drafted (i.e., Santa Clara River Valley, Cuyama River Valley, and Pleasant Valley) or medium-to-high

priority basins for water conservation (e.g., the Coastal Plain of Orange County) based on several metrics such as population growth rates, the total number of wells, and the number of irrigated acres (DWR 2020). Groundwater sustainability agencies overseeing critically overdrafted and medium-to-high priority basins are responsible for developing and realizing groundwater sustainability plans (GSPs) to achieve basin sustainability within a 20-year implementation horizon. However, the benefits provided by SGMA for Southern SH/RT and their habitats are uncertain, as the most commonly cited goal for GSPs thus far has been to increase groundwater storage and not the restoration of interconnected surface water flows (Ulibarri et al. 2021).

6.8.3 Estuarine Habitat

The estuaries of many coastal watersheds in southern California form freshwater lagoons that are seasonally closed to the ocean. Lagoons form when low summer baseflows are unable to displace sand deposition at the mouth of the estuary, which results in the formation of a sandbar that blocks connectivity with the ocean. This closure creates an environment characterized by warmer and slower-moving (i.e., longer residence times) freshwater that is relatively deep (Bond et al. 2008). These habitat characteristics provide important, high-quality nursery conditions for rearing juveniles and transition areas for smolts acclimating to the ocean environment. Adult steelhead also acclimate in these areas prior to upstream migration during the winter months when the estuary is fully open (NMFS 2012a). The importance of such habitats was demonstrated by the observed doubling of growth in juvenile *O. mykiss*, which reared throughout the summer in a typical northern California coastal watershed (Bond et al. 2008). The same study examined scales from returning adult steelhead and found that estuary-reared individuals dominated adult returns, despite comprising only a small part of the annual outmigrating population. Another study conducted in the same watershed also reported higher growth rates for estuary-reared juvenile steelhead than for their cohorts reared in the upper watershed (Hayes et al. 2011). Hayes et al. (2011) also found that the lagoon environment provided warmer water temperatures and a diverse abundance of invertebrate prey resources for rearing juvenile *O. mykiss* to consume. Trade-offs between accelerated growth and survival likely exist in lagoon habitats because they represent a relatively high-risk yet high-reward environment in which accelerated growth may come at the cost of increased metabolic demand and potentially increased predation risk, exposure to poor water quality, and episodic artificial breaching (Osterback et al. 2013; Satterthwaite et al. 2012; Swift et al. 2018).

The southern California Bight, which encompasses the entire southern California coastline, from Point Conception to San Diego, historically supported around 20,000 hectares of estuary habitat (Stein et al. 2014). Over half of all historical estuaries were found in San Diego County (e.g., Mission Bay and San Diego Bay), while Los Angeles and Orange counties contained about 15%

each of the total estimated historical area. Estimates of the amount of estuarine habitat loss from historical levels, based on wetland acreage, range from 48-75% (Brophy et al. 2019; NMFS 2012a; Stein et al. 2014). The magnitude of the loss varies depending on the watershed. For example, the estuaries of the Santa Maria and Santa Ynez rivers in the northern portion of the Southern SH/RT range remain almost entirely intact, while the estuaries of the Los Angeles, San Gabriel, and Santa Ana rivers have been reduced to 0-2% of their historical extent (NMFS 2012a). Overall, estuary habitat loss in southern California is likely underestimated because early landscape modifications (e.g., housing and transportation development and associated filling of wetlands with sediment) had substantially altered the landscape before attempts were made to quantify the extent of historical habitat (Brophy et al. 2019).

The primary cause of estuarine loss in southern California is the conversion of habitat to other land use practices such as agriculture, grazing, and urban development activities, which require the construction of infrastructure and the subsequent filling, diking, and draining of coastal wetlands (NMFS 2012a). Currently, estuary habitats in the range of Southern SH/RT remain highly degraded and prone to further degradation by urban impacts such as point and nonpoint source pollution, coastal development, and dams. These environmental stressors can cause declines in water quality and the proliferation of harmful algal blooms that can lead to the rapid die-off of both aquatic and terrestrial organisms (Lewitus et al. 2012; Smith et al. 2020). Artificial breaching of estuaries also poses a mortality risk to Southern SH/RT. Seven moribund juvenile steelhead were observed in the lagoon at the mouth of the Santa Clara River shortly after the sandbar was artificially breached in 2010 (Swift et al. 2018). The authors of this study noted that the Santa Clara River, upstream of the lagoon, was dry during this time and that the observed fish were relatively large and in robust condition, indicating that favorable rearing conditions existed prior to the artificial breaching.

6.8.4 Water Quality and Temperature

Contaminants and pollutants are well-documented to alter water quality parameters that affect the growth and survival of Pacific salmonids in both freshwater and estuarine environments (Arkoosh et al. 1998; Baldwin et al. 2009; Laetz et al. 2008; Sommer et al. 2007; Sullivan et al. 2000). Both are generally introduced into southern California rivers and streams by urban runoff, agricultural and industrial discharges, wastewater treatment effluent, and other anthropogenic activities. Recent monitoring conducted by the USGS measured between 20 and 22 current-use pesticides in samples collected from urban sites at Salt Creek and the Sweetwater River in Orange and San Diego counties (Sanders et al. 2018). Diminished water quality conditions, including contaminants and associated toxicity, elevated nutrients, low dissolved oxygen, increased temperature, and increased turbidity, can all adversely affect Southern SH/RT as well as other native fish and aquatic organisms. The effects of individual

pollutants and combinations thereof can impact populations by altering growth, reproduction, and mortality rates of individual fish (Sommer et al. 2007). These impacts can ultimately manifest in direct mortality due to acute and long-term physiological stress or may act through indirect pathways such as changes to food webs, ecosystem dynamics, increased susceptibility to disease and predation, and more frequent occurrences of harmful algal blooms. Aquatic stressors that impair water quality can also interact with each other in an additive or synergistic fashion, such that they are generally interdependent and can greatly amplify negative impacts on aquatic ecosystems (Sommer et al. 2007). Dissolved oxygen concentrations, turbidity, and water temperatures are all parameters directly influenced by flow management. Lower flows can lead to warmer water temperatures that hold less dissolved oxygen than cold water. Higher water temperatures also increase the metabolic and oxygen consumption rates of aquatic organisms, making these conditions particularly stressful for aquatic life (Myrick and Cech 2000). See Section 6.2.1 in this Status Review for a full description of air and water temperature influences and trends.

Many watersheds that support Southern SH/RT are listed under Section 303(d) of the federal Clean Water Act (CWA). Section 303(d) requires states to maintain a list of waters that do not meet prescribed water quality standards. For waters on this list, states are required to develop TMDLs that account for all sources (i.e., point and non-point sources) of the pollutants that caused the water to be listed as impaired under the CWA. In southern California, there are many impaired water bodies and pollutant combinations listed under Section 303(d). While contaminant and discharge sources have changed over the years and there have been significant improvements in controlling many of these sources, many 303(d)-listed waters do not yet have approved TMDLs (SWRCB 2020). All four of the major rivers in the Monte Arido Highlands BPG region are listed as 303(d)-impaired, and each system contains over five sources of pollutants. Seven Southern SH/RT-supporting watersheds in the Conception Coast BPG region and three in the Santa Monica Mountains BPG region are 303 (d) listed, including Jalama, Gaviota, Mission, Carpinteria, Rincon, Big Sycamore Canyon, Malibu, and Topanga creeks. All three of the major watersheds in the Mojave Rim BPG region, as well as eight out of ten in the Santa Catalina Gulf Coast BPG region, are 303(d)-listed, including the Los Angeles, San Gabriel, Santa Ana, Santa Margarita, San Diego, and Sweetwater rivers and the San Juan, San Mateo, San Luis Rey, and San Dieguito creeks. Essentially, all rivers and streams supporting Southern SH/RT that are 303(d)-listed are impaired by multiple pollutants, including water temperature, benthic community effects, indicator bacteria, trash, toxicity, and invasive species. Furthermore, southern California's coastal and bay shorelines, estuary environments, and tidal wetlands are also frequently 303(d)-listed as impaired. As examples, the estuaries of Malibu, Aliso, San Juan, and Los Penasquitos creeks; the entirety of Santa Monica Bay; and the estuaries

of the Los Angeles, Santa Clara, Santa Margarita, and Tijuana rivers are all listed as 303(d)-impaired waterbodies.

6.8.5 Agricultural Impacts

The impacts of agricultural development have lessened over time as farm and pasturelands continue to be converted to urban development in southern California (NMFS 2012a). Historically, the loss of riparian and floodplain habitat was due first to conversion by livestock ranching, followed by irrigated row-crop agriculture, and then urban development. For example, interior portions of the Santa Clara River floodplain were originally converted to agriculture but are now dominated by urban growth and major human population centers, such as the cities of Santa Paula and Fillmore. Today, the South Coast hydrologic region supports approximately 159,000 acres of agricultural land, with avocados, citrus, truck crops, and strawberries comprising the highest agricultural production by acreage (DWR 2021). Approximately 530,000 acre-feet of groundwater are annually pumped from underlying basins to support agricultural production in southern California (DWR 2021). Agricultural activities produce wastewater effluent containing nutrients that can either directly or indirectly be introduced into the rivers, streams, and estuaries that support Southern SH/RT, particularly when agricultural best management practices and water quality objectives have not been established. Agricultural production is prevalent in several watersheds, including the lower Santa Maria and Santa Ynez rivers; many of the smaller coastal watersheds along the Santa Barbara coast, such as the Goleta Slough complex and Rincon Creek; the upper Ventura River and the Ojai basin; and portions of the San Mateo Creek, San Luis Rey, and San Dieguito River tributaries in the southernmost portion of the range. Statewide, the counties of Ventura, Santa Barbara, and San Diego are each ranked in the top fifteen for total value of agricultural production (CDFA 2021).

While the impacts of agricultural development on Southern SH/RT and their habitats have decreased over time due to land use conversion, both activities have resulted in considerable cumulative regional habitat loss and degradation. These changes have led to greatly reduced habitat complexity and connectivity in the lower and middle reaches of many southern California watersheds. Currently, agricultural impacts on Southern SH/RT are most evident during the summer dry season, when agricultural and residential water demands are the highest. This period coincides with the juvenile *O. mykiss* rearing life-history stage, which is dependent on adequate summer base flows to maintain suitable habitat conditions for growth and survival (Grantham et al. 2012). Agricultural groundwater diversions can lead to rapid stream drying by depleting aquifer groundwater that contributes to stream base flows, which limits the extent of summer rearing habitat for fish (Moyle et al. 2017). Naturally occurring surface waters supported only by groundwater recharge can be rapidly dewatered due to

excessive groundwater pumping or diversions. These areas have been shown to provide adequate depth, surface area, and habitat for steelhead in streams lacking cold-water refuges (Tobias 2006).

The cultivation, manufacturing, and distribution of cannabis products have increased since recreational use became legal in California in 2016 (Butsic et al. 2018). Threats and stressors on aquatic ecosystems associated with the cultivation of cannabis include stream flow and bank modifications, water pollution, habitat degradation, and species invasions (CDFW 2018b). Cannabis is a water-and nutrient-intensive crop that requires an average of up to 6 gallons of water per day, per plant, during the growing season, which usually spans a total of 150 days from June to October (Zheng et al. 2021). Water diversions can lead to changes in flow regimes, the creation of fish passage barriers, the loss of suitable spawning and foraging habitat, and the rerouting and dewatering of streams, especially during drought years or during the dry season (CDFW 2018b; see Section 6.8.2).

6.8.6 Invasive Species

Invasive and non-native species are abundant and widely distributed in many watersheds that support Southern SH/RT. Non-native species frequently occur in both anadromous and non-anadromous waters that have been extensively stocked by a variety of public and private entities (NMFS 2012a). Most reservoirs contain non-native species, such as largemouth and smallmouth bass, carp, sunfish, bullfrogs, and bullhead catfish, that can all establish reproducing populations in the river and stream reaches above and below the dams. Range-wide habitat alteration has also facilitated the widespread distribution and increased abundance of non-native fish species, which typically favor slower-moving, warmer-water habitats with lower dissolved oxygen concentrations and higher sediment loads (Moyle et al. 2017). While the introduction of non-native game species has historically been viewed as a fishery enhancement, these species can have negative impacts on Southern SH/RT due to predation, competition, disease, habitat displacement and alteration, as well as behavior modifications (Cucherousset and Olden 2011).

Non-native species have recently been documented in high densities in Sespe Creek, an unregulated tributary to the Santa Clara River and a Department-designated Wild Trout Water (Stillwater Sciences 2019). High abundances of invasive species are due to the historic and ongoing stocking of non-native fish in the Rose Valley Lakes on Howard Creek, a tributary to Sespe Creek. In both Malibu and Topanga creeks, red swamp crayfish abundances have increased with recent warmer stream temperatures and lower flow conditions despite regular removal efforts (Dagit et al. 2019). High densities of crayfish likely have a direct (predation) and indirect (competition) effect on Southern SH/RT in both creeks. A variety of warm-water, non-

native fish species are frequently observed in the lower Santa Ynez River, including multiple species of sunfish and catfish, carp, and largemouth bass, all of which are known predators of Southern SH/RT early life stages. In the lower Ventura River, annual monitoring efforts have consistently detected higher numbers of non-native fish species than Southern SH/RT in recent years (CMWD 2021).

Non-native plant and amphibian species also occur in several watersheds that support Southern SH/RT. Invasive plants such as giant reed and tamarisk have displaced extensive areas of native riparian vegetation in major drainages, such as the Santa Clara and San Luis Rey rivers (NMFS 2012a). These water-intensive plant species both reduce instream flows through groundwater uptake and severely reduce the extent of riparian cover and shading. These habitat changes often affect stream flow and thermal regimes, potentially increasing susceptibility of Southern SH/RT to predation, disease, and competitive exclusion. Other non-native plant species, such as water primrose and hyacinth, both of which form dense, sprawling mats on the water's surface, can alter the structure and function of aquatic ecosystems by outcompeting native aquatic plants, reducing the amount of open water habitat, altering the composition of invertebrate communities, physically blocking fish movement, and inducing anoxic conditions detrimental to fish (Khanna et al. 2018). In the Santa Clara River watershed, bullfrogs and African clawed frogs are abundant and widespread throughout the mainstem reaches, from the estuary upstream to Fillmore, including tributaries such as Santa Paula Creek and Hopper Canyon Creek (NMFS 2012a). Both species represent a threat to native aquatic communities because they opportunistically consume a variety of native prey, and eradication of either species is unlikely (Wishtoyo Foundation 2008).

6.9 Fishing and Illegal Harvest

Southern SH/RT traditionally supported important recreational fisheries for both winter adults and summer juveniles in coastal streams and lagoons (NMFS 2012a, Swift et al. 1993). Angling-related mortality may have contributed to the decline of some small populations but is generally not considered a leading cause of the decline of the Southern California Steelhead DPS as a whole (Good et al. 2005; Busby et al. 1996; NMFS 1996b). After the southern California steelhead DPS was federally listed as endangered in 1997, Department fishing regulation modifications led to the closure of recreational fisheries for Southern SH/RT in marine and anadromous waters with few exceptions. That closure continues, and there is currently no legal recreational fishery for Southern SH/RT (CDFW 2023).

Southern SH/RT take is primarily from poaching rather than legal commercial and recreational fishing. While illegal harvest rates appear to be very low, the removal of even a few individuals in some years could be a threat to the population because of such low adult abundance in most

populations (Moyle et al. 2017). Southern SH/RT are especially vulnerable to poaching due to their high visibility in shallow streams. Estimates of fishing effort from self-report cards for 1993–2014 suggest extremely low levels of angling effort for Southern SH/RT, primarily due to the statewide prohibition of angling in anadromous waters starting in 1998 (NMFS 2016; Jackson 2007). Historic commercial driftnet fisheries may have contributed slightly to localized declines; however, Southern SH/RT are targeted in commercial fisheries, and reports of incidental catch are rare. Commercial fisheries are not thought to be a leading cause of the widespread declines of Southern SH/RT over the past several decades (NMFS 2012a).

7. INFLUENCE OF EXISTING MANAGEMENT EFFORTS

7.1 Federal and State Laws and Regulations

Several state and federal environmental laws apply to activities undertaken in California that provide some level of protection for Southern SH/RT and their habitat. There are also restoration, recovery, and management plans, along with management measures specific to habitat restoration, recreational fishing, research, and monitoring that may benefit Southern SH/RT. The following list of existing management measures is not exhaustive.

7.1.1 National Environmental Policy Act and California Environmental Quality Act

The National Environmental Policy Act (NEPA) was enacted in 1970 to evaluate the environmental impacts of proposed federal actions. The NEPA process begins when a federal agency proposes a major federal action. The process involves three levels of analysis: 1) Categorical Exclusion determination (CATEX); 2) Environmental Assessment (EA) or Finding of No Significant Impact (FONSI); and 3) Environmental Impact Statement (EIS). A CATEX applies when the proposed federal action is categorically excluded from an environmental analysis because it is not deemed to have a significant impact on the environment. If a CATEX does not apply, the lead federal agency for the proposed action will prepare an EA, which concludes whether the action will result in significant environmental impacts. A lead agency will issue a FONSI document if significant impacts are not expected. Alternatively, if the action is determined to have a potentially significant effect on the environment, an EIS containing an explanation of the purpose and need for the proposed action, a reasonable range of alternatives that can achieve the same purpose and need, a description of the affected environment, and a discussion of environmental consequences of the proposed action is required (EPA 2017). The United States Environmental Protection Agency is responsible for reviewing all EIS documents from other federal agencies and must provide NEPA documentation for its own proposed actions. Because the Southern California DPS is listed as endangered under the federal ESA, proposed actions that may impact this population are

evaluated as biological resources in the project area concurrently and interdependently with the federal ESA Section 7 consultation process.

The California Environmental Quality Act (CEQA) is similar to NEPA in that it requires environmental review of discretionary projects proposed by state and local public agencies unless an exemption applies (Pub. Resources Code, § 21080). Under CEQA, the lead agency is responsible for determining whether an EIR, Negative Declaration, or Mitigated Negative Declaration is required for a project (Cal. Code Regs., tit. 14, § 15051). When there is substantial evidence that a project may have a significant effect on the environment and adverse impacts cannot be mitigated to a point where no significant effects would occur, an EIR must be prepared that identifies and analyzes environmental impacts and alternatives (Pub. Resources Code, § 21082.2, subds. (a) & (d)). Significant effects for a proposed project may occur if project activities have the potential to substantially reduce the habitat, decrease the number, or restrict the range of any rare, threatened, or endangered species (Cal. Code Regs., tit. 14, §§ 15065, subd. (a)(1) & 15380). CEQA requires public agencies to avoid or minimize significant effects where feasible (Cal. Code Regs., tit. 14, § 15021); NEPA does not include this requirement. Further, CEQA requires that when a lead agency approves a project which will result in significant effects which are identified in the final EIR but are not avoided or substantially lessened, the agency shall make a statement of overriding considerations in which the agency states in writing the specific reasons to support its action based on the final EIR and/or other information in the record (Cal. Code Regs., tit. 14, § 15093).

7.1.2 Federal Endangered Species Act

The ESA was established in 1973 to conserve and protect fish, wildlife, and plants that are listed as threatened or endangered. The ESA provides a mechanism to add or remove federally listed species, cooperate with states for financial assistance, and develop and implement species recovery. The ESA also provides a framework for interagency coordination to avoid take of listed species and for issuing permits for otherwise prohibited activities. The lead federal agencies for implementing the ESA are the USFWS and NMFS. Federal agencies are required to consult with either the USFWS or NMFS to ensure that actions they undertake, fund, or authorize are not likely to jeopardize the continued existence of any listed species or their designated critical habitat. The federal ESA prohibits the take, import, export, or trade in interstate or foreign commerce of ESA-listed species.

NMFS listed the Southern California Steelhead DPS as endangered under the federal ESA in 1997 as part of the South-Central/Southern California Coast recovery domain and designated critical habitat for that DPS in 2005 (NMFS 2012a). The scope of the DPS is naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the

Santa Maria River to the U.S.-Mexico border. NMFS's West Coast Region manages recovery planning and implementation for this domain, and in 2012 the region adopted a Recovery Plan for the Southern California Steelhead DPS, which provides the foundation for recovering populations to healthy levels. The listing of the DPS afforded the DPS ESA protections through the consultation provisions of ESA Section 7(a)(2); habitat protection and enhancement provisions of ESA Section 4 and 5; take prohibitions through ESA Sections 4(d) and 9; cooperation with the State of California through ESA Section 6; and research, enhancement, and species conservation by non-federal actions through ESA Section 10.

Section 7(a)(2) of the ESA requires federal agencies to ensure their actions are not likely to jeopardize the continued existence of the species or adversely modify designated critical habitat. The agency requesting consultation will typically produce and submit a biological assessment that documents potential effects on listed species or their habitats to either the USFWS or NMFS. USFWS or NMFS then produces and submits a Biological Opinion to the requesting agency that contains conservation recommendations and actions to minimize any harmful effects of the proposed action. Currently, NMFS spends a significant amount of its resources and time fulfilling Section 7 consultation requirements for federal actions that may impact the Southern California Steelhead DPS (NMFS 2012a). This includes working with agencies to avoid and minimize the potential impacts of proposed actions and to ensure project activities do not jeopardize the species or destroy critical habitat. NMFS has issued Biological Opinions for several large federally owned and operated projects, including the Santa Felicia Hydroelectric Project on Piru Creek (2008), USBR's operation and maintenance of the Cachuma Project on the Santa Ynez River (2000), USBR's construction and operation of the Robles Diversion Fish Passage Facility on the Ventura River (2003), the U.S Army Corp of Engineer's (USACE) Matilija Dam Removal and Ecosystem Restoration Project on Matilija Creek (2007), USACE's Santa Paula Creek Flood Control Project (2013). However, the application of Section 7(a)(2) is limited in scope because it applies only to federal actions and areas under federal ownership, and without a related federal action it does not apply to the significant areas of public and private ownership in southern California (NMFS 2012a).

7.1.3 Clean Water Act and Porter-Cologne Water Quality Act

The CWA was established in 1972 to regulate the discharge of pollutants into the waters of the United States and create surface water quality standards. Section 401 of the CWA requires any party applying for a federal permit or license for a project that may result in the discharge of pollutants into the waters of the United States to obtain a state water quality certification. This certification affirms that the project adheres to all applicable water quality standards and other requirements of state law. Section 404 of the CWA prohibits the discharge of dredged or fill material into the waters of the United States without a permit from the USACE. Activities

regulated under this program include fill for development, water resource projects, infrastructure development, and mining projects. Applicants for a 404 permit must demonstrate that all steps have been taken to avoid impacts to wetlands, streams, and aquatic resources and that compensation is provided for unavoidable impacts prior to permit issuance from the USACE.

Since 1969, the Porter-Cologne Water Quality Act (Porter-Cologne Act) has been the principal law governing water quality in California. The Porter-Cologne Act includes goals and objectives that align with those of the federal CWA, such as water quality standards and discharge regulations. The SWRCB and nine regional water quality control boards share responsibility for the implementation and enforcement of the Porter-Cologne Act. These entities are required to formulate and adopt water quality control plans that describe beneficial uses, water quality objectives, and a program of implementation that includes actions necessary to achieve objectives, a time schedule for the actions to be taken, and monitoring to determine compliance with water quality objectives and the protection of beneficial uses of water.

Under Section 401 of the CWA, a federal agency may not issue a permit or license to conduct any activity that may result in any discharge into waters of the United States unless a Section 401 water quality certification is issued or certification is waived. The SWRCB and the regional water quality control boards administer Section 401 water quality certifications in California.

In accordance with Section 303(d) of the CWA, the U.S. Environmental Protection Agency (EPA) assists the SWRCB and the regional water boards in listing impaired waters and developing TMDLs for waterbodies within the state. TMDLs establish the maximum concentration of pollutants allowed in a waterbody and serve as the starting point for restoring water quality. The primary purpose of the TMDL program is to assure that beneficial uses of water, such as cold freshwater and estuarine habitat, are protected from detrimental increases in sediment, water temperature, and other pollutants defined in Section 502 of the CWA. TMDLs are developed by either the regional water quality control boards or the EPA. TMDLs developed by the regional water quality control boards are included as water quality control plan amendments and include implementation provisions, while those developed by the EPA contain the total load and load allocations required by Section 303(d) but do not contain comprehensive implementation provisions. The EPA is required to review and approve the list of impaired waters and each TMDL. If the EPA cannot approve the list or a TMDL, it is required to develop its own. There can be multiple TMDLs on a particular waterbody, or there can be one TMDL that addresses numerous pollutants. TMDLs must consider and include allocations to both point and non-point sources of the listed pollutants.

Approved TMDLs and their implementation plans are incorporated into water quality control plans required by the Porter-Cologne Act of 1969. For a specified area, a water quality control plan designates the beneficial uses and water quality objectives established for the reasonable protection of those beneficial uses. Such beneficial uses may include warm freshwater habitat; cold freshwater habitat; rare, threatened, or endangered species; and migration of aquatic organisms. The beneficial uses, together with the water quality objectives that are contained in a water quality control plan and state and federal antidegradation requirements, constitute California's water quality standards for purposes of the CWA.

Waters within the range of the Southern SH/RT are under the jurisdiction of the Central, Los Angeles, Santa Ana, and San Diego regional water quality control boards. There are many 303(d)-listed impaired waterbodies within the jurisdiction of each of these regional boards, and most waterbodies have more than one pollutant that exceeds water quality standards designed to protect beneficial uses of water, water quality criteria, or objectives. More information on 303(d) listed waters in southern California can be found at: https://www.waterboards.ca.gov/water_issues/programs/water_quality_assessment/2018_integrated_report.html

The National Pollution Discharge Elimination System (NPDES) delegated implementation responsibility for the regulation of wastewater discharges to the State of California through the SWRCB and the regional water quality control boards. In southern California, tertiary wastewater treatment plants commonly discharge treated water into the rivers, streams, and estuaries that support Southern SH/RT. For example, the Tapia Water Reclamation Facility discharges tertiary treated effluent into Malibu, Las Virgenes, and Arroyo Calabasas creeks. While wastewater effluent is often the primary source of streamflow for southern California rivers and streams during the summer months, the potential impacts of wastewater effluent on adult and juvenile life stages are not well understood (NMFS 2012a). The review, assessment, and potential modification of NPDES wastewater discharge permits is a key recovery action in the federal recovery plan for the Southern California DPS to address the threat of urban effluents (NMFS 2016).

7.1.4 Federal and California Wild and Scenic Rivers Act

In 1968, Congress enacted the National Wild and Scenic Rivers Act (WSRA) to preserve certain rivers with outstanding natural, cultural, and recreational values in a free-flowing state. Under the National Wild and Scenic Rivers System, rivers are classified as either wild, scenic, or recreational. Designation neither prohibits development nor gives the government control over private property; recreation, agricultural practices, residential development, and other land uses may continue. However, the WSRA does prevent the federal government from licensing,

funding, or otherwise assisting in dam construction or other projects on designated rivers or river segments. Designation does not impact existing water rights or the existing jurisdiction of states and the federal government over waters. In California, approximately 2,000 miles of river are designated as wild and scenic, which comprises about one percent of the state's total river miles. The California Wild and Scenic Rivers Act was passed by the California Legislature in 1972. The state act mandates that "certain rivers which possess extraordinary scenic, recreational, fishery, or wildlife values shall be preserved in their free-flowing state, together with their immediate environments, for the benefit and enjoyment of the people of the state." (Pub. Res. Code, § 5093.50). Designated waterways are codified in Public Resources Code Sections 5093.50-5093.70.

The designated state and federal wild and scenic rivers within the range of Southern SH/RT are the Sisquoc River, Piru Creek, and Sespe Creek. The Sisquoc River, which is a tributary of the Santa Maria River, contains 33 miles of designated water from its origin in the Sierra Madre Mountains downstream to the Los Padres National Forest boundary. Piru and Sespe creeks are both tributaries of the Santa Clara River and encompass a combined 38 miles of designated waters. The downstream end of Pyramid Dam and the boundary between Los Angeles and Ventura counties constitute the start and end points of the designated reach for Piru Creek. The designated reach for Sespe Creek is the main stem from its confluence with Rock Creek and Howard Creek downstream, near its confluence with Tar Creek. Both Sespe Creek and the Sisquoc River have comprehensive river management plans that address resource protection, development of lands and facilities, user capacities, and other management practices necessary or desirable to achieve the purposes of the WSRA (USDA 2003a; USDA 2003b).

7.1.5 Lake and Stream Bed Alteration Agreements

Fish and Game Code Section 1602 requires entities to notify the Department prior to beginning any activity that may "divert or obstruct the natural flow of, or substantially change or use any material from the bed, channel, or bank of any river, stream, or lake, or deposit or dispose of debris, waste, or other material containing crumbled, flaked, or ground pavement where it may pass into any river, stream, or lake." The requirement applies to both intermittent and perennial waterbodies. If an activity will adversely affect an existing fish and wildlife resource, the Department's Lake and Streambed Alteration Program is responsible for issuing a Lake or Streambed Alteration (LSA) Agreement that includes reasonable measures necessary to protect the resource (Fish & G. Code, §1602, subd. (a)(4)(B)). There are several types of LSA agreements that entities can request from the Department, including standard; general cannabis; gravel, sand, or rock extraction; routine maintenance; timber harvest; and master.

Recently, severe storms during the winter of 2023 in southern California caused flooding, landslides, and mudslides within the watersheds that Southern SH/RT occupy. As a result, multiple emergency actions were conducted to protect life and property. In these circumstances, Fish and Game Code Section 1610 exempts entities that conduct certain emergency work from notification requirements prior to the start of any work activity and instead requires them to notify in writing within fourteen days after the work begins.

In the South Coast Region, legal cannabis cultivation is currently focused in Santa Barbara County, with a concentration of the larger notifications in the Santa Ynez River watershed. The Santa Ynez River and its tributaries are a high priority wildlife resource that supports *O. mykiss*, the Southern California Steelhead DPS listed as endangered under the federal ESA; southwestern willow flycatcher, which is listed as endangered under both the federal ESA and CESA; least Bell's vireo, which is listed as endangered under both the federal ESA and CESA; and California red-legged frog, which is listed as threatened under the federal ESA. There are currently about 453 acres of permitted cannabis in the Santa Ynez watershed. Project water use adjacent to the Santa Ynez River can have significant individual and/or cumulative impacts on Southern SH/RT and other species along this reach and adjacent up- and downstream areas. The predominant water source for these large grows along the Santa Ynez River and within the region are well diversions that can be located within or immediately adjacent to the stream. These diversions have the potential to substantially affect surface flows, hydrology, and vegetation within the Santa Ynez River. Where this situation occurs along the Santa Ynez River, Department staff have included appropriate measures to report on water use in any agreements that have been issued. Such measures include having an established protocol for monitoring and reporting water use throughout the season. Permittees must also abide by the SWRCB forbearance period for diversion of surface water during the dry season, from April 1 through October 1 of each calendar year.

7.1.6 Medicinal and Adult-Use Cannabis Regulation and Safety Act

Regulation of the commercial cannabis cultivation industry under the Medicinal and Adult-Use Cannabis Regulation and Safety Act requires that any entity applying for an annual cannabis cultivation license from the California Department of Food and Agriculture include "a copy of any final lake or streambed alteration agreement... or written verification from the California Department of Fish and Wildlife that a lake or streambed alteration agreement is not required" with their license application (Cal. Code Regs., tit. 3, § 8102, subd. (w)). Waste discharge and water diversions associated with cannabis cultivation are regulated by the SWRCB (Cal. Code Reg., tit. 3, § 8102, subd. (p)).

7.1.7 Federal Power Act

The Federal Energy Regulatory Commission (FERC) implements and enforces the Federal Power Act. FERC has the exclusive authority to license most non-federal hydropower projects that are located on navigable waterways, federal lands, or are connected to the interstate electric grid. The term for a hydropower license granted by FERC is typically 30-50 years. FERC must comply with federal environmental laws prior to issuing a new license or relicensing an existing hydropower project, including NEPA and ESA. Section 10(a) of the Federal Power Act instructs FERC to solicit recommendations from resource agencies and tribes (when applicable) on ways to make a project more consistent with federal or state comprehensive plans. Section 10(j) allows NMFS, USFWS, and the Department to submit recommendations to protect, mitigate damage to, and enhance fish and wildlife resources affected by a proposed project. FERC is not required to incorporate these recommendations into a hydropower license if it determines the recommendations are outside the scope of Section 10(j) or inconsistent with the Federal Power Act or any other applicable law.

Pursuant to Section 401 of the CWA, FERC may not issue a FERC license to a project unless a Section 401 water quality certification is issued to that project or that certification is waived. The SWRCB administers 401 water quality certifications for projects that involve a FERC license.

UWCD owns and operates Santa Felicia Dam, which is the main component of the Santa Felicia Project (*FERC Project Number 2153*). The project is located on Piru Creek, a tributary of the Santa Clara River, in Ventura County. Santa Felicia Dam, which is located five miles north of the town of Piru, impounds Piru Creek to form Lake Piru Reservoir. Lake Piru has a usable storage capacity of 67,997 acre-feet, and the spillway of the Santa Felicia Dam has a capacity of 145,000 cfs. A small powerhouse located on the west embankment of the dam is capable of producing up to 1,420 kilowatts of energy. UWCD owns two appropriative water rights for the project for the purposes of power, domestic, industrial, municipal, irrigation, and recreational uses. The project currently operates under a 2014 water quality certification that contains provisions to protect fish and wildlife beneficial uses in lower Piru Creek, including a reservoir release schedule to protect Southern SH/RT migration flows each year from January 1 through May 31 (see https://www.waterboards.ca.gov/waterrights/water_issues/programs/water_quality_cert/santafelicia_ferc2153.html for more information).

7.1.8 Sustainable Groundwater Management Act

In September 2014, the Governor signed legislation to strengthen the management and monitoring of groundwater basins. These laws, known collectively as the Sustainable Groundwater Management Act (SGMA), established a timeline and process for forming local

GSAs in designated groundwater basins. GSAs are responsible for developing and implementing GSPs to achieve basin sustainability within a 20-year implementation horizon. DWR is the agency responsible for reviewing and approving individual GSPs, while the SWRCB serves as the regulatory backstop for groundwater basins found to be out of compliance with SGMA. Since 2014, the Department's Groundwater Program has developed multiple documents to assist GSAs in developing and implementing effective GSPs, including a groundwater consideration planning document and a habitat-specific document for wetlands (CDFW 2019). These documents highlight scientific, management, legal, regulatory, and policy considerations that should be accounted for during GSP development. DWR is currently in the process of reviewing GSP plans for critically overdrafted and medium-to-high priority basins. Within the range of Southern SH/RT, there are over fifteen GSPs that are currently being reviewed by DWR. SGMA requires GSAs to submit annual reports to DWR each April 1 following the adoption of a GSP. Annual reports provide information on groundwater conditions and the implementation of the GSP for the prior water year (see <https://water.ca.gov/Programs/Groundwater-Management/SGMA-Groundwater-Management/Groundwater-Sustainability-Plans> for more information).

7.1.9 State Water Resources Control Board Water Rights Administration

Water rights are a legal entitlement authorizing water to be diverted from a specified source and put to a beneficial, non-wasteful use. Riparian water rights are based on ownership of land bordering a waterway, while appropriative water rights are issued without regard to the relationship of land to water but rather the priority in which the water was first put to beneficial use. The exercise of most water rights (i.e., appropriative water rights) requires a permit or license from the SWRCB. The goal of the SWRCB in making water rights-related decisions is to develop water resources in an orderly manner, prevent waste and unreasonable use of water, and protect the environment. The SWRCB has several other major water rights - related duties, including but not limited to: participating in water rights adjudications; enhancing instream uses for fish and wildlife beneficial uses; approving temporary water transfers; investigating possible illegal, wasteful, or unreasonable uses of water; and revoking or terminating water rights. SWRCB-issued water right permits contain public trust provisions for the protection of instream aquatic resources. While these provisions (i.e., maximum diversion amounts and diversion seasons) are meant to protect aquatic resources, they do not have an explicit regulatory mechanism to implement protections required in other state statutes. Furthermore, prior to recent advancements in groundwater management, the SWRCB generally lacked the authority to regulate groundwater diversions and development. Overlying landowners may extract percolating groundwater without approval from the SWRCB as long as the extracted water is put to beneficial uses and the region in which the groundwater diversion occurs has not been formally adjudicated.

7.1.10 Fish and Game Code Section 5937

Fish and Game Code Section 5937 states “the owner of any dam shall allow sufficient water at all times to pass through a fishway, or in the absence of a fishway, allow sufficient water to pass over, around, or through the dam, to keep in good condition any fish that may be planted or exist below the dam.”

7.2 Recovery Plans and Regional Management Plans

7.2.1 Southern California Steelhead Recovery Plan

The Southern California Steelhead Recovery Plan (Recovery Plan) was adopted in 2012 following the listing of the Southern California Steelhead DPS in 1997. The goal of the Recovery Plan is to prevent the extinction of Southern California Steelhead in the wild; ensure the long-term persistence of viable, self-sustaining populations of steelhead distributed across the DPS; and establish a sustainable sport fishery (NMFS 2012a). Generally, recovery of the DPS, which consists of naturally spawned anadromous steelhead originating below natural and manmade impassable barriers from the Santa Maria River to the U.S.-Mexico Border, entails the protection, restoration, and maintenance of a range of habitats in the DPS to allow all life-history forms to be fully expressed (e.g., anadromous and resident). The Recovery Plan outlines key objectives that address factors limiting the DPS’s ability to survive and naturally reproduce, including preventing extinction by protecting populations and habitats, maintaining the current distribution of steelhead and restoring distribution to historically occupied areas, increasing abundance, conserving existing genetic diversity, and maintaining and restoring habitat conditions to support all of its life-history stages. NMFS defines a viable population as a population that has a less than 5% risk of extinction due to threats from demographic variation, non-catastrophic environmental variation, and genetic diversity changes over a 100-year time frame (NMFS 2012a).

The Recovery Plan organizes the recovery plan area into five BPGs: Monte Arido Highlands, Conception Coast, Santa Monica Mountains, Mojave Rim, and Santa Catalina Gulf Coast. The BPGs were initially divided based on whether individual watersheds within them are ocean-facing systems subject to marine-based climate inversion and orographic precipitation from ocean weather patterns. Secondly, population groups were then organized based on similarity in physical geography and hydrology. The rationale for this approach is that steelhead populations utilizing unique individual watersheds have different life histories and genetic adaptations that enable the species to persist in a diversity of different habitat types represented by the BPGs. The Recovery Plan’s strategy emphasizes larger watersheds in each BPG that are more capable of sustaining larger and more viable populations than smaller watersheds. Core 1 populations are identified as having the highest priority based on their

intrinsic potential for meeting viable salmonid population criteria, the severity of the threats facing the populations, and the capacity of the watershed and population to respond to recovery actions (NMFS 2012a).

Like all federal recovery plans, the Recovery Plan for the Southern California Steelhead DPS contains recovery criteria, recovery actions, and estimates of the time and costs to achieve recovery goals. Recovery criteria are objective, measurable criteria that, when met, would result in a determination that the DPS be delisted. Recovery criteria for the Southern California Steelhead DPS Recovery are based on both DPS-level and population-level criteria. At the population level, criteria include characteristics such as mean annual run-size, spawner density, and anadromous fraction, while the DPS-level criteria are informed by the minimum number of populations that must be restored in each BPG. Recovery actions are site-specific management actions necessary to achieve recovery. Actions for the Southern California DPS are organized based on the BPG and core population approaches. High-priority recovery actions include, but are not limited to, physically modifying passage barriers such as dams to allow natural rates of migration to upstream spawning and rearing habitats, enhancing protection of natural in-channel and riparian habitats, reducing water pollutants, and conducting research to better understand the relationship between resident and anadromous forms (NMFS 2012a).

7.2.2. Forest Plans

Land Management, or Forest Plans, were developed by the United States Department of Agriculture for the southern California National Forests (the Angeles, Cleveland, Los Padres, and San Bernardino National Forests) in 2006 to provide a framework for guiding ongoing land and resource management operations. The southern California Forest Plans contain various protections for Southern SH/RT that occur within national forests. These include, but are not limited to, mitigating the effects of visitor use within watersheds occupied by steelhead, working collaboratively with federal and state agencies and water management entities to restore steelhead trout access to upstream habitat, reducing risks from wildland fires to maintain water quality, and eliminating and limiting the further spread of invasive nonnative species (USDA 2005). For example, in 2014, the Cleveland National Forest initiated an effort to restore Southern SH/RT migratory corridors in the San Juan and Santiago watersheds by removing numerous small, outdated, and non-functional concrete barriers constructed by Orange County to force groundwater to the surface (C. Swift, Emeritus, Section of Fishes Natural History Museum of Los Angeles County, personal communication; Donnell et al. 2017). Thus far, up to 81 passage barriers on Silverado, Holy Jim, Trabuco, and San Juan creeks have been removed. Forest Plans are required to be updated every 10 to 15 years. In recent years,

several amendments to the Southern California National Forest Plans have been adopted in response to monitoring and evaluation, new information, and changes in conditions.

7.2.3 Habitat Conservation Plans and Natural Community Conservation Plans

A Habitat Conservation Plan (HCPs) is a planning document that authorizes the incidental take of a federally listed species when it occurs due to an otherwise lawful activity. HCPs are designed to accommodate both economic development and the permanent protection and management of habitat for species covered under the plan. At minimum, HCPs must include an assessment of the impacts likely to result from the proposed taking of one or more federally listed species, the measures that the permit applicant will undertake to monitor, minimize, and mitigate such impacts, the funding available to implement such measures, procedures to deal with unforeseen or extraordinary circumstances, alternative actions to the taking that the applicant analyzed, and the reasons why the applicant did not adopt such alternatives (USFWS 2021).

The Natural Community Conservation Planning Act authorized the Department to develop Natural Community Conservation Plans (NCCPs). NCCPs identify and provide for the regional protection of plants, animals, and their habitats, while allowing compatible and appropriate economic activity. The development of a NCCP by a local agency requires significant collaboration and coordination with landowners, environmental organizations, and state and federal agencies. Most approved HCP/NCCP documents are joint documents that fulfill the requirements of both Section 10 of the ESA and the Natural Community Conservation Planning Act.

Within the range of the Southern SH/RT, there are at least nine HCP or NCCPs that are either in the implementation phase or the planning phase. The majority of HCP and NCCP plans are for the southern portion of the Southern SH/RT range and include multiple plan subareas. For example, the San Diego County Multiple Species Conservation Program contains six subareas, including the City of San Diego, Poway, Santee, La Mesa, Chula Vista, and South San Diego County. Generally, rivers, streams, and riparian vegetation communities in HCP and NCCP plan areas are considered ecologically important areas that are targeted for conservation. HCP/NCCP plans typically contain provisions to conserve fish and wildlife habitat, including fire management, invasive species control, fencing, trash removal, and annual monitoring.

7.2.4 Other Management and Restoration Plans

The Steelhead Restoration and Management Plan for California is a Department-statewide steelhead management plan that provides guidelines for steelhead restoration and

management that can be incorporated into stream-specific project planning (McEwan and Jackson 1996).

<https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=3490>

7.3 Habitat Restoration and Watershed Management

7.3.1 Fisheries Restoration Grant Program

The goal of the Department's Fisheries Restoration Grant Program (FRGP) is to recover and conserve salmon and steelhead trout populations through restoration activities that reestablish natural ecosystem functions. The FRGP annually funds projects and activities that provide a demonstrable and measurable benefit to anadromous salmonids and their habitat; restoration projects that address factors limiting productivity as specified in approved, interim, or proposed recovery plans; effectiveness monitoring of habitat restoration projects at the watershed or regional scales for anadromous salmonids; and other projects such as outreach, coordination, research, monitoring, and assessment projects that support the goal of the program. Uniquely, the FRGP provides CWA Section 401 certification and CWA Section 404 coverage for all eligible projects funded through the program. In recent years, several FRGP proposals have been funded to support conservation efforts for Southern SH/RT, including the Upper Gaviota Fish Passage Project (2022), Life Cycle Monitoring on Topanga Creek and the Ventura River (2021), Fish Passage Barrier Removal on San Jose Creek, Gaviota Creek, and Maria Ygnacio Creek (2021), and the South Coast Steelhead Coalition (2021) (see <https://wildlife.ca.gov/Grants/FRGP> for more information.)

7.3.2 Proposition 68 and Proposition 1

The Water Quality, Supply, and Infrastructure Improvement Act of 2014 (Proposition 1) and the California Drought, Water, Parks, Climate, Coastal Protection, and Outdoor Access for All Act of 2018 (Proposition 68) authorized both the Wildlife Conservation Board and the Department to award significant grant funding to restoration projects that are intended to benefit Southern SH/RT. Both entities distribute Proposition 68 and Proposition 1 funds on a competitive basis to projects that specifically address river and stream restoration (Proposition 68; Proposition 1), Southern SH/RT habitat restoration (Proposition 68), fish and wildlife habitat restoration (Proposition 68; Proposition 1), or stream flow enhancements (Proposition 1). Proposition 68 funded projects that benefit Southern SH/RT and their habitat include the Harvey Diversion Fish Passage Restoration Project on Santa Paula Creek, the Matilija Dam Ecosystem Restoration Project on Matilija Creek, and the Santa Margarita River Fish Passage Project and Bridge Replacement. Proposition 1 funded projects include, but are not limited to, *Arundo donax* removal at the Sespe Cienega on the Santa Clara River, the Santa Clara River Riparian

Improvement, and the Integrated Water Strategies Project for Flow Enhancement in the Ventura River Watershed (WCB 2021).

7.3.3 Other Habitat Restoration Funding Sources

In addition to funding provided by the Department and Wildlife Conservation Board, Southern SH/RT conservation projects are also supported by numerous other funding sources. These sources include local, state, and federal sources such as the California Coastal Conservancy, Pacific Coastal Salmon Recovery Fund, the National Fish and Wildlife Foundation, the NOAA Restoration Center, the California Department of Water Resources Integrated Regional Water Management Plan grant program (Proposition 50), the California Natural Resources Agencies Parkways Program (Proposition 40), the CalTrans Environmental Enhancement and Mitigation Program, the Santa Barbara County Coastal Resource Enhancement Fund, and the San Diego Association of County Government TransNet Environmental Mitigation Program (NMFS 2016).

7.3.4 California Steelhead Report and Restoration Card

The California Steelhead Report and Restoration Card program has funded various types of conservation projects since 1993, including instream habitat improvement, species monitoring, outreach and education, and watershed assessment and planning. However, no restoration projects within the Southern SH/RT range were funded between 2015 and 2019, as most funds were granted to projects in more northern watersheds (CDFW 2021c).

7.3.5 Non-Governmental Organization (NGOs) Efforts

Several NGOs contribute funding and staff time to implement restoration projects for the benefit of Southern SH/RT, often with the support of federal, state, or local grants. For example, the South Coast Steelhead Coalition under the guidance of California Trout, has received grant funding from the Department's FRGP to implement several restoration projects that benefit Southern SH/RT, including the Harvey Diversion Fish Passage Project on Santa Paula Creek; the Interstate 5 Trabuco Fish Passage Project on San Juan Creek in Orange County, the Santa Margarita River Fish Passage Project on Sandia Creek in San Diego County; the Rose Valley Restoration Project on Sespe Creek; invasive vegetation removal in the Santa Clara River floodplain; and *O. mykiss* protection in the upper Santa Margarita River, West Fork San Luis Rey River, and upper tributaries to the Santa Clara and Ventura rivers (NMFS 2016). Other NGOs that promote funding and implementation of steelhead recovery actions include the Santa Clara River Steelhead Coalition under the direction of California Trout, the Tri-Counties Fish Team, the Environmental Defense Center, the San Gabriel and Lower Los Angeles Rivers Mountain Conservancy, the West Fork San Gabriel River Conservancy, and the Council for Watershed Health (San Gabriel and Los Angeles rivers). Additionally, there are many other

groups or agencies that are also involved in Southern SH/RT conservation efforts: Concerned Resource and Environmental Workers; Heal the Ocean; Santa Barbara ChannelKeeper; Matilija Coalition; Ojai Valley Land Conservancy; Friends of the Ventura River; Friends of the Santa Clara River; Friends of the Los Angeles River; Friends of the Santa Monica Mountains; Heal the Bay; Friends of the Santa Margarita River; San Dieguito River Valley Conservancy; and the Endangered Habitat League (NMFS 2016).

7.3.6 Other Regional and Local Public Institution Efforts

The Southern California Wetlands Recovery Project (SCWRP) consists of directors and staff from 18 public agencies, which collectively coordinate to protect, restore, and enhance coastal wetlands and watersheds between Point Conception and the Mexican Border. The SCWRP, which was founded in 1997, is chaired by the California Natural Resources Agency with support from the California State Coastal Conservancy. The mission of the SCWRP is to expand, restore, and protect wetlands in southern California. The SCWRP is guided by long-term goals, specific implementation strategies, and quantitative objectives articulated in its 2018 regional strategy report (SCWRP 2018).

The Southern California Coastal Water Research Project (SCCWRP) is a public research and development agency whose mission is to enhance the scientific foundation for management of southern California's ocean and coastal watersheds. Since its creation in 1969, the focus of the SCCWRP has been to develop strategies, tools, and technologies to improve water quality management for the betterment of the ecological health of the region's coastal ocean and watersheds. SCCWRP research projects are guided by comprehensive annual plans for major research areas, including ecohydrology, climate change, eutrophication, microbial water quality, and stormwater best management practices (SCCWRP 2022). Currently, the SCCWRP, in cooperation with other local and state agencies, is leading the Los Angeles River Environmental Flows Project. The project's goals are to quantify the relationship between flow and aquatic life, account for flow reduction allowances to the river from multiple wastewater reclamation plants during the summer months and develop flow criteria for the Los Angeles River using the California Environmental Flows Framework.

The City of Santa Barbara supports a Creeks Restoration and Water Quality Improvement Division (Creeks Division), whose mission is to improve creek and ocean water quality and restore natural creek systems through storm water and urban runoff pollution reduction, creek restoration, and community education programs. The Creeks Division's goal for restoration includes increasing riparian vegetation and wildlife habitat, removing invasive plants, and improving water quality through shading, bank stabilization, and erosion control. The Division has completed several restoration projects in Santa Barbara County, including the Mission

Creek Fish Passage project, the Arroyo Burro Estuary and Mesa Creek restoration project, and the upper Las Positas Creek restoration project. The Creeks Division also conducts removal efforts of invasive giant reed from the Arroyo Burro, Mission, and Sycamore Creek watersheds and participates in water quality improvement projects, creek and beach cleanups, and education outreach efforts throughout Santa Barbara County.

The California Conservation Corps Fisheries Program gives U.S. military veterans opportunities to develop skills and work experience by restoring habitat for endangered salmon and steelhead and conducting fisheries research and monitoring. The program, which is a partnership between the California Conservation Corps, NMFS, and the Department, trains participants on a variety of fisheries monitoring techniques, including riparian restoration, dual-frequency identification sonar (DIDSON) techniques, adult and juvenile fish identification, downstream migrant trapping, and instream flow and habitat surveys.

7.4 Commercial and Recreational Fishing

California freshwater sport fishing regulations prohibits fishing in virtually all anadromous coastal rivers and streams in southern California that are accessible to adult steelhead. However, recreational angling for *O. mykiss* above impassable barriers is permitted in many coastal rivers and streams (CDFW 2023a). The Department has expanded its use of sterile “triploid” fish to prevent interbreeding of hatchery fish with native Southern SH/RT (NMFS 2016). The freshwater exploitation rates of Southern SH/RT are likely very low given the Department’s prohibition of angling within the geographic range of the Southern California Steelhead DPS listed under the federal ESA (NMFS 2016). Additionally, sport and commercial harvest of Southern SH/RT greater than 16 inches in length in the Department’s Southern Recreational Fishing Management Zone is prohibited (CDFW 2023b). All incidentally captured steelhead in the ocean must be released unharmed and should not be removed from the water.

7.5 Research and Monitoring Programs

7.5.1 California Coastal Monitoring Program

The purpose of the CMP is to gather statistically sound and biologically meaningful data on the status of California’s coastal salmonid populations to inform salmon and steelhead recovery, conservation, and management activities. The CMP framework is based on four viable salmonid population metrics: abundance, productivity, spatial structure, and diversity (Adams et al. 2011; McElhany et al. 2000). Boughton et al. (2022b) updated the CMP approach for the southern coastal region to address the scientific uncertainty on Southern SH/RT ecology due to lower abundances and a more arid climate compared to more northern populations, for which the original CMP framework was designed.

Currently, the Department leads monitoring efforts in the southern coastal region, with most efforts focused on obtaining abundance estimates for anadromous adults in Core 1 and Core 2 populations (NMFS 2016). As of March 2023, Department CMP staff operate fixed-point counting stations and conduct summer-low flow juvenile surveys, redd surveys, and PIT tagging arrays on the Ventura River, Topanga Creek, and Carpinteria Creek, including the various tributaries to these watersheds. Fixed-point counting stations for anadromous adults are also operated on the Santa Ynez River and its primary tributary, Salsipuedes Creek. Redd surveys and juvenile low-flow surveys also occur in coastal watersheds of the Santa Monica Mountains, such as Big Sycamore Creek, Malibu Creek, Arroyo Sequit Creek, and Solstice Creek. Additionally, the Department conducts spawning surveys in the many watersheds of the Conception Coast, including Jalama, Gaviota, Glenn Annie, San Pedro, Maria Ygnacio, and Mission creeks. Department CMP staff anticipate expanding the number of southern coastal watersheds monitored as landowner agreements and available funding increase (K. Evans, CDFW, personal communication).

7.5.2 Other Monitoring Programs

Several special districts or local governments monitor Southern SH/RT on an annual basis in watersheds that contain federally owned or operated infrastructure. Such monitoring is often required for compliance with monitoring and reporting measures set forth in federal ESA Section 7 Biological Opinions. Although the level of monitoring effort and protocol methods vary between monitoring programs, the data produced by these special districts or local governments are often the longest time-series data available for Southern SH/RT.

The Cachuma Operation and Maintenance Board (COMB) has conducted monitoring within the Lower Santa Ynez River and its tributaries since 1994 as part of the assessment and compliance measures required in the Cachuma Project Biological Opinion. Redd and adult spawner surveys typically occur throughout the winter months, while juvenile snorkel surveys are conducted in the spring, summer, and fall months. Estuary monitoring is also periodically conducted to complement upstream trapping during the migration seasons.

Since 2005, the Casitas Mutual Water District (CMWD) has monitored fish migration at the Robles Fish Passage facility (14 miles upstream from the ocean) on the Ventura River using a VAKI Riverwatcher remote fish monitoring system. CMWD also conducts reach-specific spawner and redd surveys and snorkel surveys at index sites throughout the Ventura River watershed from the winter through late spring (Dagit et al. 2020).

The United Water Conservation District (UWCD) monitors both upstream and downstream migration at the Vern Freeman Diversion Dam (approximately 10 miles upstream from the ocean) using both video-based and motion detection surveillance systems. Monitoring occurs

from January to June when streamflow in the Santa Clara River is high enough to maintain water levels at the passage facility (Booth 2016).

The Resource conservation District of the Santa Monica Mountains (RCDSMM) has monitored Arroyo Sequit, Malibu, and Topanga creeks since the early 2000s. Monitoring typically occurs from January through May and includes snorkel surveys, spawning and rearing surveys, instream habitat surveys, and periodic lagoon surveys (Dagit et al. 2019). Since 2016, the South Coast Steelhead Coalition, under the direction of California Trout, has conducted post-rain reconnaissance surveys in San Juan Creek, San Mateo Creek, the Santa Margarita River, and the San Luis Rey River (Dagit et al. 2020).

8. SUMMARY OF LISTING FACTORS

The Commission's CESA implementing regulations identify key factors relevant to the Department's analyses and the Commission's decision on whether to list a species as endangered or threatened. A species will be listed as endangered or threatened if the Commission determines that the species' continued existence is in serious danger or is threatened by any one or any combination of the following factors: (1) present or threatened modification or destruction of its habitat; (2) overexploitation; (3) predation; (4) competition; (5) disease; or (6) other natural occurrences or human-related activities (Cal. Code Regs., tit. 14, § 670.1, subd. (i)). This section provides summaries of information from the preceding sections of this Status Review, arranged under each of the factors to be considered by the Commission in determining whether listing is warranted.

8.1 Present or Threatened Modification or Destruction of Habitat

The decline of Southern SH/RT can be attributed to a wide variety of human activities, including, but not limited to, urbanization, agriculture, and water development. These activities have degraded range-wide aquatic habitat conditions, particularly in the lower and middle reaches of individual watersheds (see Section 6.8). Southern California is home to over 20 million people and 1.8 million acres of urban area (DWR 2021). As a result, the majority of watersheds, currently occupied by Southern SH/RT, are highly urbanized and impacted by surface and groundwater diversions and associated agricultural, residential, and industrial uses.

Although some deleterious activities have been eliminated or mitigated, habitat conditions for Southern SH/RT have continued to deteriorate over time due to numerous stressors associated with human population growth and climate change impacts. Water diversions, storage, and conveyance for agriculture, flood control, and domestic uses have significantly reduced much of their historical spawning and rearing habitat. Water storage facilities, reservoir operations, instream diversions and groundwater extractions have altered the natural flow regime of

southern California rivers and streams and have led to warmer water temperatures, shifts in aquatic community structure and composition, and reduced downstream recruitment of gravel and sediments. High road densities and the presence of in-stream artificial barriers have reduced habitat connectivity by impeding and restricting volitional fish passage in many watersheds, especially in the lower reaches. Development activities associated with agriculture, urbanization, flood control, and recreation have also substantially altered Southern SH/RT habitat quantity and quality by increasing ambient water temperatures, increasing nutrient and pollutant loading, degrading water quality, eliminating riparian habitat, and creating favorable conditions for non-native species. Range-wide and coastal estuarine habitat conditions are highly degraded and are at risk of loss and further degradation. Legal cannabis cultivation is a relatively new yet potentially serious threat to Southern SH/RT watersheds if best management practices, instream flow requirements, and diversion season regulations are not complied with. Our review of habitat conditions in southern California supports the conclusions of other review efforts, which conclude that populations continue to be at risk of extinction unless significant restoration and recovery measures are implemented (Moyle et al. 2017; NMFS 2012a).

The Department considers present or threatened modification or destruction of habitat to be a significant threat to the continued existence of Southern SH/RT.

8.2 Overexploitation

Exploitation rates of Southern SH/RT are relatively low across its range (see Section 6.9). While angling-related mortality may have historically contributed to the decline of some small populations, it is generally not considered a leading cause of the decline of the Southern California Steelhead DPS as a whole (Good et al. 2005; Busby et al. 1996; NMFS 1996b). After southern California steelhead was first listed as endangered under the federal ESA as an ESU in 1997, the Commission closed recreational fisheries for Southern SH/RT in California marine and anadromous waters with few exceptions. The closure continues, and there is currently no recreational fishery for Southern SH/RT (CDFW 2023a; CDFW 2023b).

Marine commercial driftnet fisheries in the past may have contributed slightly to localized declines; however, Southern SH/RT are not targeted in commercial fisheries and reports of incidental catch are rare. Commercial fisheries are not thought to be a leading cause of the widespread declines over the past several decades (NMFS 2012a).

Illegal harvest is likely the leading source of exploitation. Southern SH/RT are especially vulnerable to poaching due to their visibility in shallow streams. Estimates of fishing effort from self-report cards for 1993-2014 suggest extremely low levels of angling effort for Southern SH/RT (NMFS 2016; Jackson 2007). Though illegal harvest rates appear to be very low, because

of low adult abundance, the removal of even a few individuals in some years could be a threat to the population (Moyle et al. 2017).

The Department does not consider overexploitation to be a substantial threat to the continued existence of Southern SH/RT, but further directed study is warranted to confirm this threat level.

8.3 Predation

Southern SH/RT experience predation in both the freshwater and marine environments, but specific predation rates, particularly in marine environments, are not well understood (see Section 6.5). While Southern SH/RT have evolved to cope with a variety of natural predators, a suite of non-native predators has also become established within its watersheds (Busby et al. 1996; NMFS 2016; Stillwater Sciences 2019; Dagit et al. 2019; COMB 2022). Established populations of non-native fishes, amphibians, and aquatic invertebrates combined with anthropogenic habitat alterations that provide favorable conditions for the persistence of these non-native species have led to increased predation rates in much of its range (NMFS 1996b). Habitat modification and degradation has also likely increased predation rates from terrestrial and avian predators (Grossman 2016; Osterback et al. 2013).

Further directed study is warranted to assess the level of impact of these predation threats on Southern SH/RT.

8.4 Competition

Southern SH/RT populations are subject to competitive forces across their range (see Section 6.6). The extent to which competition impacts the distribution, abundance, and productivity of Southern SH/RT populations is not well understood. Southern SH/RT are the only salmonid that occur in their range. Therefore, the potential for inter-specific competition with other salmonids is unlikely to occur. Interspecific competition with other non-salmonid fishes occurs to varying degrees across the Southern SH/RT range. In addition to competing with juvenile steelhead for food resources, juvenile non-native fish species can limit the distribution and abundance of juvenile steelhead. Non-native fish species can competitively exclude and confine the spatial distribution of juvenile steelhead to habitats such as shallower, higher velocity riffles, where the energetic cost to forage is higher (Rosenfeld and Boss 2001).

Further directed study is warranted to assess the level of impact of competition from non-native fish species.

8.5 Disease

Southern SH/RT survival is impacted by a variety of factors including infectious disease (see Section 6.3). A myriad of diseases caused by bacterial, protozoan, viral, and parasitic organisms can infect *O. mykiss* in both the juvenile and adult life stages (NMFS 2012a). Degraded water quality and chemistry in much of the Southern SH/RT range is likely to increase infection rates and severity (Belchik et al. 2004; Stocking and Bartholomew 2004; Crozier et al. 2008). There is very little current information available to quantify present infection and mortality rates in Southern SH/RT.

The Department does not consider disease to currently be a significant threat to the continued existence of Southern SH/RT, however further directed study is warranted to confirm the level of current and potential future impact.

8.6 Other Natural Occurrences or Human-related Activities

Southern SH/RT populations have evolved notably plastic and opportunistic survival strategies and are uniquely adapted to wide-ranging natural environmental variability, characterized by challenging and dynamic habitat conditions (Moyle et al. 2017). However, combined anthropogenic and climate change-driven impacts may ultimately outpace Southern SH/RT's capacity to adapt and persist, potentially leading to extirpation within the next 25–50-year time frame (Moyle et al. 2017; see Section 6.2). This prediction is underscored by the fact that Southern SH/RT already encounters water temperatures that approach and may, at times, exceed the upper limit of salmonid thermal tolerances, across portions of its current distribution (Moyle et al. 2017). Southern SH/RT has, therefore, been characterized as having potential for severe climate change impacts (Moyle et al. 2017). With increasing exposure to periods of higher water temperatures and flow variability, along with extended droughts, more frequent and intense wildfires, catastrophic flooding and associated sediment movement, sea level rise, and ever-increasing human demands for natural resources, the combined impacts to Southern SH/RT will be interdependent, synergistic, and are expected to intensify without intensive and timely human intervention (NMFS 2012b; Hall et al. 2018; OEHHA 2022).

Human-related activities are considered by the Department to be significant threats to the continued existence of Southern SH/RT.

9. SUMMARY OF KEY FINDINGS

Southern California steelhead (*Oncorhynchus mykiss*) inhabit coastal streams from the Santa Maria River system south to the U.S.-Mexico border. Non-anadromous resident *O. mykiss*, familiar to most as Rainbow Trout, reside in many of these same streams and interbreed with

anadromous adults, contributing to the overall abundance and resilience of the populations. Southern SH/RT as defined in the Petition include both anadromous (ocean-going) and resident (stream-dwelling) forms of *O. mykiss* below complete barriers to anadromy in these streams.

Less than half of the watersheds historically occupied by Southern SH/RT remain occupied below complete barriers to anadromy, most commonly with individuals able to express only a freshwater-resident life-history strategy (NMFS et al. 2012). Adult steelhead runs have declined to precariously low levels, particularly over the past five to seven years, with declines in adult returns of 90% or more on major watersheds that historically supported the largest anadromous populations (e.g., the Santa Maria, Santa Ynez, Ventura, and Santa Clara rivers). Additionally, our analysis of resident populations indicates a sharp decline over this same time period.

While recent genetic findings suggest that the anadromous life-history form can be sustained and reconstituted from resident individuals residing in orographic drought refugia, in southern California, nearly all drought refugia habitats are currently above impassable barriers. Therefore, the anadromous phenotype is at an increasingly high risk of being entirely lost from the species within its southern California range, in large part due to the lack of migration corridors between drought refugia and the ocean, and the inability of resident progeny to successfully migrate downstream in years with sufficient rainfall and streamflow.

Southern SH/RT continues to be most at risk from habitat degradation, fragmentation, and destruction resulting from human-related activities. Specifically, dams, surface water diversions, and groundwater extraction activities restrict access to most historical spawning and rearing habitats and alter the natural flow regime of rivers and streams that sustain ecological, geomorphic, and biogeochemical functions and support the specific life history and habitat needs of Southern SH/RT. Agricultural and urban development negatively affect nearby rivers and streams through increased pollution and surface runoff, which degrade water quality and habitat conditions. Furthermore, the rapid rate of climate change and the increasing presence of non-native species present another challenge to the persistence of Southern SH/RT.

Based on the best scientific information available at the time of the preparation of this review, the Department concludes that the Southern SH/RT is in danger of extinction throughout all of its range. Intensive and timely human intervention, such as ecological restoration, dam removal, fish passage improvement projects, invasive species removal, and groundwater management, are required to prevent the further decline of Southern SH/RT. The extinction of Southern SH/RT would represent an insurmountable loss to the *O. mykiss* diversity component in California due to their unique adaptations, life histories, and genetics, which have allowed them to persist at the extreme southern end of the species' West Coast range.

10. RECOMMENDATION FOR THE COMMISSION

CESA requires the Department to prepare this report regarding the status of Southern SH/RT in California based upon the best scientific information available to the Department (Fish & G. Code, § 2074.6). CESA also requires the Department to indicate in this Status Review whether the petitioned action (i.e., listing as endangered) is warranted (Fish & G. Code, § 2074.6; Cal. Code Regs., tit. 14, § 670.1, subd. (f)).

Under CESA, an endangered species is defined as “a native species or subspecies...which is in serious danger of becoming extinct throughout all, or a significant portion, of its range due to one or more causes, including loss of habitat, change in habitat, overexploitation, predation, competition, or disease” (Fish & G. Code, § 2062). A threatened species is defined as “a native species or subspecies...that, although not presently threatened with extinction, is likely to become an endangered species in the foreseeable future in the absence of the special protection and management efforts required by [CESA]” (Fish and G. Code, § 2067).

Based on the criteria described above, the best scientific information available to the Department indicates that Southern SH/RT is in serious danger of becoming extinct in all of its range due to one or more causes including: 1. present or threatened modification or destruction of habitat; and 2. other natural occurrences or human-related activities. The Department recommends that the Commission find the petitioned action to list Southern SH/RT as an endangered species to be warranted.

11. PROTECTION AFFORDED BY LISTING

It is the policy of the State to conserve, protect, restore, and enhance any endangered or threatened species and its habitat (Fish & G. Code, § 2052). The conservation, protection, and enhancement of listed species and their habitat is of statewide concern (Fish & G. Code, § 2051, subd. (c)). If listed, unauthorized take of Southern SH/RT would be prohibited under state law. CESA defines “take” as hunt, pursue, catch, capture, or kill, or attempt to hunt, pursue, catch, capture, or kill (Fish & G. Code, § 86). Any person violating the take prohibition would be punishable under state law. The Fish and Game Code provides the Department with related authority to authorize “take” of species listed as threatened or endangered under certain circumstances (see, e.g., Fish & G. Code, §§ 2081, 2081.1, 2086, & 2835). If Southern SH/RT is listed under CESA, take resulting from activities authorized through incidental take permits must be minimized and fully mitigated according to state standards (Fish & G. Code, § 2081, subd. (b)). Take of Southern SH/RT for scientific, educational, or management purposes could be authorized through permits or memorandums of understanding pursuant to Fish and Game Code Section 2081(a).

Additional protection of Southern SH/RT following listing would also occur during required state and local agency environmental review under CEQA. CEQA requires affected public agencies to analyze and disclose project-related environmental effects, including potentially significant impacts on endangered, threatened, and rare special status species. Under CEQA’s “substantive mandate,” state and local agencies in California must avoid or substantially lessen significant environmental effects to the extent feasible. With that mandate, and the Department’s regulatory jurisdiction generally, the Department expects related CEQA review will likely result in increased information regarding the status of Southern SH/RT in California as a result of pre-project biological surveys. Where significant impacts are identified under CEQA, the Department expects project-specific required avoidance, minimization, and mitigation measures will also benefit the species. While CEQA may require analysis of potential impacts to Southern SH/RT regardless of its listing status under CESA, the act contains specific requirements for analyzing and mitigating impacts to listed species. In common practice, potential impacts to listed species are scrutinized more in CEQA documents than are potential impacts to unlisted species. State listing, in this respect, and required consultation with the Department during state and local agency environmental review under CEQA, is expected to benefit the species by reducing impacts from individual projects to a greater degree than may occur absent listing.

CESA listing may prompt increased interagency coordination specific to Southern SH/RT conservation and protection. Listing may also increase the likelihood that state and federal land and resource management agencies will allocate additional funds toward protection and recovery actions.

12. MANAGEMENT RECOMMENDATIONS AND RECOVERY MEASURES

CESA directs the Department to include in its Status Review recommended management activities and other recommendations for recovery of Southern SH/RT (Fish & G. Code, § 2074.6; Cal. Code Regs., tit. 14, § 670.1, subd. (f)). Department staff generated the following list of recommended management actions and recovery measures.

1. Implement comprehensive monitoring in all streams with extant Southern SH/RT populations and produce statistically robust population estimates. Fully implement the California Coastal Monitoring Program and integrate the updated south coastal region monitoring strategy (Boughton et al. 2022b) to resolve the various ecological and methodological factors that currently impede monitoring. The main features of this updated strategy are:

- Estimates of average density for each BPG;
- Research on the location and extent of drought refugia in each BPG;

- Adult steelhead abundance estimates in selected populations that are robust enough to evaluate Southern SH/RT resilience to catastrophic events and the ability to adapt over time to long-term environmental changes;
- Adult *O. mykiss* abundance estimates that are sufficient to develop an estimate for total abundance in the region; and
- Greater emphasis on monitoring methods that are unbiased or can be corrected for bias (NMFS 2016).

2. Support and participate in the development of watershed-specific plans to effectively maintain and restore Southern SH/RT habitat by focusing on the combination of factors currently limiting their distribution and abundance, such as dams, agriculture, and water extraction. This includes continuing to coordinate and collaborate with NMFS, NGOs, state and local governments, landowners, and other interested entities to implement recovery actions identified in the 2012 Recovery Plan for the southern California Steelhead DPS and other management and conservation strategies. High priority actions include (NMFS 2012a):

- Remove manmade passage barriers in all population watersheds and re-establish access to upper watersheds in both small coastal streams and the larger interior rivers within each BPG identified in the federal Recovery Plan;
- Establish fishways or assisted migration practices at manmade passage barriers that cannot be removed in the near-term with an emphasis on re-establishing passage for above-barrier populations that still contain significant native ancestry;
- Complete planning and removal of Matilija Dam on Matilija Creek and Rindge Dam on Malibu Creek;
- Provide ecologically meaningful flows below major dams and diversions in all population watersheds by re-establishing adequate flow regimes and restoring groundwater aquifers in dewatered areas to sustain surface flows in both small coastal streams and large interior rivers;
- Reevaluate the efficacy of existing fish passage structures at instream surface water diversions, dams, culverts, weirs, canals, and other infrastructure in all watersheds historically and currently occupied by Southern SH/RT; and
- Minimize the adverse effects of exotic and non-native plant and animal species on aquatic ecosystems occupied by Southern SH/RT through direct removal and control efforts.

3. Improve and expand suitable and preferred habitat used by Southern SH/RT for summer holding, spawning, and juvenile rearing. Prioritize habitat restoration, protection, and enhancement in Southern SH/RT holding, spawning, and rearing areas. Habitat projects should focus on improving habitat complexity, riparian cover, fish passage, and sediment transport, as

well as enhancing essential deep, cold-water habitats for holding adults. Restoration should also be considered in potential habitats not currently occupied by Southern SH/RT.

4. Continue research on *Omy5* haplotypes and other relevant genomic regions to better understand: the mechanism for anadromy in Southern SH/RT, the impact of migration barriers on the frequency of the “A” haplotype in individuals, and the risk of progressively losing the genetic basis for anadromy over time in above-barrier populations despite the current presence of the “A” haplotype.

5. Continue to investigate the population structure and ancestry of Southern SH/RT at the extreme southern end of the species distribution in southern California, including further research on identifying genetically introgressed populations and the potential benefit of these populations for maintaining the persistence of viable networks of Southern SH/RT, given recent findings of limited native ancestry in the region and the importance of variation in adaptation.

6. Initiate research into Southern SH/RT ecology identified in the Southern California Steelhead Recovery Plan (NMFS 2012a). Important research topics include:

- Environmental factors that influence anadromy;
- The relationship between migration corridor reliability and anadromous fraction;
- Identification of nursery habitat types that promote juvenile growth and survival;
- The role of seasonal lagoons and estuaries in the life history of Southern SH/RT and the extent to which these areas are used by juveniles prior to emigration;
- Investigation on the role that mainstem habitats play in the life history of steelhead, including identification of the ecological factors that contribute to mainstem habitat quality;
- The role of naturally intermittent creeks and stream reaches;
- Determining whether spawner density is a reliable indicator of a viable population;
- Determining the frequency of return adult spawners;
- Recolonization rates of extirpated watersheds by source populations;
- Dispersal rates between watersheds, including interactions among and between populations through straying;
- Intra-and interannual variation in diet composition and growth rate; and
- Partial migration and life-history crossovers.

7. Formalize minimization and avoidance measures on a Department-wide basis to minimize incidental take of the CESA-listed species due to otherwise lawful activities resulting from construction, research, management, and enhancement activities. This includes working with federal agencies to coordinate and develop efficient permitting processes for incidental take authorization for actions that contribute to the recovery of Southern SH/RT.

8. Explore other means of conserving individual populations of Southern SH/RT that may face the risk of extirpation due to catastrophic events, such as wildfires, droughts, and oil spills (e.g., conservation translocations to other existing facilities at academic institutions or museums, or natural refugia habitats). This includes ensuring that translocations of Southern SH/RT conducted by the Department for conservation purposes significantly contribute to species and ecosystem conservation and are planned, executed, and supported in a manner consistent with best scientific practices and the Department's Policy and Procedures for Conservation Translocations of Animals and Plants (CDFW 2017).
9. Strengthen law enforcement in areas occupied by Southern SH/RT to reduce threats of poaching, illegal water diversions, and instream work used for cannabis cultivation.
10. Evaluate current fishing regulations to determine any potential changes that could be implemented for further protection of Southern SH/RT, and update regulations, using clear and transparent communication, in response to restoration actions, such as dam removal projects, that could change the sport fishing regulation boundary (e.g., inland anadromous waters).
11. Conduct a robust outreach and education program that works to engage with tribes and interested parties, including federal, state, local, NGOs, landowners, underserved communities, and interested individuals, to promote and implement conservation actions. This includes developing outreach and educational materials to increase public awareness and knowledge of the ecological and societal benefits that can be gained by recovering Southern SH/RT.

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Personal Communication

Kyle Evans, CDFW, personal communication, 03/08/2023

John O'Brien, CDFW, personal communication, 12/05/2022

Dane St. George, CDFW, personal communication, 05/24/2023

David Boughton, NOAA, personal communication, 09/20/2023

Camm Swift, Emeritus, Section of Fishes, Natural History Museum of Los Angeles County, personal communication, 09/20/2023

APPENDIX A: ANNUAL *O. MYKISS* OBSERVATIONS AND DATA SOURCES FOR THREE EXTANT POPULATIONS IN THE CONCEPTION COAST BPG.

Year	Arroyo Sequit Creek ^a	Topanga Creek ^b	Malibu Creek ^b
2001	0	2	NA
2002	0	95	NA
2003	0	59	NA
2004	0	103	230
2005	0	71	87
2006	0	170	80
2007	0	86	12
2008	0	316	2,245
2009	0	209	130
2010	0	253	160
2011	0	114	281
2012	0	96	156
2013	0	56	99
2014	0	57	31
2015	0	59	32
2016	0	34	7
2017	0	98	6
2018	0	55	1
2019	NA	160	0
Total	0	2,093	3240

"NA" indicates no survey conducted or data not yet available.

^a Source: Dagit et al. (2019)

^b Source: Dagit et al. (2019). Sum of the average number of *O. mykiss* observed per month.

APPENDIX B: ANNUAL ADULT STEELHEAD OBSERVATIONS AND DATA SOURCES FOR THREE EXTANT POPULATIONS IN THE CONCEPTION COAST BPG.

Year	Arroyo Sequit Creek ^a	Topanga Creek ^b	Malibu Creek ^c
2001	0	2	NA
2002	0	0	NA
2003	0	0	NA
2004	0	0	0
2005	0	d	0
2006	0	d	1
2007	0	d	2
2008	0	d	4
2009	0	d	1
2010	0	d	2
2011	0	d	2
2012	0	d	3
2013	0	d	3
2014	0	d	5
2015	0	d	1
2016	0	d	0
2017	2	2	1
2018	0	0	0
2019	NA	0	0
Total	2	12	25

"NA" indicates no survey conducted or data not yet available.

^a Source: Dagit et al. 2020

^b Source: Dagit et al. (2019; 2020)

^c Source: Dagit et al. (2019;2020)

^d Passage barriers prevented access to Arroyo Sequit from 2005-2016. Two adult observations occurred after the removal of barriers (Dagit et al. 2019).

APPENDIX C: ANNUAL *O. MYKISS* OBSERVATIONS AND DATA SOURCES FOR FOUR EXTANT POPULATIONS IN THE MONTE ARIDO HIGHLANDS BPG.

Year	Santa Maria River ^a	Santa Ynez River ^b	Ventura River ^c	Santa Clara River ^d	
1994	NA	NA	NA	87	e
1995	NA	NA	NA	115	e
1996	NA	NA	NA	96	e
1997	NA	NA	NA	422	e
1998	NA	NA	NA	6	e
1999	NA	NA	NA	5	e
2000	NA	NA	NA	876	e
2001	NA	266	NA	124	e
2002	NA	116	NA	3	e
2003	NA	196	NA	41	
2004	NA	238	NA	3	
2005	NA	117	0	NA	
2006	NA	653	17	21	
2007	NA	665	63	74	
2008	NA	561	47	157	
2009	NA	610	807	170	
2010	NA	367	147	100	
2011	NA	484	640	23	
2012	NA	199*	378	96	
2013	NA	NA	17	1	
2014	NA	137*	14	19	
2015	NA	134*	65	NA	
2016	NA	103*	14	NA	
2017	NA	5*	9	NA	
2018	NA	27*	1	NA	
2019	NA	39*	0	NA	
2020	NA	147*	0	NA	
2021	NA	205*	0	NA	

"NA" indicates no survey conducted or data not yet available.

* NMFS incidental take provisions in place. Take limits have not been exceeded since 2014.

^a Source: Santa Maria River does not appear to be monitored for any viability metrics (NMFS 2016)

^b Source: COMB (2022). Data represent the total number of upstream and downstream migrant captures at three trapping locations in the Lower Santa Ynez River basin for each water year (WY).

^c Source: CMWD (2005-2021). Data are derived from snorkel counts and bankside observations from index reaches of the Ventura River near the Robles Diversion.

^d Source: Booth (2016)

^e Inconsistent monitoring from 1994-2002 (Booth 2016)

APPENDIX D: ANNUAL ADULT STEELHEAD OBSERVATIONS AND DATA SOURCES FOR FOUR EXTANT POPULATIONS IN THE MONTE ARIDO HIGHLANDS BPG.

Year	Santa Ynez		Ventura River ^c	Santa Clara River ^d	
	Santa Maria River ^a	River ^b			
1994	NA	NA	NA	1	e
1995	NA	0	NA	1	e
1996	NA	0	NA	2	e
1997	NA	2	NA	0	e
1998	NA	1	NA	0	e
1999	NA	3	NA	1	e
2000	NA	0	NA	2	e
2001	NA	4	NA	2	e
2002	NA	0	NA	0	e
2003	NA	1	NA	0	
2004	NA	0	NA	0	
2005	NA	1	NA	0	
2006	NA	1	4	0	
2007	NA	0	4	0	
2008	NA	16	6	2	
2009	NA	1	0	2	
2010	NA	1	1	0	
2011	NA	9	0	0	
2012	NA	0	0	3	
2013	NA	NA	0	0	
2014	NA	0	0	0	
2015	NA	0	0	0	
2016	NA	0	0	0	
2017	NA	0	0	0	
2018	NA	0	0	0	
2019	NA	0	1	NA	
2020	NA	0	0	NA	
2021	NA	0	1	NA	

"NA" indicates no survey conducted or data not yet available.

^a Source: Santa Maria River does not appear to be monitored for any viability metrics (NMFS 2016)

^b Source: Dagit et al. (2020), COMB (2022)

^c Source: Dagit et al. (2020), CDFW R5 internal data from DIDSON monitoring (2019, 2021)

^d Source: Dagit et al. (2020), Booth (2016)

^e Inconsistent monitoring from 1994-2002 (Booth 2016)

**APPENDIX E: COMMENTS FROM TRIBES AND AFFECTED AND INTERESTED PARTIES ON THE
PETITIONED ACTION**

APPENDIX F: PEER REVIEW SUMMARY