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)

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Southern California Steelhead Rainbow Trout, CDFW photo

Prepared by
California Department of Fish and Wildlife

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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 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 Kamachatka 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**
Oncorhynchus mykiss	A species of Pacific salmonid composed of both anadromous and freshwater-resident forms, which all spawn in freshwater rivers and streams.
Steelhead	Individuals: *O. mykiss* 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: *O. mykiss* 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 *O. mykiss*	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 *Oncorhynchus mykiss* 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. tschawytscha*, 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).

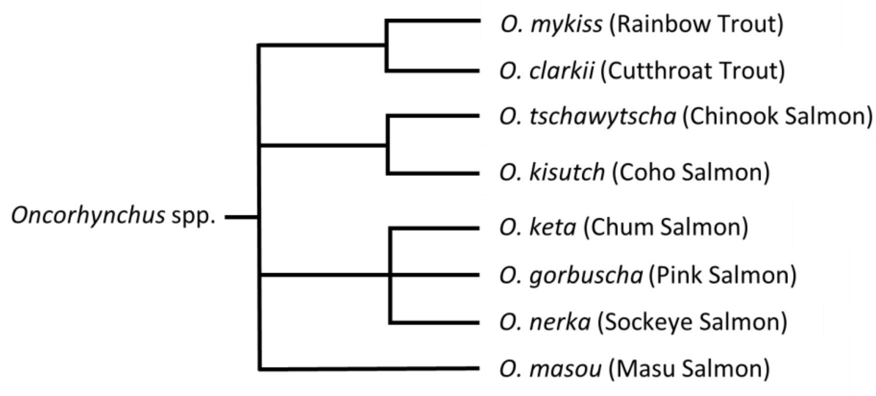


Figure 1. Consensus relationships of Oncorhynchus species from morphological, allozyme, ribosomal RNA, mitochondrial DNA, and short interspersed repetitive elements data across multiple studies. Adapted from Figure 1 in Kitano et al. (1997)

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 $\geq$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).

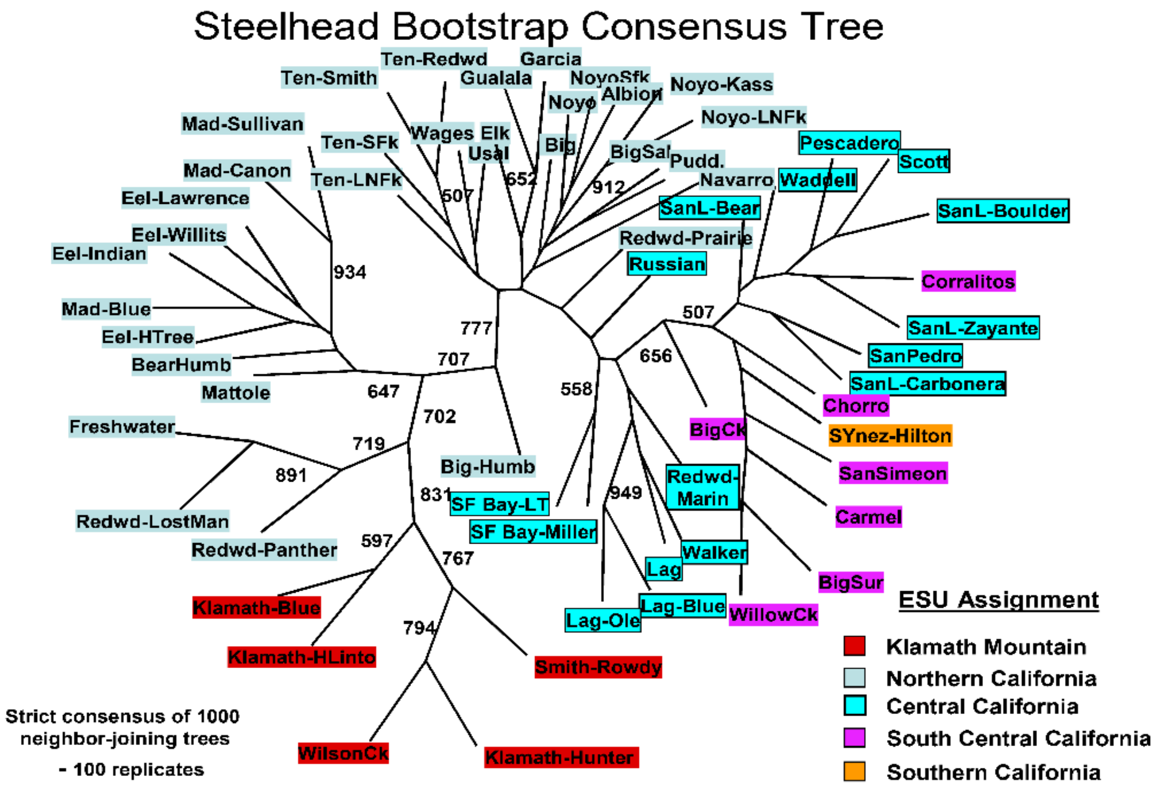


Figure 2. Majority-rule consensus tree, with genetic data bootstrapped 1,000 times, showing chord distances and neighbor-joining trees for 62 coastal California steelhead populations. (from Garza et al. 2004).

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).

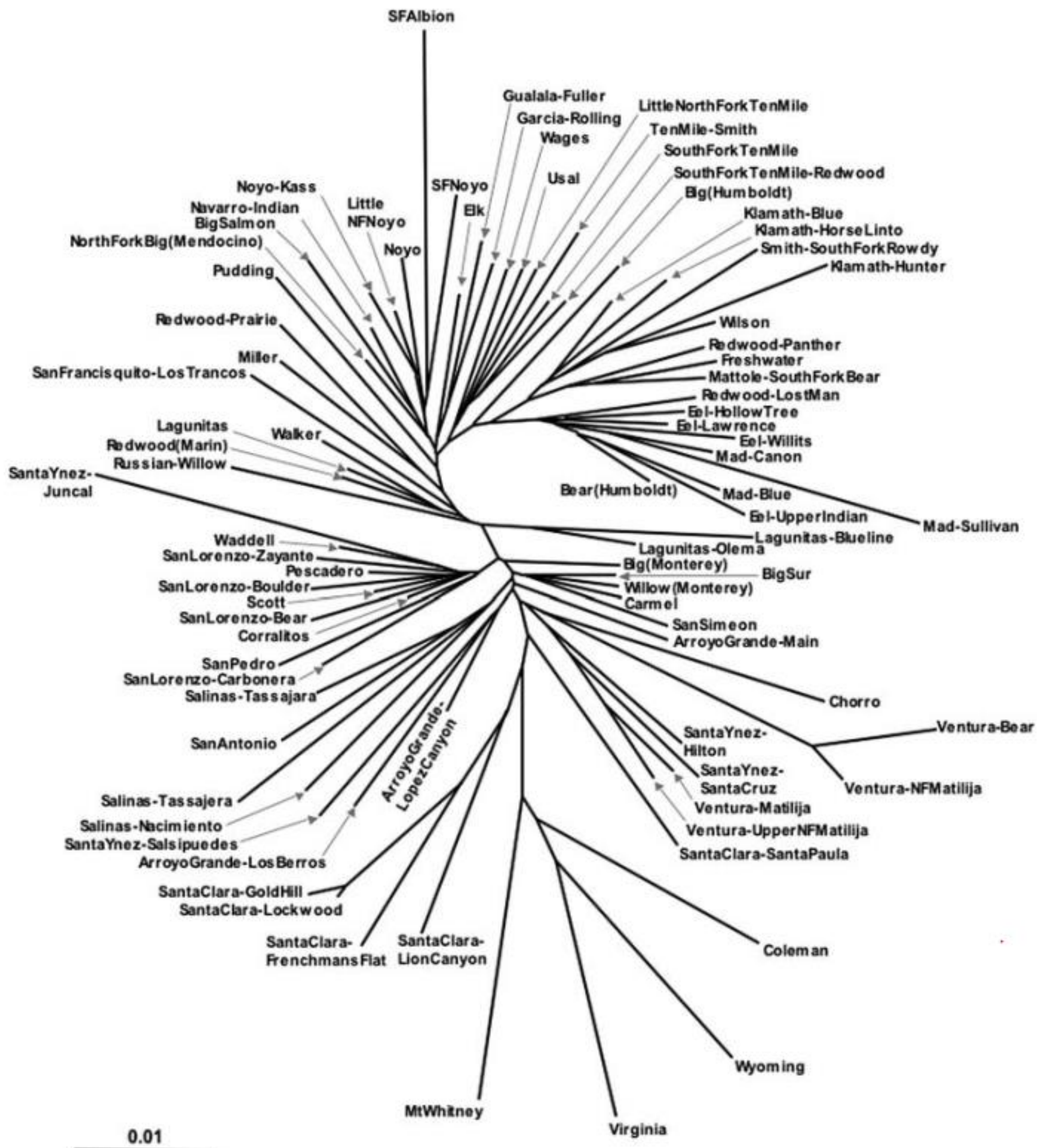


Figure 3. Unrooted neighbor-joining chord distance tree of 84 coastal O. mykiss populations in California (from Clemento et al. 2009).

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.

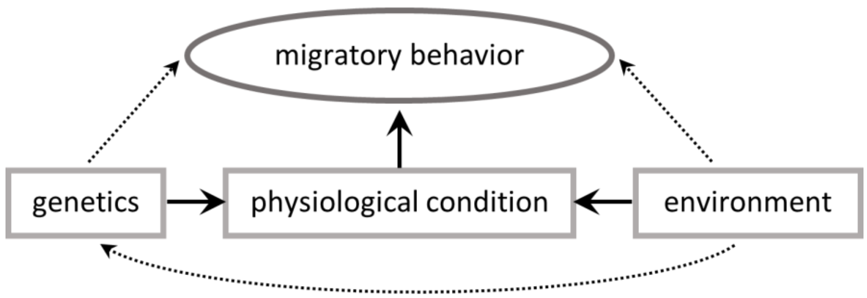


Figure 4. Schematic of indirect genetic control of migratory behavior. Genetic variation and the environment influence physiology, which then impacts migratory behavior (adapted from Kelson et al. 2020).

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).

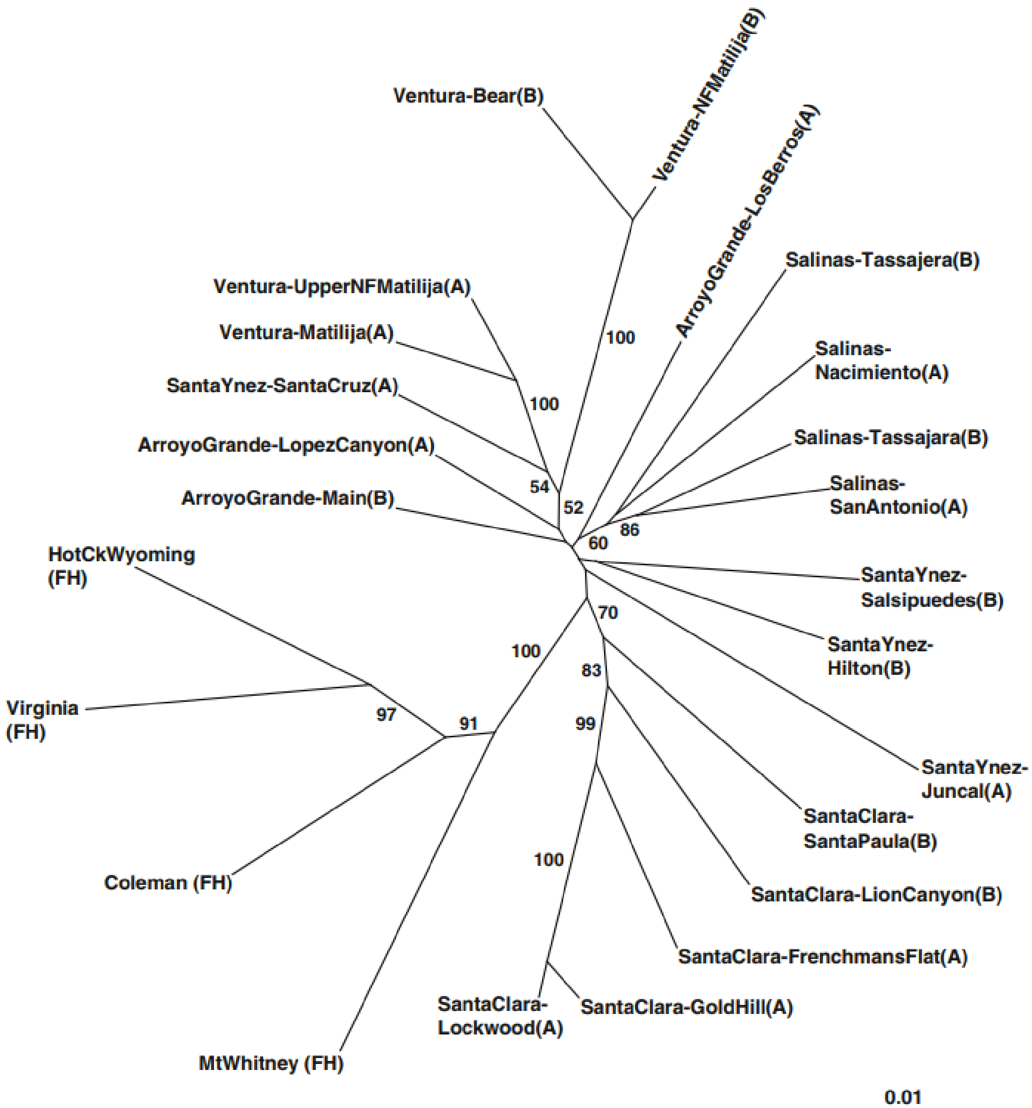


Figure 5. Unrooted neighbor-joining dendogram showing chord distances between 24 sampled naturally spawning populations both above and below barriers, denoted with A and B, respectively. Strains of Rainbow Trout from Fillmore Hatchery used for regional stocking are indicated with FH. Numbers associated 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. Secondarily, 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; Clemento 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.

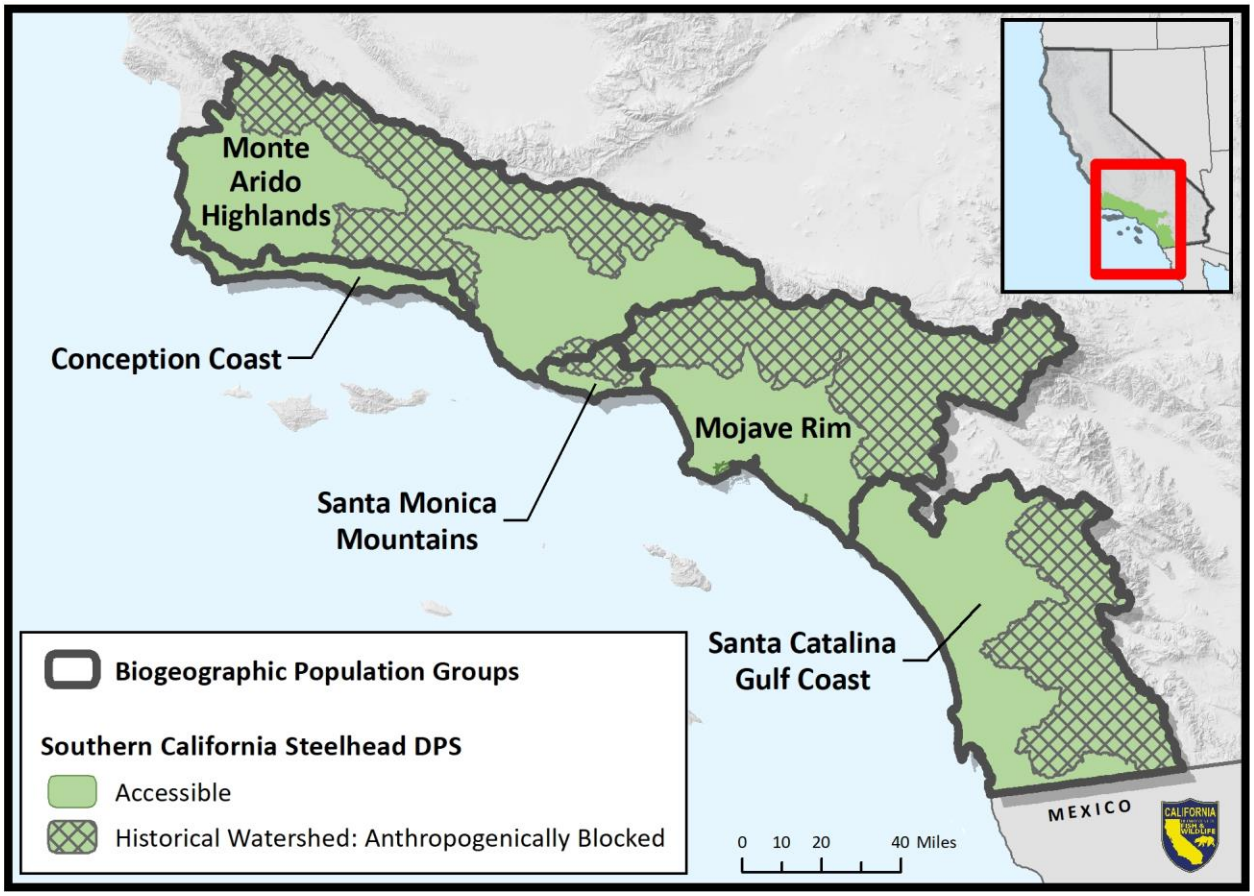


Figure 6. Map of the current and historical distribution of Southern SH/RT. BPGs represented are the Monte Arido Highlands, Conception Coast, Santa Monica Mountains, Mojave Rim, and Santa Catalina Gulf Coast.

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).

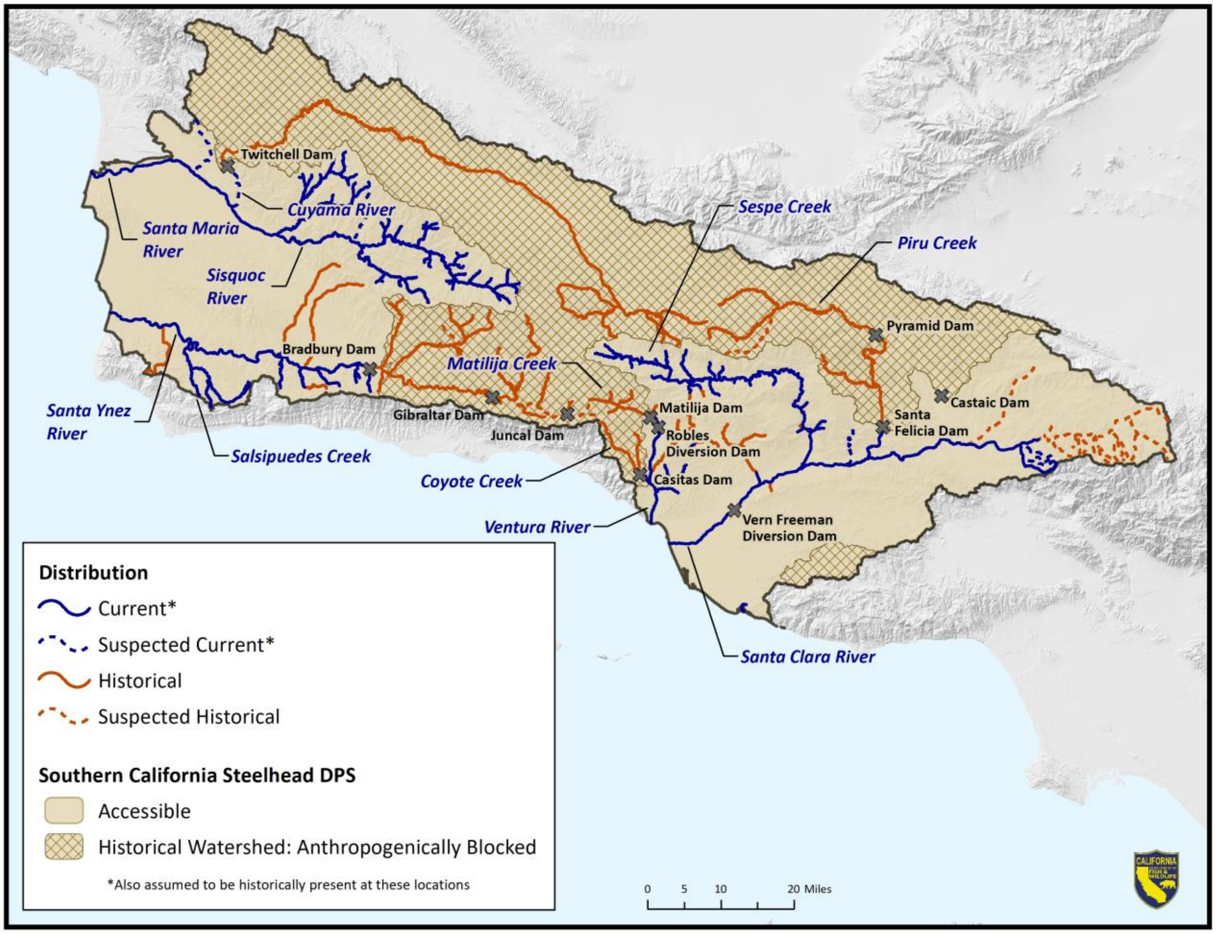


Figure 7. Map of the Monte Arido Highlands BPG depicting known and suspected current and historical distribution.

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-Prepration-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 Sisar 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).

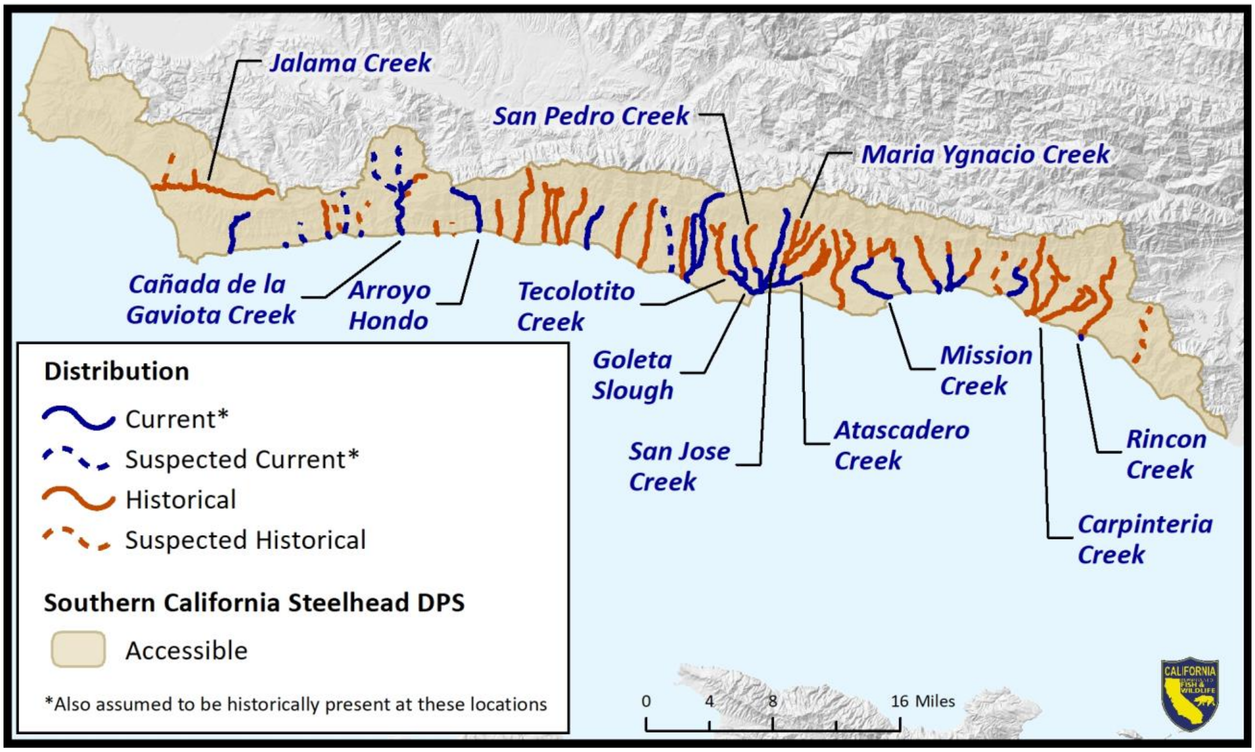


Figure 8. Map of the Conception Coast BPG depicting known and suspected current and historical distribution.

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.

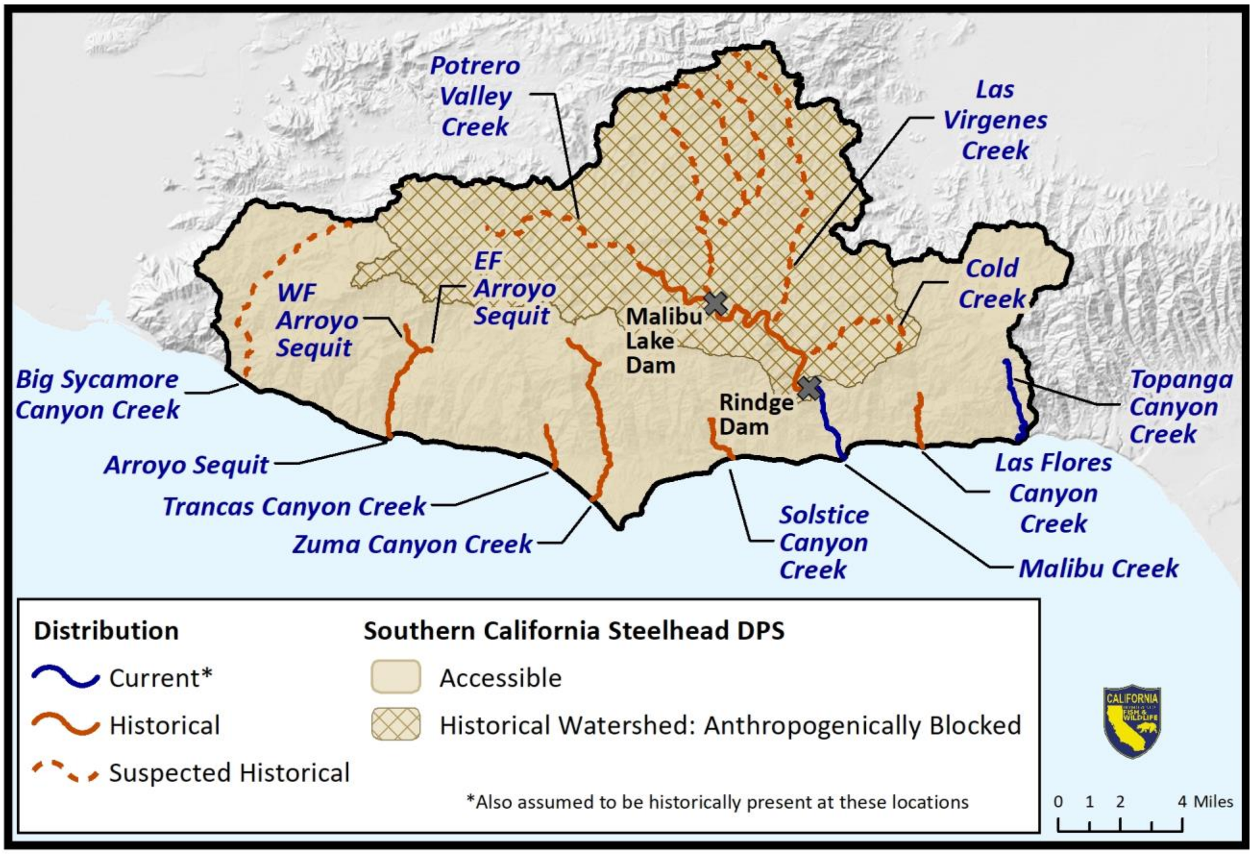


Figure 9. Map of the Santa Monica Mountains BPG depicting known and suspected current and historical distribution. Abbreviations: EF = East Fork, WF = West Fork.

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 a. 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).

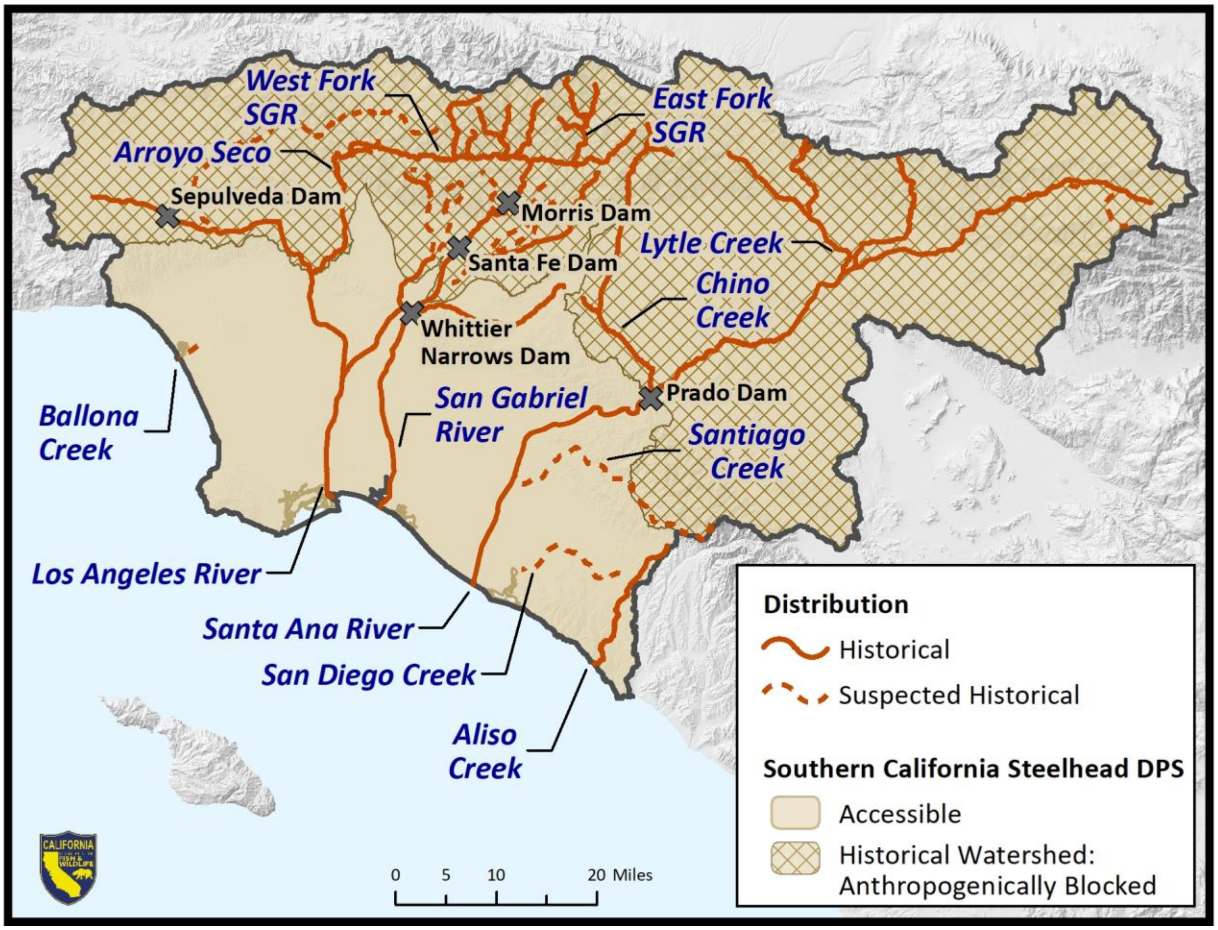


Figure 10. Map of the Mojave Rim BPG depicting known and suspected current and historical distribution. Abbreviations: SGR= San Gabriel River.

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).

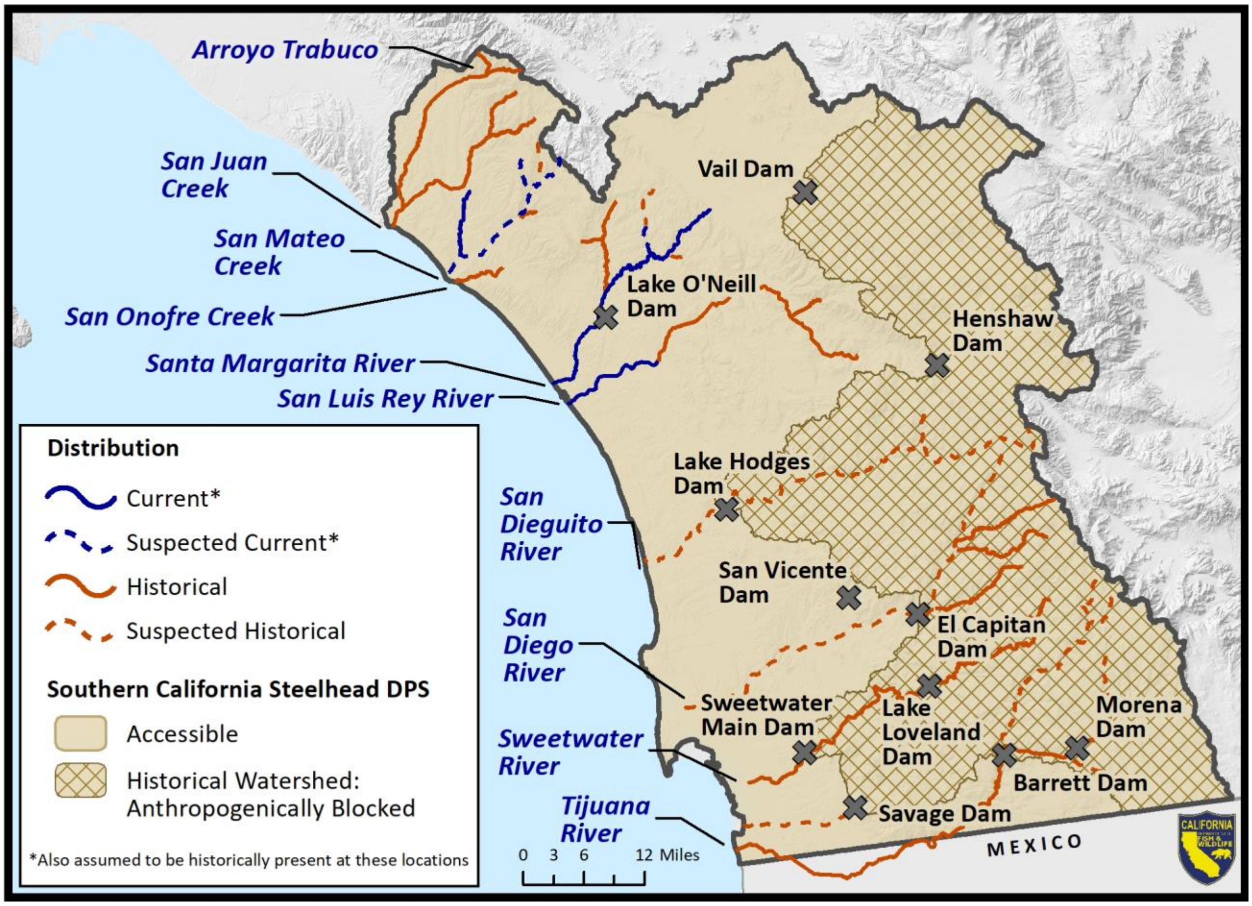


Figure 11. Map of the Santa Catalina Gulf Coast BPG depicting known and suspected current and historical distribution.

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 considered 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 >400m), 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).

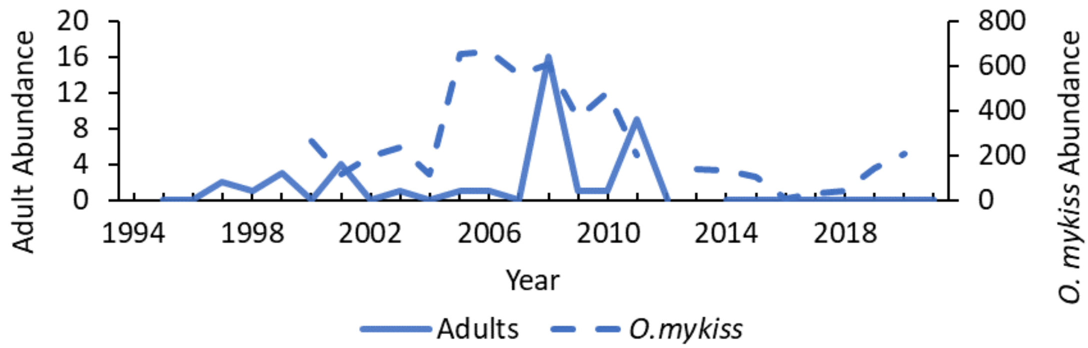
4.4.1 Time Series of Abundance

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.

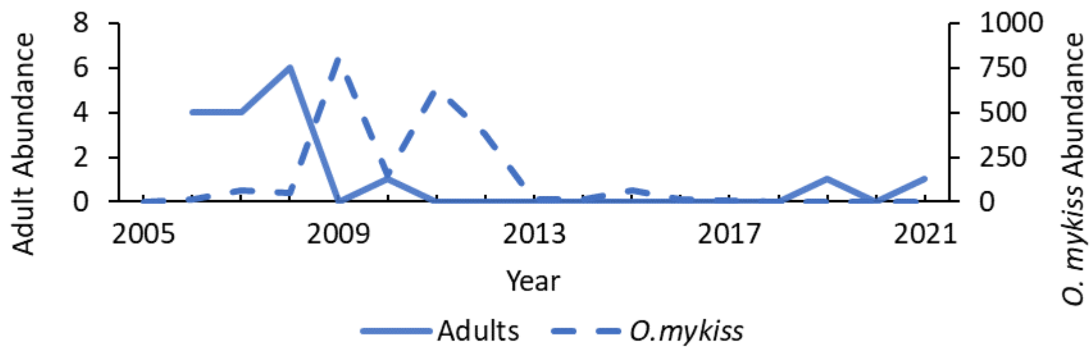
More recently, monitoring has been intermittently conducted on Carpinteria, Mission, and Arroyo Hondo in the Conception Coast BPG by the Department (Boughton et. al 2022a). Malibu, Topanga, and Arroyo Sequit creeks in the Santa Monica Mountains BPG have been actively monitored since the early 2000s (Dagit 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

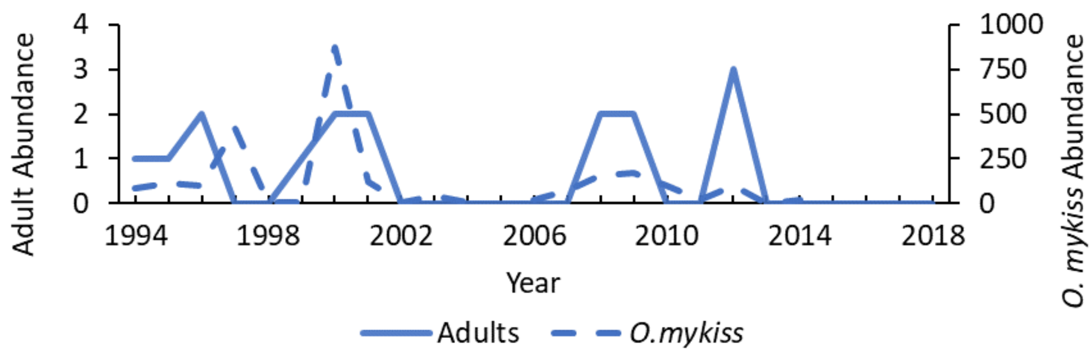


Figure 12. Adult steelhead (Adults) and other O. mykiss (O. mykiss) abundances for the Monte Arido Highlands BPG. A) Santa Ynez River; no data 2013. Biological Opinion Incidental Take provisions have been required since 2014. B) Ventura River. C) Santa Clara River. Adult abundance is on the left -axis 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

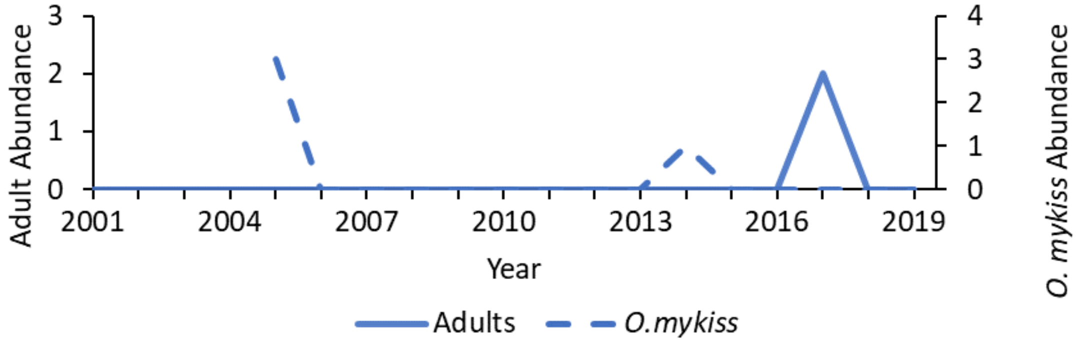
Very few monitoring activities have occurred throughout the Conception Coast BPG, and most of the work that has occurred in more recent years was conducted by the Department. We were unable to develop a full-time series of Southern SH/RT abundance for Conception Coast populations.

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).

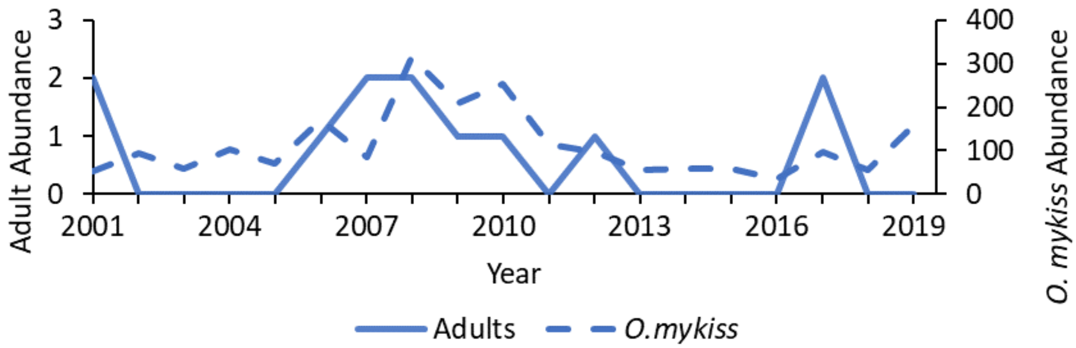
There is also limited data for *O. mykiss* in the Conception Coast BPG. No *O. mykiss* have been documented in Carpinteria Creek since 2016. In Mission Creek, no *O. mykiss* were observed from bankside surveys during the 2018-2019 spawning season (Carmody et al. 2019). In recent years, the largest number 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

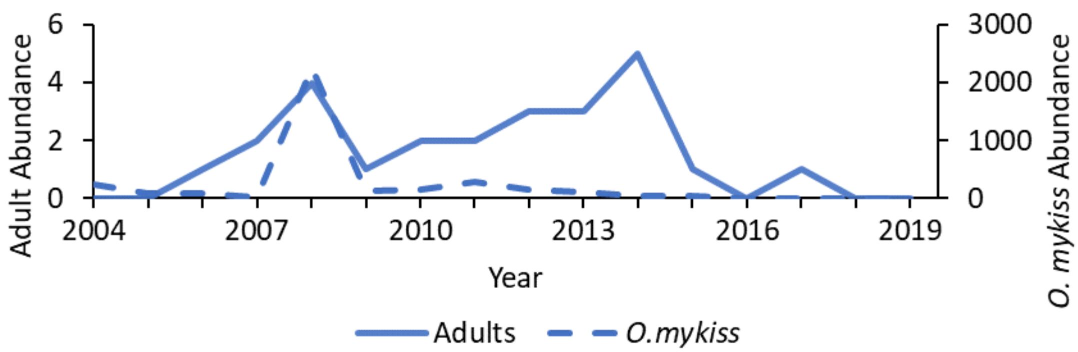


Figure 13. Adult steelhead (Adults) and other O. mykiss (O. mykiss) abundances for the Santa Monica Mountains BPG. A) Arroyo Sequit Creek. B) Topanga Creek. C) Malibu Creek. Adult abundance is indicated on the left -axis and delineated by the solid blue line and O. mykiss abundance is indicated on the 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 (Na) 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:

$$Na\ (geom) = \left(\prod Na(i)\right)^{1/n}$$

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., $ln\ (\bar{N}_a + 1)$]. Using methods from Good et al. (2005), the linear regression can be expressed as:

$$ln\ (\bar{N}_a + 1) = \beta_0 + \beta_1 X + \in$$

Where $\bar{N}_a$ is annual adult steelhead abundance, β_0 is the intercept, β_1 is the slope of the equation, and $\in$ 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<.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.

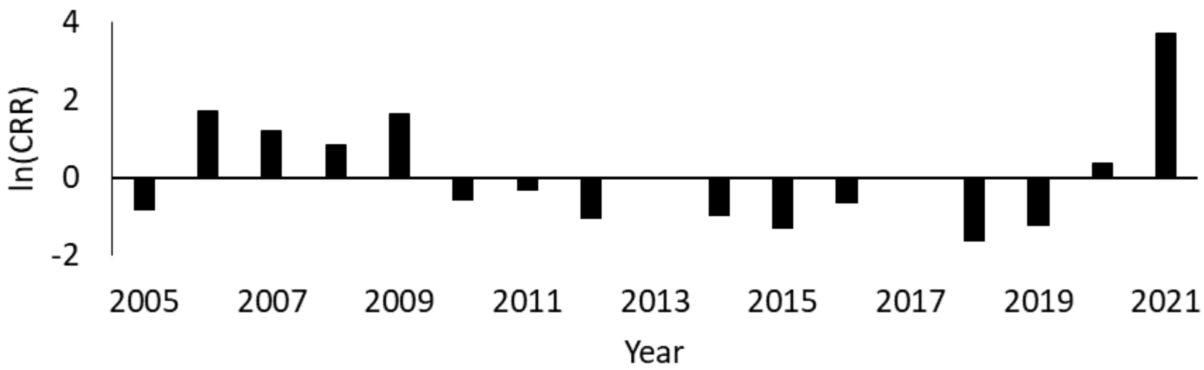
The CRR is defined as:

$$CRR = \ln (N_{t+4}/N_t)$$

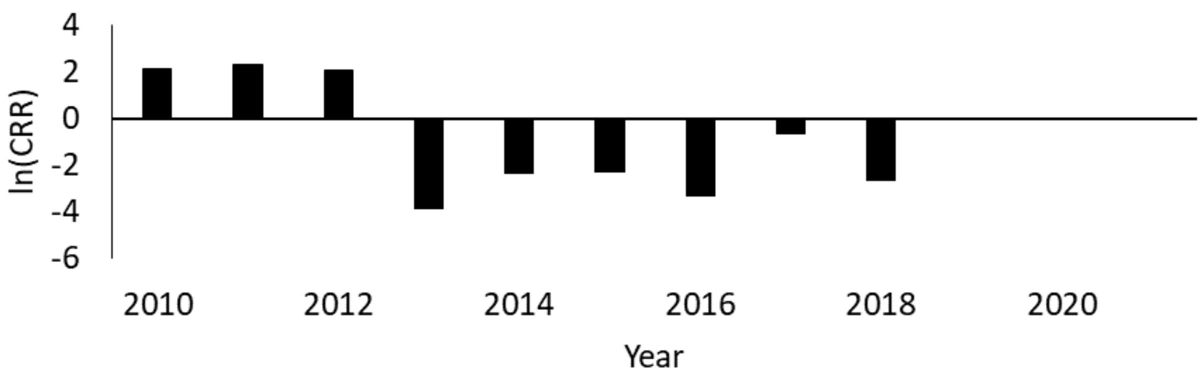
Natural log transformed CRRs greater than zero indicate that the cohort increased in size that year in relation to the brood year three years earlier, whereas a CRR less than zero indicates that the cohort decreased in size. This analysis assumes a generation time of four years, which has been determined 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.

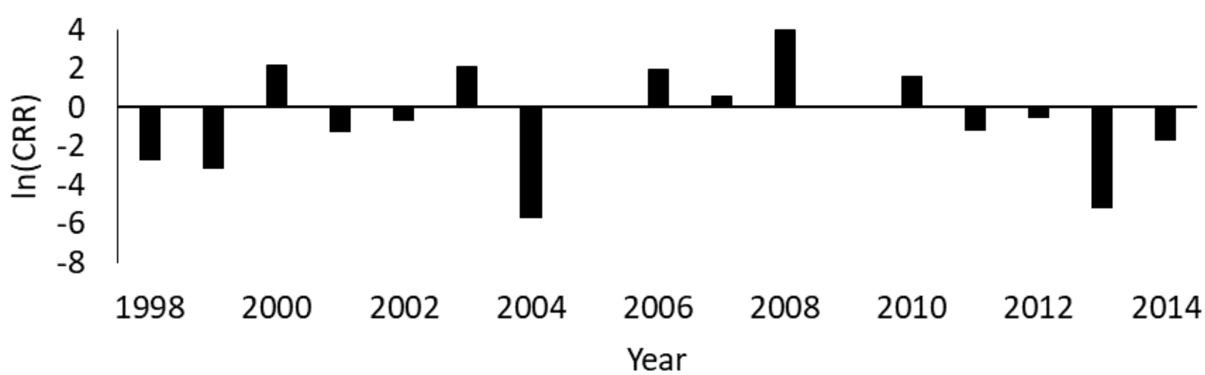
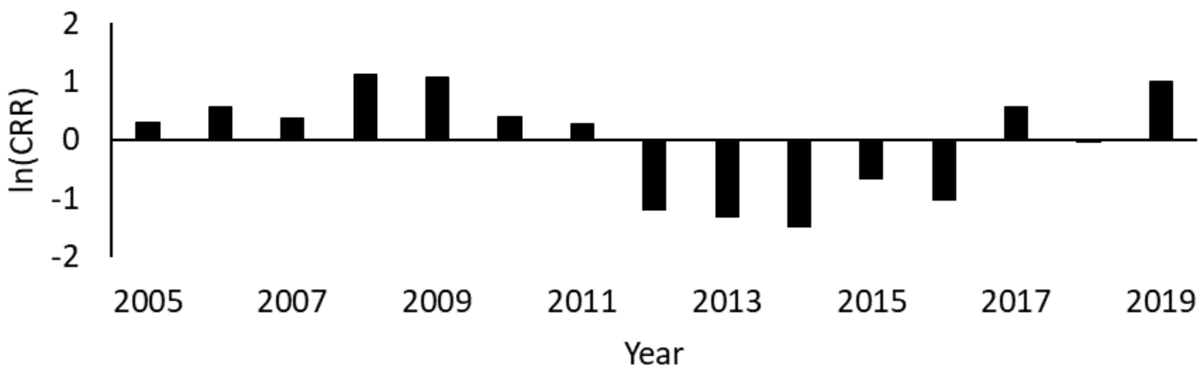


Figure 14a. Ln-Cohort Replacement Rates for O. mykiss (Other O. mykiss) populations, A) Santa Ynez River, B) Ventura River, and C) Santa Clara River; Biological Opinion Incidental Take provisions have been required since 2014. Gaps are a result of missing years of data. Note different scales on the Y-axis.

D.



E.

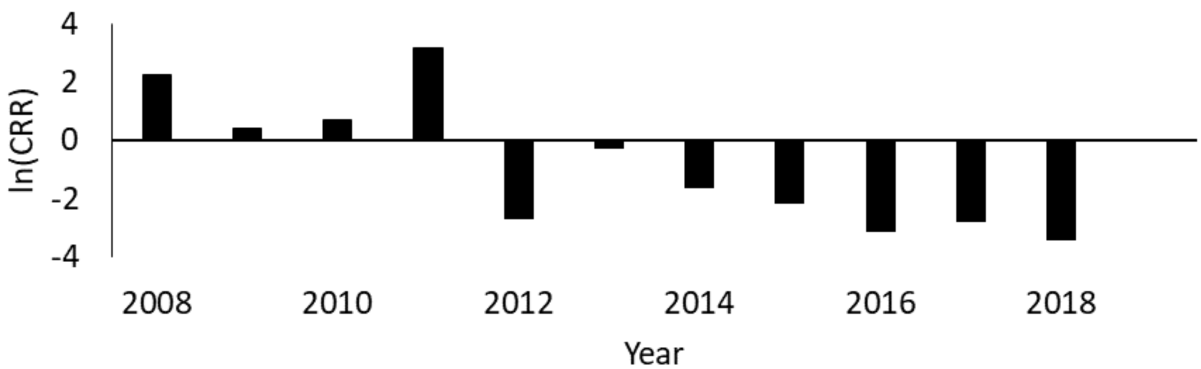


Figure 14b. Ln-Cohort Replacement Rates for O. mykiss (Other O. mykiss) populations, D) Topanga Creek, and E) Malibu Creek. Gaps are a result of missing years of data. Note different scales on the Y-axis.

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 orcutti*) 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 CO_2-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).

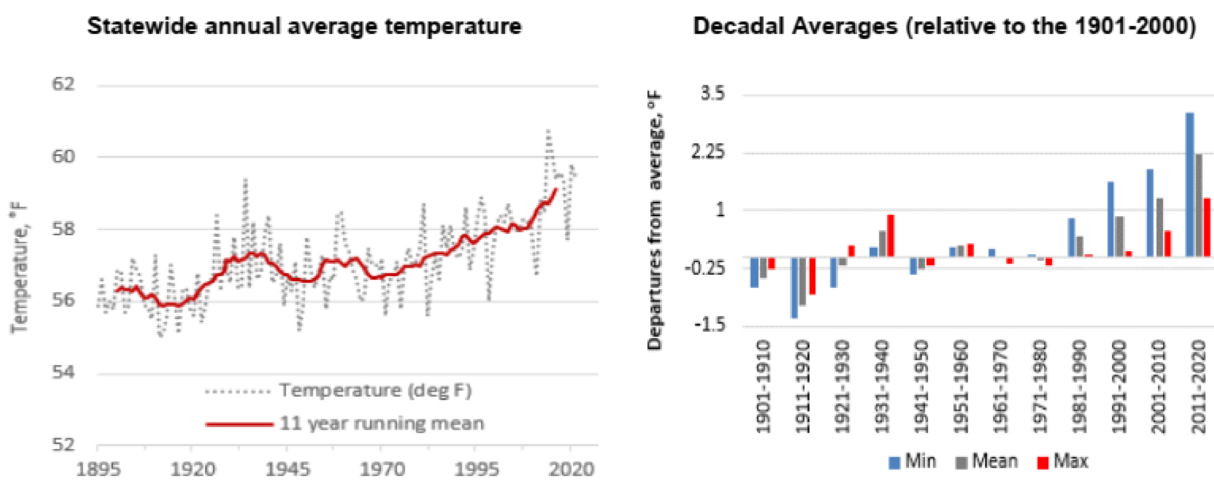


Figure 15. Temperature trend (left) and departure from average (right) graphs for California, from about 1900-2020 (source: OEHHA 2022).

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).

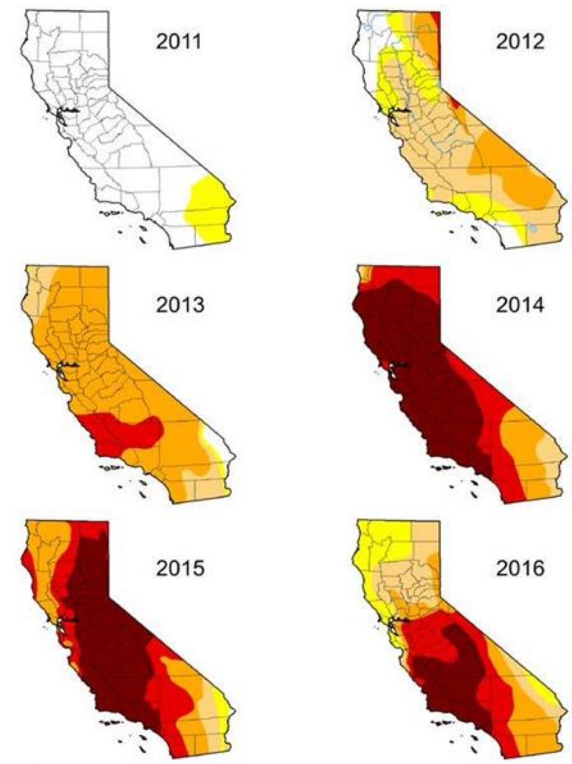


Figure 16. The distribution and progression of drought conditions in California from 2011 to 2016, depicting the level of drought at the beginning of each Water Year (October 1). White indicates no drought conditions, whereas yellow to dark red indicates increasing drought conditions, including duration 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).

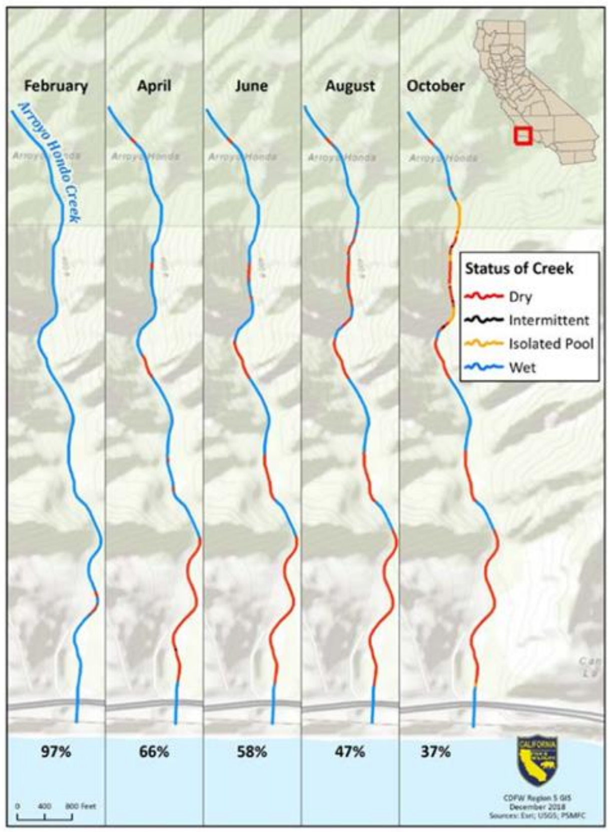


Figure 17. Example southern California stream (Arroyo Hondo Creek, Santa Barbara County), showing seasonal desiccation across 60% of its study area wetted length during February-October 2015 (source: CDFW 2018a). 2015 was a notably bad drought year in California, but the large extent of stream drying 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; Siirla-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). Largescale 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 "triploiding" 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 overspilling, 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 overspill, 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 rifles, 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. Secondarily, 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

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John O'Brien, CDFW, personal communication, 12/05/2022

Dane St. George, CDFW, personal communication, 05/24/2023

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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	0
2006	0 [d]	1	1
2007	0 [d]	2	2
2008	0 [d]	2	4
2009	0 [d]	1	1
2010	0 [d]	1	2
2011	0 [d]	0	2
2012	0 [d]	1	3
2013	0 [d]	0	3
2014	0 [d]	0	5
2015	0 [d]	0	1
2016	0 [d]	0	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 Maria River[a]	Santa Ynez River[b]	Ventura River[c]	Santa Clara River[d]	
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