STAFF SUMMARY FOR APRIL 14, 2021

7. NORTHERN CALIFORNIA SUMMER STEELHEAD (CONSENT)

Today's Item **Information ☒** **Action ☐**

Receive DFW's status review report on the petition to list northern California summer steelhead (*Onchorhynchus mykiss irideus*) as a threatened or endangered species under the California Endangered Species Act (CESA).

Summary of Previous/Future Actions

• Received petition	Sep 28, 2018
• FGC transmitted petition to DFW	Oct 8, 2018
• Published notice of receipt	Oct 26, 2018
• Public received petition and FGC approved DFW request for 30-day extension	Dec 12-13, 2018; Oceanside
• Received DFW's 90-day evaluation	Feb 6, 2019; Sacramento
• FGC determined listing may be warranted, initiating DFW's one-year status review	Jun 12 -13, 2019; Redding
• FGC approved DFW's request for a six-month extension	Aug 7-8, 2019; Sacramento
• Scheduled receipt of DFW's status review report (item continued)	Feb 10, 2021; Webinar/Teleconference
• **Today's receipt of DFW's status review report**	**Apr 14, 2021; Webinar/Teleconference**
• Determine if listing is warranted	Jun 16-17, 2021; Webinar/Teleconference

Background

In Sep 2018, FGC received a petition from Friends of the Eel River to list northern California summer steelhead as endangered under the California Endangered Species Act (CESA). At its Jun 2019 meeting, FGC determined that listing may be warranted and FGC subsequently provided notice regarding northern California summer steelhead's protected, candidate species status. The notice prompted DFW's status review of the species as required by California Fish and Game Code Section 2074.6. On Aug 7, 2019, FGC approved DFW's request for a six-month extension of time to complete its review.

DFW has completed and submitted its staus review report to FGC (exhibits 1 and 2). The report represents DFW's final written review of the status of northern California summer steelhead and delineates each of the categories of information required for a petition, evaluates the sufficiency of the available scientific information for each of the required components, and incorporates additional relevant information that DFW possessed or received during its review. Based on the information provided, possessed, or received, DFW has concluded that there is not sufficient scientific information available to justify classifying northern California summer steelhead as endangered.

California Fish and Game Code Section 2075 requires FGC to receive DFW's recommendation and consider the petitioned action at its next available meeting. At its Jun 16-17, 2021 meeting, FGC may consider the petition, DFW's written evaluation and status review report, written and oral comments received, and the remainder of the administrative record, to determine if listing is warranted. Findings would be adopted at a future meeting.

Significant Public Comments (N/A)

Recommendation

FGC staff: Receive DFW's status review report under a motion to adopt the consent calendar, accept any public comment, and schedule presentations and a potential decision for the Jun 2021 FGC meeting.

Exhibits

1. DFW memo, received Mar 11, 2021
2. DFW's status review report, received Mar 11, 2021 (Appendix E, peer review comments, is not included due to accessibility issues, but is available upon request)

Motion/Direction

Moved by __________ and seconded by __________ that the Commission adopts the FGC staff recommendation for items 3-10 on the consent calendar.

State of California
Department of Fish and Wildlife

Original on file,
received March 11, 2021

M e m o r a n d u m

Date: March 11, 2021

To: Melissa Miller-Henson
Executive Director
Fish and Game Commission

From: Charlton H. Bonham
Director

Subject: California Endangered Species Act Status Review for Northern California Summer Steelhead (*Oncorhynchus mykiss*)

The California Department of Fish and Wildlife (Department) has completed its Status Review for Northern California summer steelhead (*Oncorhynchus mykiss*) (Status Review) under the California Endangered Species Act (CESA; Fish and Game Code section 2050 et seq.). The California Fish and Game Commission (Commission) published the Notice of Candidacy Findings on June 28, 2019 directing the Department to prepare a Status Review. On July 10, 2019, in accordance with Fish and Game Code section 2074.6, the Department requested a 6-month extension to complete the Status Review. The Commission approved this request on July 11, 2019.

Pursuant to Fish and Game Code section 2074.6, this report contains the Department's review of the best scientific information available to the Department on the status of Northern California summer steelhead, and serves as the basis for the Department's recommendation to the Commission on whether to list Northern California summer steelhead as a threatened or endangered species under CESA.

If you have any questions or need additional information, please contact Mr. Jonathan Nelson, Environmental Program Manager, Fisheries Branch, at (916) 376-1641 or by email at Jonathan.Nelson@wildlife.ca.gov or Mr. Kevin Shaffer, Branch Chief, Fisheries Branch, at (916) 376-1654 or by email at Kevin.Shaffer@wildlife.ca.gov.

Attachment

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STATE OF CALIFORNIA
NATURAL RESOURCES AGENCY
DEPARTMENT OF FISH AND WILDLIFE
REPORT TO THE FISH AND GAME COMMISSION

California Endangered Species Act Status Review for Northern California Summer Steelhead (*Oncorhynchus mykiss)*

CHARLTON H. BONHAM, DIRECTOR
CALIFORNIA DEPARTMENT OF FISH AND WILDLIFE
March 11, 2021

Suggested citation:

California Department of Fish and Wildlife (CDFW). 2021. California Endangered Species Act Status Review for Northern California Summer Steelhead *(Oncorhynchus mykiss).* A Report to the California Fish and Game Commission. California Department of Fish and Wildlife, 1416 Ninth Street, Sacramento CA 95814. 188 pp., with appendices.

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ACKNOWLEDGEMENTS

Vanessa Gusman (California Department of Fish and Wildlife [CDFW or Department] Fisheries Branch) prepared this report. Ryon Kurth (CDFW Fisheries Branch) provided crucial guidance and oversight in the development of the report. Christina Parker (CDFW Fisheries Branch) contributed invaluable writing and research to sections of the report including those on genetics and genomics and habitat necessary for survival. Janet Brewster (CDFW Fisheries Branch) developed all maps included in the report. Great appreciation is extended to Shaun Thompson, Michael Sparkman, Scott Harris, and Allan Renger (CDFW Northern Region) who contributed essential data and information relevant to summer steelhead in Redwood Creek, Mad River, and Eel River watersheds. Staff from partner agencies and organizations who contributed valuable data and information on summer steelhead in Redwood Creek, Mad River, and Mattole River watersheds include David Anderson (Redwood National Park), Kyle Max (Redwood National Park), Jacob Pounds (Blue Lake Rancheria and Mad River Alliance), Patrick Righter (Green Diamond Resource Company), and Nathan Queener (Mattole Salmon Group). Thanks go out to Michael Lacy (CDFW Fisheries Branch) and Laura Patterson (CDFW Wildlife Branch) for their guidance and insight on listing criteria and status review development. Thank you to Jeff Rodzen (CDFW Fisheries Genetics) for providing expert review of the section on genetics and genomics and to John Kelly (CDFW Fisheries Branch) for thorough document review.

The Department is extremely grateful for the valuable comments provided on this report by the following peer reviewers: Alan Byrne, Kevin Goodson, Brian Marston, and Dennis Lee. Conclusions and recommendations in this report are those of the Department and do not necessarily reflect those of the reviewers.

Special thanks to Shaun Thompson for the cover photo of summer steelhead in the Middle Fork Eel River.

ACRONYMS

AHCP	Aquatic Habitat Conservation Program
ASP	Anadromous Salmonid Protection
BEUTI	Biologically Effective Upwelling Transport Index
BLM	Bureau of Land Management
CalTrout	California Trout, Inc.
CASWRCB	California State Water Resources Control Board
CATEX	Categorical Exclusion
CCE	California Current Large Marine Ecosystem
CDFA	California Department of Food and Agriculture
CDFW	California Department of Fish and Wildlife, also “Department” (previously California Department of Fish and Game [CDFG])
CEQA	California Environmental Quality Act
CESA	California Endangered Species Act
CHERT	County of Humboldt Extraction Review Team
CMP	California Monitoring Program
CRR	cohort replacement rate
CUTI	Cumulative Upwelling Transport Index
CWA	Clean Water Act
DIDSON	dual frequency identification sonar
DO	dissolved oxygen
DPS	Distinct Population Segment
EA	Environmental Assessment
EIR	Environmental Impact Report
EIS	Environmental Impact Statement
ENSO	El Niño/Southern Oscillation
ESA	Federal Endangered Species Act
ESU	Evolutionarily Significant Unit
FERC	Federal Energy Regulatory Commission
FONSI	Finding of No Significant Impact
GDRCO	Green Diamond Resource Company
GRTS	Generalized Random Tessellation Stratification
HBMWD	Humboldt Bay Municipal Water District
HCP	Habitat Conservation Plan
HGMP	Hatchery and Genetic Management Plan
LWD	large woody debris
MRH	Mad River Salmon and Steelhead Hatchery, CA
mtDNA	mitochondrial DNA
NCFR	North Coast Fisheries Restoration
NCRWQCB	North Coast Regional Water Quality Control Board
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration

NOI	Notice of Intent
NPGO	North Pacific Gyre Oscillation
NPS	National Park Service
NWFP	Northwest Forest Plan
ONI	Ocean Niño Index
PALCO	Pacific Lumber Company
PDO	Pacific Decadal Oscillation
PG&E	Pacific Gas & Electric Company
PNI	proportionate natural influence
PVP	Potter Valley Project
RNP	Redwood National Park, CA
SNP	single nucleotide polymorphism
SST	sea surface temperature
SWFSC	NMFS Southwest Fisheries Science Center
THP	Timber Harvest Plan
TMDL	Total Maximum Daily Load
USACOE	United States Army Corps of Engineers
USEPA	United States Environmental Protection Agency
USFS	United States Forest Service
USFWS	United States Fish and Wildlife Service
WCR	NMFS West Coast Region
WQC	Water Quality Certification
WSRA	Wild and Scenic Rivers Act
YOY	young-of-the-year
Y+	yearling plus

EXECUTIVE SUMMARY

Recommendation: Based on the best scientific information available, the California Department of Fish and Wildlife (Department) has determined that Northern California (NC) summer steelhead (*Oncorhynchus mykiss*) do not qualify as a separate species or subspecies for the purposes of listing under the California Endangered Species Act (CESA). Although NC summer steelhead populations have relatively low abundances as compared to their estimated historical numbers and will continue to face environmental and anthropogenic challenges, the Department has concluded that NC summer steelhead and NC winter steelhead are not fully reproductively isolated and summer and winter steelhead within the NC steelhead Distinct Population Segment (DPS) are not genetically distinct. Therefore, the Department recommends that NC summer steelhead should not be listed as a distinct species or subspecies under CESA.

Reasons for recommendation: Within the NC steelhead DPS there are two ecotypes with different run timing, NC winter steelhead and NC summer steelhead. NC winter steelhead adults enter freshwater during the winter in an already sexually mature state and spawn shortly thereafter. In contrast, NC summer steelhead enter their natal streams during the summer months as sexually immature fish and hold in deep pools until they are ready to spawn, typically in winter or early spring of the next year. Summer steelhead are usually found in reaches higher in a watershed than winter steelhead by surpassing flow-dependent barriers, which can provide some spatial reproductive isolation between the two run types. However, there is likely some degree of spatial overlap and interbreeding, especially in low water years when summer steelhead may not be able to access areas above flow-dependent barriers.

Genetic studies of steelhead population structure using neutral genetic markers have generally found that steelhead genetic population structure is significantly driven by geography. The NC steelhead DPS was defined based on available information suggesting that populations within the geographic bounds of the DPS are closely related and continue to have some amount of gene flow. Early and late migration ecotypes (e.g., summer and winter steelhead) within a watershed have also been found to be more closely related to each other than they are to their migration timing equivalents in nearby watersheds.

Recently, studies of “adaptive” genetic markers have become more prevalent. Adaptive genetic markers are associated with specific life history traits such as adult run timing in steelhead. A specific region of the steelhead genome was recently found to be highly associated with adult migration phenotypes. Variations in this small genomic region associated with the early migration phenotype (i.e., summer steelhead) have been found across many populations of steelhead on the west coast of the United States (hereafter referred to as west coast), suggesting that a single evolutionary event produced the early migration timing variants and, if lost, they would not be expected to re-evolve within a reasonable time frame.

Although there is significant variation in this small genomic region between summer and winter steelhead, very little variation is found between their genomes, suggesting that geography is still the main influence on population genetic structure of west coast steelhead. It remains unchanged that summer and winter steelhead within the same watershed are the most closely related to each other than they are to their migration timing equivalents in other watersheds. In addition, interbreeding between the two runs is evident from the presence of heterozygotes; steelhead that possess both the early and late genetic variants and typically express an intermediate run timing. In fact, full siblings can possess different migration timing genotypes and display alternate run timing phenotypes. Listing only one run timing ecotype under CESA, i.e., summer steelhead, when family units could be comprised of summer, winter, and intermediate run timing phenotypes, poses significant difficulties for management. Additionally, defining conservation units based on a small genomic region associated with a specific life history trait does not reflect the patterns of genetic diversity found by looking across the whole steelhead genome. Although the sequences found in the genomic region associated with adult migration timing appear to have only evolved once, they are highly conserved across multiple west coast salmon and steelhead populations. Thus, if the early migration variants were lost from one population, they could potentially be naturally reintroduced from another nearby source population. Early migration variants may also be preserved to some degree by resident forms of *O. mykiss* (Rainbow Trout).

There are still uncertainties surrounding phenotypic expression of run timing in steelhead and conservation units should continue to be defined by patterns of genetic diversity across the genome instead of by variation within small genomic regions associated with specific life history characteristics. Pacific salmonid management units have generally been defined using patterns of genetic diversity based on neutral genetic markers, which show population structure to be highly influenced by geography. At this time, based on the best available scientific information, the Department does not consider NC summer steelhead to be a distinct subspecies eligible for listing under CESA.

Range and distribution: NC summer steelhead are currently found in select Northern California streams including Redwood Creek (Humboldt County), the Mad River, select parts of the Eel River system including the Middle Fork Eel River and Van Duzen River, and the Mattole River. The Middle Fork Eel River NC summer steelhead population is the most robust and accounts for a significant portion of the total NC summer steelhead in all presently occupied streams. Historically, NC summer steelhead occupied additional tributaries of the Eel River drainage including the North Fork Eel River, South Fork Eel River, Larabee Creek, and the upper Eel River basin. The construction of Scott Dam, which completely blocked upstream habitat to anadromous fish, likely led to the extirpation of summer steelhead populations in the upper Eel River. Robert W. Matthews Dam also prevents steelhead from accessing upstream habitat in the Mad River but to a much lesser extent.

Status and Trend: Five NC summer steelhead streams are surveyed annually for abundance of adults holding in pools over summer. Some NC summer steelhead populations are at critically low abundances and none of the populations are meeting current viability targets. The most robust population is in the Middle Fork Eel River, although this population has declined in recent years. Current population levels are likely much lower than what has been suggested by historical accounts. In addition, multiple NC summer steelhead populations have already been functionally extirpated, reducing distribution and overall abundance of the ecotype. Population losses and declines can increase the risk of depensation, genetic bottlenecks, and loss of diversity. Small populations can be at a higher risk of extinction than large populations because they may not respond as well to demographic and environmental stochasticity or catastrophic events. Long-term and recent trend analyses generally showed slightly negative trends, though they were mostly statistically non-significant. Trend analyses were not completed for some populations due to inconsistent or missing data.

Major listing factors: NC summer steelhead are at risk for continued habitat loss and modification resulting from natural conditions and anthropogenic activities. Compounding effects of a long legacy of human activities in NC summer steelhead watersheds and natural events have been exacerbated by unstable geology of the region. Timber harvest, gravel mining, livestock grazing, and agriculture continue to be the primary land uses in NC summer steelhead watersheds and contribute to sediment loading, turbidity, decreased flow, and increased water temperatures among other impacts on the stream environment. Large dam construction on the upper Eel River has eliminated access to hundreds of miles of potential steelhead habitat and has altered natural flow regimes and sediment transport. Construction of Scott Dam was likely the main reason NC summer steelhead populations were lost from the upper Eel River. Another more recent land use threat is the growing cannabis industry including both legal and illicit operations, which can destroy habitat and remove stream surface water and groundwater. Existing threats to NC summer steelhead will likely be amplified by climate change. Impacts of climate change on the north coast of California may include rising water temperatures, intensified flooding, more frequent and persistent drought conditions, lower summer baseflows, altered hydrography especially in watersheds impacted by snowmelt and large-scale historical timber harvest, ocean acidification, and sea level rise. NC summer steelhead are likely to endure accelerated habitat loss and degradation as climate change exacerbates the cumulative effects of natural and anthropogenic factors.

The NC summer steelhead ecotype is an important diversity component of anadromous *O. mykiss* and encompasses genetic variation that will help sustain NC steelhead through short- and long-term changes in the environment. Should the California Fish and Game Commission (Commission) decide not to list NC summer steelhead, the Department will continue to support and participate in programs intended to benefit NC summer steelhead. These efforts include improving fish passage, minimizing impacts of gravel extraction and other land uses, restoration and enhancement of essential habitat, ongoing research and monitoring, and interagency coordination on the aforementioned activities. The Department also recommends a suite of management measures to help preserve NC summer steelhead regardless of their CESA-listing status. These management recommendations address habitat improvement and expansion, comprehensive population monitoring, assessment of the effects of the Potter Valley Project on the upper Eel River, continued research on the genetic mechanisms involved in adult migration timing, evaluation of the effects of recreational fishing and predation on NC summer steelhead, and increasing law enforcement in remote areas to minimize poaching and illegal water diversions. Implementing these management measures will help maintain the NC summer steelhead ecotype as a crucial component of the NC steelhead DPS.

1. INTRODUCTION

1.1 Petition Evaluation Process

The Friends of the Eel River (Petitioner) submitted a petition (Petition) on September 28, 2018, to the Commission to list the Northern California (NC) summer steelhead (*Oncorhynchus mykiss irideus*[1]) as endangered pursuant to the California Endangered Species Act (CESA), Fish and Game Code Section 2050 *et seq*.

The California Fish and Game Commission (Commission) referred the Petition to the California Department of Fish and Wildlife (Department) on October 8, 2018, in accordance with Fish and Game Code § 2073 (Cal. Reg. Notice Register 2017, No. 13-Z, 479.). On January 24, 2019, the Department produced an evaluation report of the Petition (Petition Evaluation) evaluating the scientific information presented in the Petition as well as other relevant information the Department possessed at the time of review. The Department did not receive any information from the public during the Petition Evaluation period pursuant to Fish and Game Code § 2073.4. Pursuant to Fish and Game Code § 2072.3 and § 670.1, subdivision (d)(1), of Title 14 of the California Code of Regulations, a petition to list or delist a species under CESA must include sufficient scientific information regarding each of the following components to indicate that the petitioned action may be warranted:

- population trend,
- range,
- distribution,
- abundance,
- life history,
- kind of habitat necessary for survival,
- factors affecting ability to survive and reproduce,
- degree and immediacy of threat,
- impacts of existing management,
- suggestions for future management,
- availability and sources of information, and
- a detailed distribution map.

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[1] The subspecies name *Onchorhynchus mykiss irideus* was used in the Petition and has been used by some authors in the scientific literature to indicate "coastal Rainbow Trout" or steelhead. This taxonomy is not recognized by the American Fisheries Society (AFS), and common and scientific names used hereafter in this report conform to AFS guidelines. Common names (e.g., Rainbow Trout) are capitalized, whereas runs and life history strategies (e.g., steelhead) are not. Please see *Section 2.3 Taxonomy and Systematics* for further discussion.

The Department submitted its Petition Evaluation to the Commission on January 24, 2019 to assist the Commission in making its determination as to whether the petitioned action may be warranted based on the sufficiency of scientific information. (Fish & Game Code, §§ 2073.5 & 2074.2; California Code of Regulations, Title 14, § 670.1, subdivisions (d) & (e)). Based on the Petition and other information available to the Department relating to each of the relevant categories, the Department recommended to the Commission that the Petition be accepted. The Commission has not previously received a petition to list NC steelhead (summer or winter) under CESA. Following receipt of the Department's Petition Evaluation, at its scheduled public meeting on June 12, 2019, the Commission considered the Petition, the Department's Petition Evaluation and recommendation, and comments received. The Commission found that sufficient information existed to indicate the petitioned action may be warranted and accepted the Petition for consideration. Upon publication of the Commission's notice of its findings, NC summer steelhead was designated a candidate species on June 28, 2019.

1.2 Status Review Overview

The Commission's action designating NC summer steelhead as a candidate species triggered the Department's process for conducting a status review to inform the Commission's decision on whether listing the species is warranted. At its scheduled public meeting on July 11, 2019 in Sacramento, the Commission granted the Department a six-month extension to complete the status review and facilitate external peer review.

This status review report is not intended to be an exhaustive review of all published scientific literature relevant to NC summer steelhead; rather, it is intended to summarize the key points from the best scientific information available relevant to the status of the species. This final report, based upon the best scientific information available to the Department, is informed by independent peer review of a draft report by scientists with expertise relevant to NC summer steelhead. This review is intended to provide the Commission with the most current information on NC summer steelhead and to serve as the basis for the Department's recommendation to the Commission on whether the petitioned action is warranted. The status review report also identifies habitat that may be essential to continued existence of the species and provides management recommendations for recovery of the species (Fish & G. Code, § 2074.6). Receipt of this report is to be placed on the agenda for the next available meeting of the Commission after delivery. At that time, the report will be made available to the public for a 30-day public comment period prior to the Commission taking any action on the Petition.

Comments from external peer reviewers are contained in Appendix E. The Department received ten public comments from its 30-day public solicitation for information beginning August 22, 2019. Public comments are included in Appendix F.

1.3 Previous Federal Listing Actions

The Northern California steelhead Evolutionarily Significant Unit (ESU) was originally proposed for listing as threatened under the federal Endangered Species Act (ESA) by the National Marine Fisheries Service (NMFS) in 1996 (Endangered and Threatened Species: Proposed Endangered Status for Five ESUs of Steelhead and Proposed Threatened Status for Five ESUs of Steelhead in Washington, Oregon, Idaho, and California, 1996). NMFS deferred the final determination for NC steelhead until March 1998, when NMFS stated that the NC steelhead ESU did not warrant listing under the federal ESA. In 2000, NMFS proposed to list the NC steelhead ESU as a threatened species. The listing included only naturally spawned steelhead (anadromous form of *O. mykiss*) and their progeny residing below impassable barriers to migration for both summer and winter ecotypes. NMFS did not designate NC summer steelhead as a separate ESU mainly due to the fact that the most recent genetic data reinforced previous conclusions that sympatric summer and winter steelhead typically are more genetically similar to each other than they are to their run timing equivalents in other geographic areas (Busby et al. 1996; Endangered and Threatened Species: Threatened Status for One Steelhead Evolutionarily Significant Unit (ESU) in California, 2000). In 2006, NMFS re-classified the listing of the NC steelhead ESU to a Distinct Population Segment (DPS) and reaffirmed the listing status as threatened (NMFS 2006)[2].

1.4 Concurrent Federal Petition

NMFS received a petition from the Petitioner on November 15, 2018, requesting that NC summer steelhead be considered as a new DPS distinct from the NC Steelhead DPS and listed as endangered under the ESA (Pearse et al. 2019). NMFS determined that the petitioned action may be warranted on April 22, 2019, resulting in a “positive 90-day finding” (Endangered and Threatened Wildlife; 90-Day Finding on a Petition to List Summer-Run Steelhead in Northern California as Threatened or Endangered Under the Endangered Species Act, 2019). NMFS West Coast Region (WCR) requested that the Southwest Fisheries Science Center (SWFSC) form a panel of experts to evaluate the DPS Policy (Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the Endangered Species Act, 1996) with the best available science to determine the validity of the Petitioner’s claims and report their findings to WCR (Pearse et al. 2019). The SWFSC expert panel concluded that the NC steelhead DPS should remain as a unit including both summer and winter steelhead within its bounds, stating that “The data presented by the petition did not fundamentally change our understanding of the importance of preserving the evolutionary processes that connect populations, allowing them to maintain genetic diversity at both neutral and adaptive loci and preserve their evolutionary

[2] See *Section 2.5 The ESU/DPS Concept in Management of Pacific Salmonids* for clarification.

potential" (Pearse et al. 2019). Thus, the ESA listing of the NC steelhead DPS as threatened remains intact and includes both summer and winter ecotypes.

2. BIOLOGY AND ECOLOGY

2.1 Species Characteristics

Steelhead are the anadromous form of *Oncorhynchus mykiss* and are found in watersheds from Alaska to Southern California. They are the most widely present species of all the Pacific salmonids, occupying nearly all ocean-connected streams throughout their range. Steelhead are named for the metallic appearance of their heads and possess a streamlined, hydrodynamic body shape. In the ocean, steelhead have a blue dorsal side and silver coloration over the rest of their body (Fry 1973; Moyle 2002). Black spots typically cover their dorsal, adipose, and caudal fins, as well as their head and back (Fry 1973). As steelhead enter freshwater, their silver sheen fades and they develop a pink or red lateral band and a pink hue in the operculum, and the blue coloration on their back transitions to an olive green or brown (Barnhart 1986). These characteristics are very similar to those of the freshwater resident form of *O. mykiss*, Rainbow Trout (Fry 1973), thus it can be difficult to tell the anadromous and resident forms apart based on outward appearance. Steelhead, however, are often larger than Rainbow Trout since they spend time feeding and growing in the ocean (NWF 2020; USFWS 2020).

Onchorhynchus 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).

Different subgroups or populations of steelhead can exhibit variation in growth rate, size, and body shape depending on their life history and habitat. 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 with 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). Additionally, adult steelhead in the Northern California DPS were generally the largest of the steelhead in the four DPS units studied (Northern California, Central California Coast, Klamath Mountain Province, and Central Valley). At maturity, California summer steelhead typically have a fork length of 60 – 80 cm, though summer steelhead in the Eel River watershed exhibit a wider range of sizes at maturity and can be anywhere between 48 – 84 cm (Moyle et al. 1995).

The steelhead sexual maturation process includes development of secondary sex characteristics such as bright coloration and sexual dimorphisms including the development of a hooked snout in males known as a kype. According to observations of steelhead on the Capilano River in British Columbia (Smith 1960), there are some visible differences between the spawning forms of summer and winter steelhead. Due to the earlier river entry and over-summer holding of summer steelhead, these fish, especially the males, exhibited more pronounced secondary sex characteristics than spawning winter steelhead, including brighter coloration, bifurcated and flattened tips of the gill rakers, and a more well-defined kype. Winter steelhead were not observed to undergo as drastic of a morphological transformation due to their shorter inhabitance of the freshwater environment. They showed very little red coloring and no flattening or bifurcation of the gill raker tips (Smith 1960). These may be regionally specific morphological distinctions as they have not been observed in Middle Fork Eel River steelhead. In the Middle Fork Eel River, summer steelhead found holding between June and early winter have a healthy, thick appearance due to the high fat reserves they possess when entering freshwater. Their high fat content is metabolized throughout their holding period as steelhead generally do not feed while in freshwater. Winter steelhead that are encountered during this time as spawned out kelts will look emaciated and snake-like (S. Thompson, CDFW, pers. comm., July 8, 2020). Winter steelhead, upon initial entry of the Middle Fork Eel River, have been observed to be smaller and look brighter than the summer steelhead that have been in the river system for months (Clemento 2006). Secondary sex characteristics are typically reabsorbed once spawning is complete, though jaw shape may never fully revert (Shapovalov and Taft 1954).

Juvenile steelhead have similar coloration to resident adults but also exhibit 5 – 13 oval parr marks along the lateral line on both sides of the body (Moyle 2002). These parr marks are widely spaced with the marks themselves being narrower than the spaces between them. Anywhere between 5 and 10 dark spots line the back from the head to the dorsal fin. There are usually very few or no marks on the caudal fin. Tips of the dorsal and anal fins are white to orange. As parr transition into smolts, they lose their parr marks and develop silver coloration. Once they migrate to the ocean, they will reside in the saltwater environment for 1 – 4 years feeding and growing quickly (Moyle 2002).

2.2 Range

Oncorhynchus mykiss are native to the Pacific coast and are found spawning in streams from the Kuskokwim River in Alaska down to Baja California on the eastern pacific shore, and from Japan and South Korea north to the Russian Kamchatka Peninsula on the western pacific shore (Moyle 2002). They are present throughout the Northern Pacific Ocean in their ocean phase. Due to many decades of planting efforts, resident Rainbow Trout have become ubiquitous in nearly all California streams with suitable habitat including areas above impassible natural barriers. They have the most extensive distribution of any fish species in California and have also been introduced on every continent. Coastal steelhead within the state historically occupied all perennial coastal streams from the Oregon/California border down to San Diego

County (Moyle 2002). Some steelhead use intermittent streams as well (M. Sparkman, CDFW, pers. comm., July 6, 2020). 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 historically supported runs of steelhead (Moyle 2002).

NC summer steelhead currently occupy fluvial habitat from Redwood Creek in northern Humboldt County down to the Mattole River, though they are only found in a handful of watersheds within this range. NC summer steelhead are included in two NMFS-defined geographic diversity strata: Northern Coastal and North Mountain Interior. These two diversity strata encompass 10 historically functionally independent summer steelhead populations (NMFS 2016b). The current NC summer steelhead range encompasses Redwood Creek, the Mad River, and the Mattole River as well as sectors of the Eel River watershed including the Middle Fork Eel River and the Van Duzen River (Moyle et al. 2017). Populations in the Eel River watershed thought to be extirpated or functionally extirpated include those historically in the North Fork Eel, South Fork Eel, upper middle and upper mainstem Eel River. It is speculated that summer steelhead also may have historically occupied Larabee Creek (NMFS 2016b; Moyle et al. 2017). The Middle Fork Eel River summer steelhead population currently accounts for up to 50% of all summer steelhead in California (Moyle 2002). Other runs of summer steelhead in California are found in the Trinity River watershed (North Fork Trinity, New, South Fork Trinity rivers, and Canyon Creek), the Klamath River system (Dillon, Elk, Indian, Red Cap, Bluff, and Clear creeks), the Salmon River (including Wooley Creek), and the Smith River (Moyle et al. 1995). These populations are part of the Klamath Mountain Province steelhead DPS. Historically, summer steelhead were likely present throughout the Sacramento River basin, as well, but currently only winter steelhead are found in the Central Valley as a result of the construction of large dams beginning in the 1930's, which eliminated upstream habitat that supported the summer run (McEwan 2001).

There are two major dams that impede NC summer steelhead access to upstream spawning habitat. The more northern of these dams is Robert W. Matthews Dam on the Mad River, which created Ruth Lake. Although Matthews Dam blocks off about 30% of the mainstem river, it is estimated to eliminate only a couple miles of potential steelhead habitat (NMFS 2016b). In addition to this manmade barrier, there is a natural flow-dependent barrier about 30 miles downstream of the dam, which decreases the amount of mainstem habitat currently accessible to summer steelhead (below Matthews Dam) by about 39% during lower flow years (Spence et al. 2008; Pounds et al. 2015). The second major dam within NC summer steelhead range is Scott Dam on the upper mainstem Eel River, which prevents steelhead from accessing up to 288 miles of potential habitat upstream (Cooper 2020).

2.3 Taxonomy and Systematics

Steelhead and Rainbow Trout are members of the bony fishes 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 Thomas et al. (1986 in Quinn 2018) analyzed relatedness by looking at differences in mitochondrial DNA (mtDNA), which showed that Coho and Chinook salmon were related more closely to steelhead than they were to the other three genera of salmon. 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 trout 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 trout with regards to genetics versus morphology and behavior. Stearley and Smith (1993) and Esteve and McLennan (2007) found that the idea of monophyly (descending from a common ancestor) of these two trout species is not supported by morphological or behavioral traits, respectively, even though mitochondrial DNA suggests otherwise. Esteve and McLennan (2007) attribute this to hybridization events that have led to a high rate of genetic introgression between the two species (Chevassus 1979 as cited in Esteve and McLennan 2007). 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; Quinn 2018, Figure 2.1).

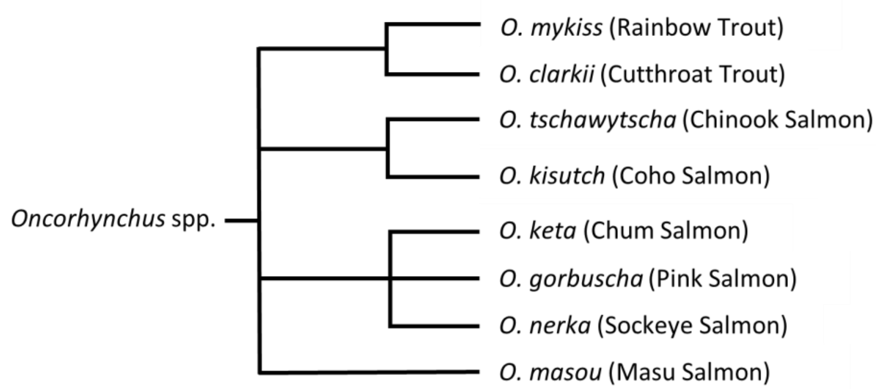


Figure 2.1. *Consensus relationships of Oncorhynchus species from morphological, allozyme, ribosomal RNA, mitochondrial DNA, and short interspersed repetitive elements data across multiple studies. Adapted from Figure 1 in Kitano et al. (1997).*

There are numerous non-taxonomic units (below the species level), or nontaxa, of Rainbow Trout in California. The most commonly recognized nontaxa are defined by their migration types (i.e., anadromous or resident) or their seasonal run timing (i.e., summer or winter), though *O. mykiss* cannot be differentiated by seasonal run timing or anadromy through classical taxonomy (Behnke 1972; Wilson et al. 1985). Salmonid nontaxa have been managed as ESUs or DPSs as described under the ESA (see section below). NC summer steelhead, as referenced in the Petition, previously have not been identified as a separate DPS from winter steelhead within the NC steelhead DPS, though the potential for this distinction may exist given the unique run timing and possible evidence of genetic differentiation (see *Section 2.6 Genetics and Genomics*).

2.4 Life History and Unique Characteristics

Of all Pacific salmonids, *O. mykiss* have the most diverse range of life history strategies. *Oncorhynchus mykiss* can either be anadromous (migrating out to sea for a portion of their life and then returning to freshwater) or resident (perpetually remaining in freshwater). These two forms, steelhead and Rainbow Trout, respectively, coexist within populations (Moyle et al. 2008) and offspring from both forms have the potential to be anadromous or resident (Busby et al. 1996). Additionally, unlike other Pacific salmonids, which are semelparous and perish almost immediately after spawning, steelhead can be iteroparous with the potential to spawn up to four times, though usually no more than once or twice (Shapovalov and Taft 1954; Everest 1973). 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 short, coastal streams. (Marston et al. 2012).

As noted previously, steelhead exhibit two seasonal run types; winter, also called ocean-maturing or mature migrating, and summer, also called stream-maturing or premature migrating. The names of these two run types are reflective of the time of year the fish reenter the estuaries and rivers as adults to reproduce (Busby et al. 1996; Moyle 2002). NC winter steelhead return to the rivers during the winter months, between December and March, with already developed gonads and spawn almost immediately (Smith 1960; McEwan and Jackson 1996). In contrast, NC summer steelhead normally return during the summer, between May and October, while still sexually immature, and hold over in pools for nine months to a year prior to spawning (Shapovalov and Taft 1954; Smith 1960; Everest 1973; Busby et al. 1996). Exact run timing can differ somewhat between streams. For example, in British Columbia, summer steelhead runs on the Coquihalla and Silver rivers migrate primarily in May and June, whereas on the Dean and Brem rivers they return mostly in July and August (Smith 1960). On the Middle Fork Eel and Van Duzen rivers, runs of summer steelhead typically return between April and June and wait in headwater areas to commence spawning (Puckett 1975). Via internal Department correspondence, Leo Shapovalov (CDFG) noted that summer steelhead in the Middle Fork Eel River probably reach the mouth in April or May to begin their upstream migration (CDFG 1953). On the Mad River, summer steelhead enter fresh water between April and July and on the Mattole River they return between March and June (Moyle et al. 2017).

Some researchers also consider there to be an intermediate “fall-run” ecotype present in some Northern California steelhead streams. These fish enter freshwater from late summer through early fall and have been seen in the Klamath, Mad, and Eel rivers (McEwan and Jackson 1996); however, there is a lack of consensus within the scientific community on whether fall-run is its own separate run of steelhead. Everest (1973) considered the “late run” in the Rogue River to be part of the summer steelhead run; included in the vast range of return timing that extends from May through October. No spatial or temporal isolation was seen between the two groups on the spawning grounds, indicating a shared gene pool. Only streams with summer steelhead runs have been observed to support a late summer and early fall recreational steelhead fishery (e.g., Klamath, Mad, and Eel rivers) (Roelofs 1983).

There is some uncertainty with respect to the degree of geographic and temporal separation of summer and winter steelhead on the spawning grounds. Although there is limited information on summer steelhead spawning, some sources have indicated that summer steelhead tend to spawn earlier than winter steelhead, at least in terms of their peak spawn timing. On the Rogue River in Oregon, it was observed that although there was temporal overlap in spawning periods, the peak of winter steelhead spawning was about 60 days later than that of summer steelhead (Everest 1973). The peak of summer steelhead spawning on the Rogue River likely occurred in January, with spawning complete by April. Winter steelhead spawning seemed to peak in early April when summer steelhead fry were already beginning to emerge. On the Middle Fork Eel River, summer steelhead likely continue to migrate upstream during the winter, spawning in the late winter or early spring (S. Thompson, CDFW, pers. comm., April 8, 2020). Summer steelhead redd construction has been observed to peak from late February to mid-March in the Middle Fork Eel River, which coincides with the peak spawning period for winter steelhead in other coastal Mendocino streams (S. Thompson, CDFW, pers. comm., July 7, 2020). Although there are no observations of winter steelhead spawning in the Middle Fork Eel River for comparison, it is likely their spawn timing is similar to that of winter steelhead in Mendocino Coast streams (S. Thompson, CDFW, pers. comm., July 7, 2020). Jones (1980) notes that Middle Fork Eel River summer steelhead have been observed spawning from late December through April. On the Mad River, anglers routinely catch kelts in November that are thought to be summer steelhead, suggesting that Mad River summer steelhead may begin spawning rather early in the winter (M. Sparkman, CDFW, pers. comm., July 8, 2020).

Summer and winter steelhead have been observed to use different stream reaches within some watersheds as a result of variation in accessibility and flows. Summer steelhead can navigate some natural barriers at moderate flows, allowing them to access areas higher in the system than winter steelhead during their return, which isolates the two runs to some extent (Puckett 1975). Withler (1966) and Smith (1969) found that summer steelhead in British Columbia occupied spawning habitat upstream of naturally formed seasonal barriers that became inaccessible to winter steelhead following a decrease in stream flows during summer. Similar patterns have been observed in the Middle Fork Eel and Van Duzen rivers where summer steelhead spawning activity is often isolated by natural barriers that prevent passage of winter steelhead when they enter the rivers (Puckett 1975; Cramer et al. 1995). These barriers include multiple areas of roughs[3] in the upper Middle Fork Eel River (CDFG 1966 – 2018) and a natural rock barrier on the Van Duzen River at Salmon Hole (Puckett 1975), which summer steelhead can pass through if flows are adequate (see Figure 3.6 in *Section 3.3.3.5 Van Duzen River*).

Kannry et al. (2020) posited that substantial spatial separation exists between summer and winter run types, with summer steelhead primarily using habitat above seasonal, flow-dependent barriers. This determination was made through genetic analysis of juvenile steelhead above and below flow-dependent barriers on the Middle Fork Eel and Van Duzen rivers at the GREB1L genomic region that has been found to be associated with steelhead migration timing (see *Section 2.6.5 Role of GREB1L Genomic Region in Migration Timing*). Kannry et al. (2020) found a notable separation of run timing genotypes at the GREB1L region upstream and downstream of flow-dependent barriers in these rivers.

Although summer steelhead may often migrate to upstream areas in a watershed that winter steelhead cannot due to flow-dependent barriers, on the Middle Fork Eel River, summer and winter steelhead have been seen using the same holding and rearing areas (Jones and Ekman 1980). It is also thought that in the Mad River, due to the fact that it is a rainfall-driven stream

[3] Roughs refer to reaches within a river that are dominated by large boulders 10 feet or more in diameter with generally steeper gradients than other sections of the river (Hutchins 1980).

rather than a snowmelt-driven stream, peak flows occur before the summer steelhead run, thus, summer steelhead are not necessarily able to access areas higher in the system than those accessible to winter steelhead (M. Sparkman, CDFW, pers. comm., February 7, 2020).

Some rivers experience a “half-pounder” run of immature fish that have spent only a few months at sea before returning to freshwater. Most half-pounders do not spawn upon their first immigration to freshwater, but instead make their way back to the ocean and return to freshwater again in a subsequent year to reproduce (Everest 1973; Hodge et al. 2014). They also continue to actively feed during their freshwater inhabitance, unlike mature adult steelhead, which mostly cease food consumption upon river entry (Barnhart 1986). The half-pounder life history strategy has been observed within a small geographic area of southern Oregon and Northern California, including the Rogue, Klamath, Mad, and Eel rivers, and Redwood Creek, and has been observed in both winter and summer steelhead runs (Everest 1973; Barnhart 1986; Busby et al. 1996; Sparkman et al. 2020). However, it has been suggested that the half-pounders found in the Mad and Eel Rivers are non-spawning “wanderers” that ultimately return to either the Rogue or Klamath rivers to spawn once mature (Knutson 1975; Cramer et al. 1995). Half-pounders have been observed to stray between basins 87% more than adult steelhead (Satterthwaite 1988 as cited in Hodge et al. 2014). Lee (2015) also suggests that half-pounders observed in the Mad River could be strays from other systems or possibly resident Rainbow Trout. Given the substantive nature of the half-pounder run on the Mad River, it seems unlikely to be a result of straying (M. Sparkman, CDFW, pers. comm., July 7, 2020). However, the origin of these half-pounders has not been confirmed. Regardless of their origin, current angler data from the Eel River suggest that half-pounders are much less abundant than they were in the late 1800s when anglers would catch dozens per day (Lee 2015). Half-pounders in the Eel River system are likely only found in the lower mainstem. There is no evidence of half-pounders in the Middle Fork Eel or Van Duzen rivers as these streams are likely too difficult for half-pounders to reach during their upstream migration in late-summer to early fall (S. Thompson, CDFW, pers. comm., July 7, 2020).

When female steelhead are ready to spawn, they will select a suitable spawning site and then dig their nest, or redd, in which they deposit their eggs to incubate. During redd construction, the female may be courted by multiple males, though following completion of the redd, the most dominant males position themselves alongside the female, depositing eggs and milt almost simultaneously (Quinn 2018). Immediately following fertilization, females cover their eggs with gravel (Barnhart 1986). Females dig multiple pits where they deposit a portion of eggs into each pocket until all the eggs are expelled (Shapovalov and Taft 1954; Quinn 2018).

Anadromous steelhead are often accompanied by resident male Rainbow Trout during spawning as they attempt to participate in the spawning activities by swimming in and out of the redd (Shapovalov and Taft 1954). These fish are sometimes referred to as “egg-eaters” although it is thought that the main purpose of their presence is to contribute to spawning rather than prey upon the newly laid eggs (Shapovalov and Taft 1954). Known resident populations in the NC steelhead DPS area have mostly been documented above Scott Dam on the Eel River and R.W. Matthews Dam on the Mad River , which are barriers to anadromy (Good et al. 2005). However, Clemento (2006) found evidence of small resident *O. mykiss* populations in tributaries of the Middle Fork Eel River. He also speculated that a few of these resident *O. mykiss* individuals may have possessed a rare allele common to summer steelhead within the drainage. This situation would likely be the result of recent or historical gene flow between anadromous summer steelhead and resident Rainbow Trout (Clemento 2006). Department scientist, Shaun Thompson, notes that resident *O. mykiss* are likely co-present with the anadromous form in the Eel River, but they have not been quantified. Thompson has caught small sexually mature male *O. mykiss* (as small as 10 cm) in smolt traps on the Mendocino Coast and states there are thousands of these fish in the Middle Fork Eel and Van Duzen rivers, though it is not possible to determine which are resident vs. anadromous. Thompson also documented residents on two summer steelhead redds during the spring of 2019. Given these observations, it is likely the resident and anadromous life history forms of *O. mykiss* co-occur in these areas (S. Thompson, CDFW, pers. comm., July 7, 2020).

Fecundity, among other biological and environmental factors, contributes substantially to steelhead reproductive success. Egg production has been shown to have a positive correlation with fish length, though there is still a high amount of variation in fecundity at size (Shapovalov and Taft 1954; Quinn 2018). Larger females tend to produce larger eggs as well as a greater number of eggs; however, there is an energy expenditure limit for gonad development that creates a tradeoff between the number of eggs and the size of the eggs produced (Quinn 2018). Thus, a female can either produce a lot of smaller eggs or fewer larger eggs, but not a large number of large 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. A report on summer steelhead at Skamania Hatchery on the Washougal River in Washington documented the average number of eggs per female to be around 2,400 (Hull and Allee n.d.). This Washougal strain was released into the Mad River, California for the first time in 1972 as part of a summer steelhead artificial propagation program at Mad River Salmon and Steelhead Hatchery (MRH) (Knutson 1975; CDFW 2016), though there is no evidence to show whether they established natural populations in the Mad River and Washougal stock was not used in propagation after 1980 (Cramer et al. 1995; CDFW 2016). Knutson (1975) found that Washougal strain summer steelhead returning to MRH had an average fecundity of almost 3,200 eggs per female. Annual reports from MRH in the mid- to late 1970s documented summer steelhead fecundity to be between 2,900 and 3,400 eggs per female (CDFG 1976, 1978, 1979, and 1980).

Eggs can incubate in the gravel for a period of several months. There are multiple factors that contribute to egg development and incubation time. Temperature has the greatest effect on incubation period; colder water slows development and warmer water increases rate of development (Quinn 2018). Incubation can take anywhere between 19 days at an average of 60°F and 80 days at an average temperature of 40°F (Shapovalov and Taft 1954). Dissolved oxygen (DO) also influences development. Higher DO levels lead to more rapid development and eggs exposed to low levels of DO during incubation produce much smaller alevins than those incubated in high DO (Quinn 2018). Steelhead eggs will begin to hatch after about a month (Shapovalov and Taft 1954). Fry emerge from the gravel about 2-3 weeks after hatching, once the yolk sac is fully or almost entirely absorbed, at which time they start to school along the stream banks (Shapovalov and Taft 1954). As they grow, individual fish develop small territories that they defend against other individuals in their age class throughout their first year of life (Shapovalov and Taft 1954; Barnhart 1986). Juvenile steelhead feed on many different species of aquatic and terrestrial insects, and sometimes on newly emerged steelhead fry (Barnhart 1986). As they grow the juveniles will move into deeper, faster water and are often found in riffle or swift run habitats (Shapovalov and Taft 1954; Barnhart 1986).

In comparison with other Pacific salmonids, steelhead have the most variable timing with regards to freshwater inhabitance, ocean entry, time spent at sea, and immigration back to freshwater (Barnhardt 1986). Steelhead can spend up to four years in freshwater before emigrating to the ocean and up to four years in saltwater before returning to spawn (Barnhardt 1986). Based on studies performed by Shapovalov and Taft (1954), most steelhead in Waddell Creek, California live to be three or four years old. According to Busby et al. (1996), age structure of California steelhead cohorts is dominated by age-3 fish that normally spend two years in the ocean after a year rearing in fresh water. Sparkman (2002), however, found that most steelhead smolts in the Mad River were age-2. Male steelhead tend not to live as many years as females due to higher post-spawn mortality rates (Shapovalov and Taft 1954). There is evidence from scale analysis that NC summer steelhead mostly return at either age-3 or age-4. Puckett (1975) showed that summer steelhead in the Van Duzen River tended to spend only one year in freshwater and two years in the ocean, while Middle Fork Eel River summer steelhead generally spent two years in freshwater and only one year in the ocean. Most fish returned to spawn after a total of three years regardless of how long they were in the river or ocean during that time. Age analysis of Middle Fork Eel River summer steelhead conducted by Michael Ward (CDFG) in the late 1980s differed from Puckett 1975 in that returning summer steelhead were split almost down the middle between age-3 and age-4 adults. Ward's results did agree with Puckett (1975) that most Middle Fork Eel River summer steelhead spent two years in freshwater before migrating to the ocean (Ward n.d.). Leo Shapovalov (CDFG) examined scales of Middle Fork Eel River summer steelhead and concluded that they spend two years in freshwater and two years in the ocean; returning for their first spawning at age-4 (CDFG 1953).

It is unclear whether observed differences in age at the parr-smolt transformation (Puckett 1975) is attributed to genetic variance or differential phenotypic responses to the environmental conditions in each river (Bjorkstedt et al. 2005). Cramer et al. (1995) suggested that growth rate affects when juveniles transition to smolts, which indicates the influence of environmental factors like flow and temperature. Bjorkstedt et al. (2005) states that rivers with highly productive environments like lagoons or warmer, fast-flowing water may provide growth opportunity enough for juvenile steelhead to reach optimum size for parr-smolt transformation after only one year. Sparkman and Holt (2017) found that Redwood Creek steelhead tend to smolt at age-1, which may be a result of temperature and sediment impairment acting as a selective force in favor of minimizing time spent in-river. Juvenile steelhead typically journey downstream during the spring and summer, with peak migration occurring in May (Shapovalov and Taft 1954; Everest 1973; Jones 1980). Sparkman et al. (2017) found that in Redwood Creek, smolt migrations may vary in response to stream flows. In higher flow years, Sparkman et al. (2017) found that migration was temporally more spread out and could be substantial in June, July, and August (Sparkman et al. 2017).

Not much is known regarding stock-specific distribution in the ocean. Other salmon species have a greater wealth of stock-specific ocean abundance information, but steelhead-specific research on this topic is lacking, and the ability to distinguish individual stocks has not been possible using standard methods (Barnhart 1986; Light et al. 1989; Moyle 2002). Unlike salmon *spp*., steelhead are rarely captured in the ocean. Thus, information specific to NC summer steelhead ocean distribution is not available. Limited tag recoveries by various fisheries research and management agencies in North America did not show any differences in ocean distribution by stock (Light et al. 1989). Fish, regardless of race or origin, comingle on shared feeding grounds (Light et al. 1988). It has frequently been observed that North American steelhead smolts quickly migrate offshore after ocean entry. Following the peak of smolt outmigration, juvenile steelhead abundance inshore decreases in June and July from Washington down to the central Oregon coast (Light et al. 1989). Fish originating from North American streams are thought to migrate north into and past the Gulf of Alaska during the summer months (Daly et al. 2014).

Ocean steelhead 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). Steelhead also tend to be closer to shore during the winter than during other times of the year (Light et al. 1989). Steelhead from California do not appear to migrate any farther west than the Gulf of Alaska (Light et al. 1989). Off the southern Oregon and Northern California coasts, however, there remains a population of juvenile steelhead thought to be the half-pounder run waiting to return to the rivers in late summer (Light et al. 1989). This population dissipates by August as the fish return inland but reassembles off the Oregon and California coasts when half-pounders make their way back into the ocean the following spring (Light et al. 1989; Hayes et al. 2016).

Steelhead migration patterns are thought to be strongly tied to “thermal avoidance,” remaining within a narrow range of acceptable sea surface temperatures, which suggests that migration may be contingent upon physiological responses to temperature (Hayes et al. 2016). However, half-pounder steelhead have been observed entering the Mad River during the fall when temperatures are higher in-river than they are in the ocean (M. Sparkman, CDFW, pers. comm., July 7, 2020). Ocean steelhead are typically found within seven meters of the sea surface in the epipelagic zone, though they have been found at more than three times that depth on some occasions (Light et al. 1989).

2.5 The ESU/DPS Concept in Management of Pacific Salmonids

Preservation of genetic diversity in salmonids can be difficult due to the need to protect nontaxa (Nehlsen et al. 1991), which are below the biological species level. Behnke (1993) stated, “Obviously, any conservation program to preserve biodiversity must begin at the lowest nontaxon level.” The objective to preserve genetic diversity, which is a key component in species adaptation to environmental changes, provides justification for protection of distinct population segments. Loss of specific population segments can contribute to the decline of the species as a whole and increase its risk of extinction. Gustafson et al. (2007) also found that population-level extirpation results in loss of ecological, genetic, and life history diversity, which can manifest at a larger geographic scale (e.g., ESU-level) and even species extinction. Thus, protection of population segments is biologically appropriate to preserve this necessary genetic variation within species.

The ESU concept was first mentioned by Ryder (1986) to try to determine which gene pools to preserve at the species, subspecies, or population level in zoos, although Ryder did not define any specific parameters for classifying an ESU. The concept was further developed by NMFS in Waples (1991b) to define how population distinctness should be assessed and determined for ESA listings. Waples (1991b p.12) defines an ESU as follows:

> “A vertebrate population will be considered distinct (and hence a “species”) for purposes of conservation under the [Endangered Species] Act if the population represents an evolutionarily significant unit (ESU) of the biological species. An ESU is a population (or group of populations) that: 1) is substantially reproductively isolated from other conspecific population units, and 2) represents an important component in the evolutionary legacy of the species.”

Waples (1991b p.12) further clarified, “isolation does not have to be absolute, but it must be strong enough to allow evolutionarily important differences to accrue in different population units.” These evolutionarily important differences are defined on a genetic basis, thus, if a population provides a substantial genetic diversity to the species it would satisfy the second criterion of what constitutes an ESU (Waples 1991b). NMFS subsequently adopted the ESU parameters defined by Waples (1991b) as policy specific to Pacific salmon under the ESA (Notice of policy: Policy on applying the definition of species under the Endangered Species Act to Pacific salmon, 1991).

The United States Fish and Wildlife Service (USFWS) and NMFS published a new policy in 1996 recognizing that the ESA definition of a species extends to "distinct population segments" and adopting the term as a new management unit for the purposes of listing, delisting, and reclassifying vertebrates under the ESA (Policy regarding the recognition of distinct vertebrate population segments under the Endangered Species Act, 1996). This policy calls upon two concepts when identifying a DPS: discreteness and significance. A population segment is discrete if it meets either of two conditions specified in the DPS Policy (Policy regarding the recognition of distinct vertebrate population segments under the Endangered Species Act, 1996, p. 4725):

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, its significance, both biologically and ecologically, can then be evaluated by considering characteristics specified in the DPS Policy. These include but are not limited to (Policy regarding the recognition of distinct vertebrate population segments under the Endangered Species Act, 1996, p. 4725):

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, or
4. Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics."

In 2001, the U.S. District Court in Eugene, Oregon determined in the *Alsea* decision that based NMFS DPS delineations, the ESA did not allow for listing only a subset of that DPS (NMFS 2006). This led to NMFS reviewing West Coast salmonid listings and continuing to use the established ESU concept but to include resident fish that co-occurred with the anadromous form. In the public comment period of this proposal, USFWS, the agency with jurisdiction over resident *O. mykiss*, disagreed that the resident form of the population should be listed or included within the steelhead ESUs. In response to these considerations, NMFS stated that anadromous and resident *O. mykiss* remained separated due to "physical, physiological, ecological, and behavioral factors," which warranted consideration of anadromous populations as separate DPSs (NMFS 2006). This led NMFS to propose a shift from applying the ESU Policy to applying the new DPS Policy to West Coast steelhead listings under the ESA, thus redefining the NC steelhead ESU, as it was originally listed, to be the NC steelhead DPS (Table 2.1) (NMFS 2006).

The ESU and DPS concepts as conservation and management units have been topics of continued discussion (Dizon et al. 1992; Moritz 1994; Vogler and Desalle 1994; Pennock and Dimmick 1997; Bowen 1998; Crandall et al. 2000; Fraser and Bernatchez 2001; de Guia and Saitoh 2007). Dissatisfaction has been expressed regarding the rigidity and consequent inability to define DPSs based on demographic and behavioral data as well as the ambiguity surrounding how to quantify distinctness (Pennock and Dimmick 1997). However, Waples (1998) argued that ESU determinations place great importance on demographic and behavioral traits, which are more often referred to as life history characteristics. With regard to the ambiguity critique, there are a variety of interpretations of evolutionary significance, which rely both on genetic differences and local environmental adaptations (Waples 1995). Often decisions on ESU parameters come down to best professional judgement based on available data and how it is evaluated (Waples 1991b, 1995).

Genetic analyses can be useful tools in evaluating ESU and DPS criteria. NMFS has relied heavily on genetics to pinpoint reproductive isolation since the relationship between genetics and life history characteristics is largely unknown (Busby et al. 1996). Additionally, in evaluating “evolutionary legacy,” NMFS has looked specifically at genetic variability that has arisen from past evolutionary events and will provide the source of future variability (Busby et al. 1996). MacLean and Evans (1981) argued that fishery managers need to introduce a genetics perspective to fishery management by way of the stock concept in order to preserve locally adapted populations within a species.

The concept of subspecies has been debated by scientists for over a century and continues to be controversial, especially due to the qualitative nature of subspecies definitions (Haig et al. 2006). Subspecies of fish have generally been defined based on allopatry (Haig et al. 2006) or geographic isolation. Thus, subspecies classifications of fish are essentially equivalent to geographic races (Haig et al. 2006). NMFS and USFWS have most often listed subspecific classifications of fish as DPSs rather than subspecies, although only 15% of federal fish listings are below the species level. Haig et al. (2006) proposes two essential biological criteria that should be met in classifying a subspecies; 1) discreteness and 2) biological significance of the population, both in relation to the remainder of the species to which it belongs (Haig et al. 2006). These two criteria are consistent with the NMFS DPS Policy (Policy regarding the recognition of distinct vertebrate population segments under the Endangered Species Act, 1996).

Table 2.1. *Steelhead Distinct Population Segments (DPS) listed under the federal Endangered Species Act (ESA).*

DPS	ESA Status
Southern California steelhead	Endangered
South Central California Coast steelhead	Threatened
California Central Valley steelhead	Threatened
Central California Coast steelhead	Threatened
Northern California steelhead	Threatened

2.6 Genetics and Genomics

2.6.1 Role of Genetics and Genomics in Evaluating Steelhead Population Structure

Most genetic studies quantifying population structure of salmon and steelhead species, including *O. mykiss* in Northern California, have used neutral genetic markers (e.g., microsatellite DNA). Neutral markers are not specifically associated with a particular life history trait and it is assumed that they are not under direct selection. This class of genetic marker has been, and continues to be, used to investigate and define salmonid listing units and population structure (e.g., Busby et al. 1996; Pearse et al. 2019) in California and across the Pacific Northwest. Neutral markers have been used successfully for decades to delineate populations and ESUs based on more or less reproductively isolated lineages. Studies using presumably neutral genetic markers have shown that, in many cases, salmon and steelhead populations with different migration timing from one river are more closely related to each other than to populations with the same migration timing in other rivers (Chilcote et al. 1980; Waples et al. 2004; Kinziger et al. 2013; Arciniega et al. 2016). Neutral markers are the standard for elucidation of species' evolutionary histories.

More recently, the advent and rapid development of "adaptive" genetic markers has sparked debate among fishery managers and geneticists. Adaptive genetic markers have putative associations with specific life history characteristics. In the case of NC steelhead, the single associated trait of interest in this status review is migration timing. Work by Hess et al. (2016) and Prince et al. (2017) identified a specific genomic region that they determined to be strongly associated with migration timing in *O. mykiss*. These findings prompted questions surrounding conservation priorities and defining management units. Waples and Lindley (2018), Pearse (2016), Shafer et al. (2015), and Allendorf et al. (2010) provide reviews and cautions regarding the use of adaptive genetic markers for defining conservation units.

On the one hand, adaptive genetic markers provide putative associations with specific life-history characteristics: the "genetic type" infers information about a phenotype of interest. In the case of *O. mykiss*, the single associated trait of interest is migration timing. Alternatively, neutral markers have been used successfully for decades to delineate populations and ESUs based on more or less reproductively isolated lineages. Neutral markers are used to estimate genetic relationships and evolutionary history of species as a whole, not specific traits. Genes may have an evolutionary history that is different from the species history.

2.6.2 Patterns of Genetic Population Structure

The genetic structure of steelhead populations has been found to be driven by geography, a pattern referred to as “isolation by distance.” Evidence of isolation by distance is shown in *O. mykiss* populations throughout their range. Bjorkstedt (2005) and Arciniega et al. (2016) suggest that salmonid populations are isolated by distance and genetic variation increases with greater distances between watersheds. For example, populations of steelhead in the Columbia River on the northern border of Oregon are more genetically distant from the Eel River populations in California than they are from populations in the Rogue or Klamath Rivers in Southern Oregon and Northern California, respectively (Figure 2.2) (Arciniega et al. 2016). Nielsen (1999) found a pattern of isolation by distance when looking at microsatellite loci of Southern California steelhead populations and Northern California populations. 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. Older studies based on neutral mtDNA analysis also showed a pattern of isolation by distance among coastal California steelhead populations, including more extensive studies of populations spanning the western coast of the United States (Hatch 1990; Reisenbichler et al. 1992; McCusker et al. 2000).

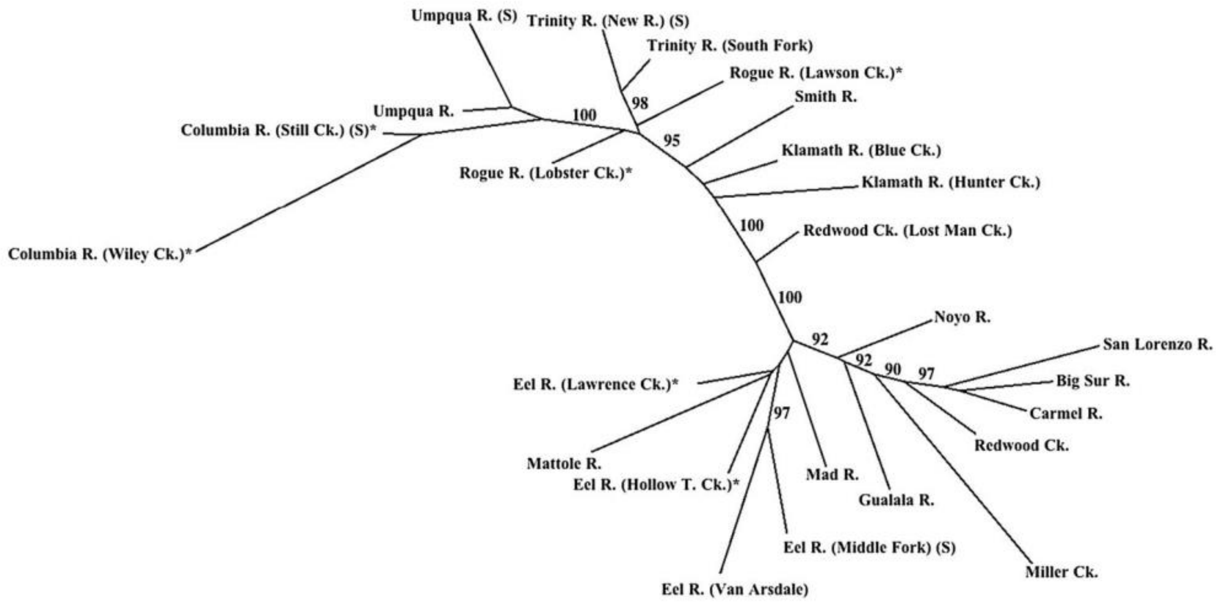


Figure 2.2. *Unrooted neighbor joining tree showing genetic variation from watersheds from Oregon and California (from Arciniega et al. 2016). Greater physical separation on the tree reflects greater degrees of genetic variation between populations.*

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 steelhead populations there was evidence that population structure is dependent on geographic distance. Phylogeographic trees suggested that population structure was almost entirely consistent with geographic proximity.

2.6.3 Defining the NC Steelhead DPS

The geographic boundaries of the ESUs of Pacific salmon and DPSs of Pacific steelhead have historically used neutral genetic variation to identify geographic boundaries between populations. Isolation by distance has been a consistent pattern when describing neutral genetic variation within and among populations. Previous status reviews conducted by NMFS relied on patterns of neutral genetic variation to help define the original ESUs. The Northern California steelhead ESU was first defined by Busby et al. (1996) and covers river basins from Redwood Creek in Humboldt County down to the Gualala River (Busby et al. 1996). The ESU contains both the winter and summer steelhead migration ecotypes, though the Mattole River is considered the southern extent of summer steelhead. Genetic data used to originally define the Northern California steelhead ESU included chromosome counts, mtDNA data, and allozyme data. Busby et al. (1994) found genetic differences between steelhead populations from north and south of the Klamath River with Redwood Creek (basin immediately south of the Klamath River) functioning as a transition zone between these groups. The Northern

California steelhead ESU (redefined as DPS in 2006) follows this divide with Redwood Creek designated as the northern boundary based off the samples showing genetic similarities with populations from the Klamath River as well as those south of the Klamath River (Busby et al. 1996) (Figure 2.3).

Nielsen (1994) and Nielsen et al. (1994, both as referenced by Busby et al. 1996), used mtDNA data from coastal steelhead and determined that those from Humboldt Bay to Gualala Point had a different mtDNA haplotype than those from central and south coast areas. Additional analysis in Busby et al. (1996) confirmed this finding and again divided the populations by latitude with the southern boundary of the NC steelhead ESU extending to just north of the Russian River.

Garza et al. (2004) evaluated population structure across coastal California populations using microsatellite loci to understand the relationship between genetic distance and geography of coastal steelhead populations. They constructed a bootstrap consensus tree that shows clustering of geographic locations corresponding to five DPS assignments in coastal California steelhead (Figure 2.3) (Garza et al. 2004). The long terminal branches in this consensus tree show 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. Another study of California coastal populations constructed phylogeographic trees and found clear separation between the Northern California steelhead DPS and the Central California Coast steelhead DPS, which are divided geographically by the Lost Coast (Bjorkstedt et al. 2005).

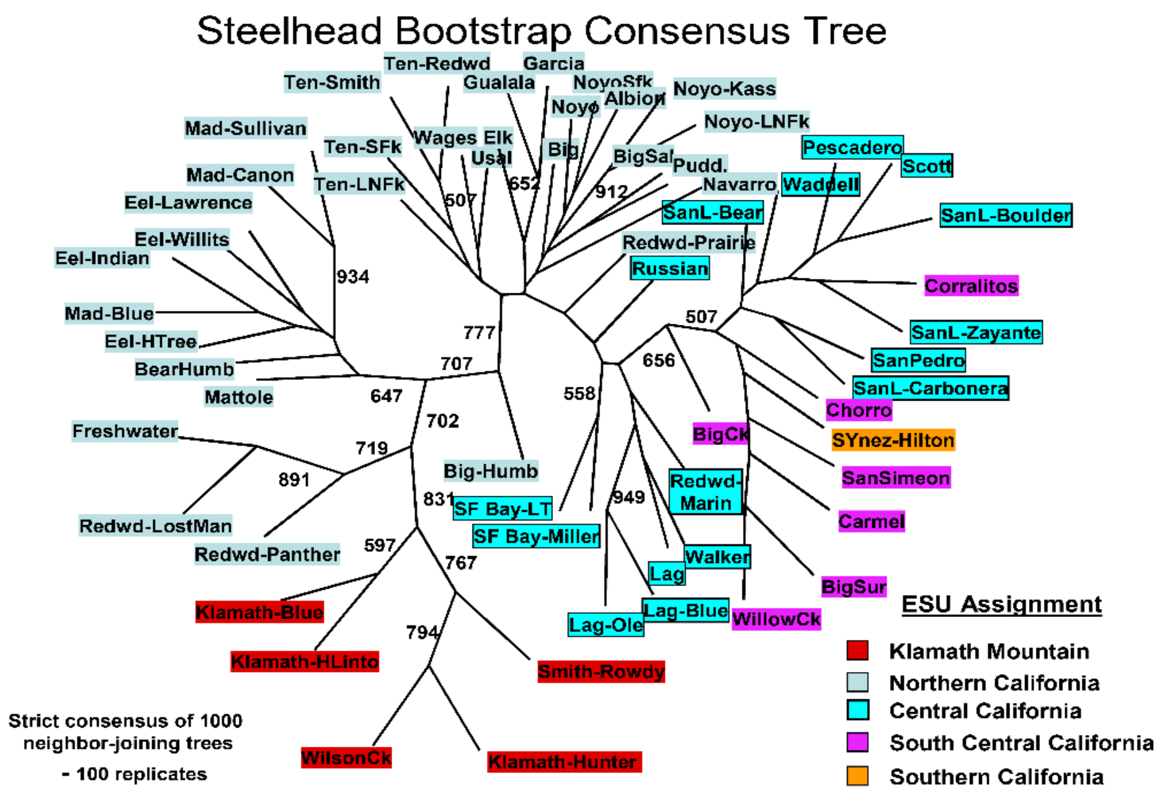


Figure 2.3. *Majority-rule consensus tree, with genetic data bootstrapped 1,000 times, showing chord distances and neighbor-joining trees for 62 coastal California steelhead populations. (from Garza et al. 2004).*

A more recent study by Garza et al. (2014) clustered samples into five genetic groupings that were mostly consistent with coastal steelhead DPS boundaries. Clusters 3 and 4 included NC steelhead DPS watersheds and cluster 5 encompassed the Klamath Mountains Province DPS plus Redwood Creek. Clusters 3, 4, and 5 did not present a pattern of isolation by distance in contrast to the other two clusters in the study. This result may be an artifact of hatchery broodstock collection and juvenile release practices within this area, however, these practices occurred throughout all regions. Because of this, the relationship observed is most likely attributable to a higher level of gene flow between tributaries in clusters 3, 4, and 5, which eliminated the isolation by distance seen in other tributary regions.

2.6.4 Role of Genetics and Genomics in Migration Phenotypes

There have been many genetic studies evaluating population structure and the role of genetics on run timing in salmonids. Studies investigating the effects of geographic distance and migration timing on population structure have shown that summer and winter steelhead populations within a basin are more closely related to each other than to the same run type in nearby basins (Chilcote et al. 1980; Nielsen & Fountain 1999; Clemento 2006; Arciniega et al. 2016; Prince et al. 2017). Thus, geographic location has a more significant effect on population genetic structure than variation in migration timing. Chilcote et al. (1980) was the first to show using allozyme data that summer and winter steelhead ecotypes from the same river were more similar to each other than to populations from other rivers with the same migration timing. Those authors analyzed five polymorphic allozyme loci from both ecotypes from Kalama River, a tributary to the Columbia River in Washington that had introductions of nonindigenous steelhead for 15 years prior to their study. They did not detect any significant genetic differentiation between different migration ecotypes. Chilcote et al. (1980) also directly observed summer and winter individuals spawning together.

Nielsen and Fountain (1999) analyzed microsatellite data from summer and winter steelhead in the Middle Fork Eel River and found that there was less genetic divergence between the two populations of steelhead in the Middle Fork than with geographically proximate coastal winter steelhead populations. Nielsen and Fountain determined only a small amount of the microsatellite variation contributed to the differences found between summer and winter steelhead and that the ecotypes within the Middle Fork Eel River were more closely related to each other than to Mendocino Coast winter steelhead. Again, geographic location was the dominant factor in describing genetic population structure as opposed to migration timing.

Prince et al. (2017) used genome-wide reduced representation (RADseq) data from populations in five coastal locations of California and Oregon, four of which had premature migration phenotypes (summer steelhead and spring Chinook Salmon). Collections focused on individuals that were either clearly premature or mature life-histories but intentionally avoided the collection and analysis of fish with intermediate migration timing that could carry both forms of the trait (heterozygotes). Prince et al. suggest that in a heterozygote the premature migration allele is recessive, thus would not result in expression of the premature migration phenotype. Consistent with previous studies, the authors found that, with the exception of the area around the GREB1L genomic region, their data showed that summer and winter populations within a watershed are more closely related to each other than those in nearby watersheds, thus further supporting the long-standing isolation by distance model of population structure.

One study conducted in the Klamath-Trinity basin found a reduction in heterozygosity among five groups of steelhead, which they attributed to a subpopulation structure within the basin and concluded that there are at least two genetically discrete populations with different migration timing (Papa et al. 2007). However, SWFSC later conducted an analysis of the same samples using two SNP loci located in the GREB1L region associated with run timing phenotypes (Pearse et al. 2019). When studied during migratory season on a temporal scale, the summer and winter phenotypes showed a continuum of allele frequencies with overlap among fish carrying the summer and winter migration variants (Pearse et al. 2019), thus demonstrating that individual fish can, and do at times, carry the genetic variants for both summer and winter migration timing and that there is not a temporal break in migration timing between individual fish carrying the different GREB1L allelic variants. The variation in analyses of the two studies based on the same samples indicates that, although aspects of Papa et al. (2007) may suggest divergent populations based on run timing, varying allele frequencies do not support the conclusion that summer and winter migration phenotypes are genetically distinct populations.

The majority of studies using multiple genetic markers (loci) that are distributed throughout the genome suggest summer migration timing arose through parallel evolution in various populations and genetic variation is distributed among steelhead populations based largely on their geographic proximity to each other (Waples et al. 2004; Arciniega et al. 2016; Pearse et al. 2019). These findings align with the genetic relationship found between anadromous and resident phenotypes, another *O. mykiss* life history trait with genetic influence (Behnke 2002; Docker and Heath 2003; Olsen et al. 2006). Arciniega et al. (2016) compared microsatellites and SNPs from multiple populations of steelhead in Oregon and Northern California to evaluate phylogeographic relationships of summer and winter steelhead ecotypes. Further reiterating what we previously stated, genetic relatedness was strongly associated with geographic distance showing limited gene flow among river basins and suggesting repeated parallel evolution of summer migrating ecotypes in multiple river basins. Arciniega et al. (2016) found that almost half of the variation in genetic differentiation was explained by geographic distance among populations. Although summer migration ecotypes were shown to have arisen through parallel evolution, Arciniega et al. (2016) note that these evolutions occurred where genetic variation already existed and where ecological conditions could support the summer migration ecotype.

2.6.5 Role of GREB1L Genomic Region in Migration Timing

The GREB1L genomic region was first connected to run timing by Hess et al. (2016) and was subsequently applied in studies evaluating life history phenotypes (Prince et al. 2017; Micheletti et al. 2018; Pearse et al. 2019). When evaluating individuals with distinct summer or winter migration phenotypes, the GREB1L region shows an association with run-timing variations in *O. mykiss* populations (Hess et al. 2016; Prince et al. 2017; Micheletti et al. 2018; Kannry et al. 2020). Individual steelhead are either homozygous or heterozygous for genetic markers in the GREB1L genomic region. Homozygotes only have alleles associated with a single run type on both homologous chromosomes. In contrast, heterozygotes have alleles associated with both run types. Fish that are heterozygous for the GREB1L genetic markers display an intermediate run timing. Studies evaluating the impact of GREB1L on salmonid run timing have found that the life history characteristic is not only associated with the GREB1L gene but with ROCK1 and other nearby genomic regions (Micheletti et al. 2018; Narum et al. 2018; Thompson et al. 2020).

Prince et al. (2017) compared SNPs between migration types and found significant variation at loci associated with what the authors termed the “premature” migration phenotype of steelhead (summer steelhead). When they compared SNPs at those loci between two geographically separated populations, the strongest associated SNPs from both sample locations were near or within the GREB1L genomic region, suggesting that the same single genomic region was associated with migration phenotype in two different steelhead DPSs. Prince et al. (2017) suggest that a single evolutionary event produced the GREB1L premature migration allele rather than multiple parallel evolutions, and that if lost, these premature migration alleles are not expected to re-evolve in a reasonable amount of time. Where other studies concluded that premature migration timing was a result of parallel evolution due to the observed geographic patterns of genetic variation, Prince et al. (2017) suggest that the presence of the premature migration allele across geographically separate populations indicates that the allele did not evolve separately in multiple populations but existed already within standing genetic variation. Additionally, when evaluating GREB1L heterozygotes from another dataset, the authors found that they displayed an intermediate run timing, which suggests that the premature migration allele is not masked in phenotypes displayed by heterozygotes. Thus, there is not a clear dominance of the early versus late migration genetic variants within individual fish. Prince et al. (2017) concluded that the premature migration allele would not continue to exist in populations without the premature migration phenotype and implied that genetic variants correlated with migration timing differences arose only once and then spread between populations.

Thompson et al. (2019) investigated the genomic basis of migration timing in Rogue River Chinook Salmon. They focused on the GREB1L region and found, similar to *O. mykiss*, a significant correlation between migration timing phenotypes and genetic variation in the GREB1L region. Spring Chinook possessed the "spring" migration allele, fall Chinook possessed the alternate "fall" migration allele, and fish of intermediate migration timing possessed copies of both spring and fall alleles. Similar to Prince et al. (2017), Thompson et al. (2019) suggested that selection against the early migration phenotype could lead to the rapid loss of the GREB1L early migration allele.

Micheletti et al. (2018) expanded the section of the genome analyzed by Prince et al. (2017) in a study of Columbia River steelhead populations. Whole genome sequencing was used to characterize the GREB1L genomic region and concluded that migration phenology is likely connected to more than just the GREB1L region of chromosome 28 and may be linked to ROCK1 and other genes near the GREB1L region. The authors also determined that GREB1L had a greater association with arrival to spawning grounds when comparing coastal and inland lineages, rather than freshwater entry as proposed by Hess et al. (2016) and Prince et al. (2017). This finding is likely more relevant to steelhead populations in the Pacific Northwest than those in Northern California because of their relatively short migration distance from the ocean to their spawning grounds. Narum et al. (2018) found similar results to Micheletti et al. for the ROCK1 and GREB1L genomic regions in Chinook Salmon. Narum et al. analyzed GREB1L in Chinook Salmon, mapping a coastal population with both spring and fall Chinook Salmon using genome-wide SNPs and found that, in addition to GREB1L, other genomic regions including ROCK1 were associated with migration timing across three distinct phylogenetic lineages. Both

Micheletti et al. (2018) and Narum et al. (2018) indicate that maturation life history is not solely determined by the GREB1L gene but is instead polygenic with multiple genomic regions contributing to divergent selection on maturation phenotypes.

Pearse et al. (2019) discussed GREB1L genotype data collected by the NMFS South West Fisheries Science Center (SWFSC) that showed multiple *O. mykiss* collections in California that contained individuals with all three genotypes of summer, winter, and hybrid summer-winter. Using GREB1L genetic markers, Pearse et al. also found homozygous winter, homozygous summer, and heterozygote steelhead in populations not known to currently support summer migration ecotypes, which suggests a larger distribution of the GREB1L haplotypic variations than found in other studies. This was not observed by Prince et al. (2017) because populations with expression of an intermediate migration phenotype (e.g., not clearly defined summer or winter phenotypes) were intentionally excluded from their study (Pearse et al. 2019). Additionally, when evaluating two SNPs in individuals that passed Van Arsdale Fisheries Station on the upper mainstem Eel River from 2009-2017, Pearse et al. (2019) found complete overlap in return timing of winter homozygote and winter-summer heterozygote genotypes, as well as some overlap of summer homozygotes that returned within the typical winter migration period. When conducting parentage analysis of this collection, although most individuals were winter homozygotes, some offspring were heterozygotes with one summer steelhead and one winter steelhead parent, which indicates interbreeding of parents with different GREB1L genotypes that resulted in offspring with a mix of GREB1L genotypes.

Kannry et al. (2020) sequenced both anadromous and resident forms of *O. mykiss* on the Eel River at GREB1L and other genomic regions for overall genetic differentiation. Kannry et al. found that resident populations above Scott Dam had maintained genetic diversity among life history forms including heterozygotes with summer migration alleles, suggesting summer steelhead were above the dam prior to its construction in 1922 and could potentially repopulate with dam removal. Individuals in the South Fork Eel River, however, were all homozygous for winter migration alleles, suggesting winter steelhead populations may not maintain any summer migratory alleles and summer migratory alleles would not arise from current genetic variation in this or other populations that lack the summer migration phenotype.

Collins et al. (2020) looked at the adaptive markers identified by Hess et al. (2016) and Micheletti et al. (2018), including GREB1L and ROCK1, as well as additional adaptive genetic markers within the same genomic region, to determine which genetic combinations resulted in various migration phenotypes of Columbia River steelhead. Collins et al. (2020) found that adaptive genetic markers separated individuals by their adult migration timing. They also found that different heterozygote haplotypes were predominant depending on geographic location (inland vs. coastal). Temperature and precipitation were found to be important selective pressures in this pattern of variation. This additional level of examining the associations of haplotype blocks with each other and with migration timing improves our understanding of patterns of genetic variation related to migration timing in steelhead. Collins et al. (2020) demonstrate that simply defining winter and summer steelhead ecotypes as two distinct units does not account for the variety of phenotypes expressed by various heterozygous haplotypes of the genomic region containing GREB1L and ROCK1.

Thompson et al. (2020) recently examined the association between the GREB1L/ROCK1 region on chromosome 28 in Chinook Salmon and spawn migration timing. While this study did not examine steelhead, the findings are relevant since the association of this genomic region in both steelhead and Chinook Salmon is now well established in the literature. Thompson et al. (2020) largely reaffirmed the strength of association between genetic variation in this genomic region with migration timing phenotype, and further that the genomic blocks associated with the early and late migration timing phenotypes are very well conserved across populations and are exchangeable. They also noted that the migration timing of fish that are heterozygous for both the early and late migrating genetic types does vary between watersheds, indicating that there are some environmental and polygenic effects on migration timing as well.

In February 2020, NMFS' Northwest Fisheries Science Center held a workshop for geneticists and scientists with experience in salmon and steelhead conservation and management to present data and discuss the associations between genetic variation and migration timing in salmonids. Results of this workshop, which explored the state of the science, conservation implications, and future research needs regarding the simple genomic association with run timing in Chinook Salmon and steelhead, are documented in Ford et al. (2020). Workshop participants did not attempt to come to a consensus on all conclusions, rather they developed points of general agreement and points of residual uncertainty. A summary of the areas of agreement and uncertainty among workshop participants is presented in *Appendix A*. Although all of the findings and discussion in Ford et al. (2020) are important, the following selected conclusions, which were areas of general, though not necessarily unanimous, consensus, are excerpted here because they are particularly relevant to this status review:

- A single region in the genome has a strong statistical association with adult run timing.
- The causal variant(s) for adult run timing remain to be identified.
- Heterozygotes are likely an important mechanism for the spread and maintenance of the early migration alleles over long time scales.
- The early and late allelic variants that have been well characterized evolved long ago in each species' evolutionary history. The allelic variants for early migration have not arisen independently via new mutations from the genomic background of late migration individuals in each watershed.
- Conservation units should continue to be defined by patterns of genetic diversity across the genome, rather than by variation in small genomic regions correlated with specific traits.

- Spring Chinook Salmon and summer steelhead occupy a specialized ecological niche—upstream areas accessible primarily during spring flow events—that has made them particularly vulnerable to extirpation or decline due to habitat degradation.
- The evaluation of risk to early returning groups (e.g., spring-returning Chinook salmon, summer steelhead) needs to consider what we now know about the genetic basis of adult return time.
- The finding that the “early run” trait has a simple genetic basis implies that the “early run” phenotype is at greater risk than if the trait resulted from many genes because loss of the “early” allele(s) equates to loss of the phenotype.

2.6.6 Conclusions on Genetics and Genomics

The Petitioner references Prince et al. (2017) in the Petition to substantiate their assertion that NC summer steelhead should be listed as an endangered species under CESA. From Prince et al. (2017) the Petitioner draws two conclusions regarding the genetics of summer steelhead: 1) summer steelhead are genetically distinct from winter steelhead in the same watersheds, and 2) if premature migration alleles are lost, summer steelhead cannot be expected to re-emerge from winter steelhead populations in time frames relevant to conservation planning, therefore, they should not be listed in the same DPS. These concepts are nuanced, and new information continues to emerge; however, at this time, the Department does not support the conclusion that summer steelhead are distinct and should be listed separately under CESA.

Heterozygosity at the GREB1L locus indicates that interbreeding occurs between summer and winter steelhead, including those in the Eel River basin (Prince et al. 2017; Pearse et al. 2019; Kannry et al. 2020), which means they are not reproductively isolated and have an interconnected genetic legacy. Results from Pearse et al. (2019) indicate that summer and winter steelhead are not reproductively isolated, but rather a single population with varying migration times among individuals. Interbreeding of the two ecotypes likely occurred historically, though quantifying natural levels of interbreeding is difficult (Ford et al. 2020). Pearse et al. (2019) found almost a complete overlap in the return timing of heterozygotes and homozygous winter genotypes at Van Arsdale on the upper Eel River in addition to some homozygous summer genotypes returning within the winter steelhead migration timing window. From this observation the authors suggest that there must be some interbreeding between alternate GREB1L genotypes, which would produce full-sibling offspring with multiple GREB1L genotypes and their corresponding run timing phenotypes. Matings between upper Eel River winter homozygotes and heterozygotes were found to produce offspring with both heterozygous and homozygous winter genotypes (Pearse et al. 2019). Early and late migration ecotypes within a watershed have consistently been found to be more closely related to each other than they are to migration ecotype equivalents in nearby watersheds (Chilcote et al. 1980; Thorgaard 1983; Nielsen and Fountain 1999; Arciniega et al. 2016; Prince et al. 2017), thus geographic location has a more significant effect on population genetic structure than variations in migration timing. This has continued to hold true in recent studies that have looked across much larger portions of the genome (Micheletti et al. 2018; Ford et al. 2020). The genomic region consisting of GREB1L, ROCK1, and the intergenic region, although strongly associated with migration timing (Prince et al. 2017; Micheletti et al. 2018, Narum et al. 2018; Pearse et al. 2019), only accounts for a small amount of genetic variation between summer and winter steelhead ecotypes.

Although the sequences within the GREB1L/ROCK1 genomic region associated with early migration timing appear to be monophyletic, having arisen once in evolutionary history and are likely to be irreplaceable if lost, given that these genetic sequences are highly conserved across multiple populations, if lost in one population there is a potential that they could be reintroduced through intra-basin migration from another source population (Thompson et al. 2020). Additionally, heterozygotes may act as a reservoir for early migration alleles (Ford et al. 2020). This is likely dependent on habitat conditions and patterns of dominance at the GREB1L/ROCK1 genomic region, which are not fully understood in steelhead. Available data for Chinook Salmon suggest that there is either an additive mechanism or that the early allele is dominant (Ford et al. 2020) The existence of heterozygotes is inherently adaptive. The resident life history form of *O. mykiss* may also act as an important reservoir for the early migration alleles. Early migration alleles have been found in resident fish in some coastal streams (Ford et al. 2020; Kannry et al. 2020). The preservation of this allele in the resident life history form of *O. mykiss* suggests that overall risk of losing genetic variation at GREB1L/ROCK1 may be reduced in streams where summer steelhead exist or were extirpated but residents still reside (Ford et al. 2020).

The strong genomic association of the GREB1L/ROCK1 region of chromosome 28 with migration timing is an important scientific discovery that adds to our overall understanding of the genetics of summer migration timing in the NC steelhead DPS; however, it does not in itself show that summer steelhead are a genetically distinct subspecies. Delineation of management units based off run timing alone would not accurately reflect the evolutionary legacy of Northern California coastal steelhead. Review of available literature supporting the concept that genetic distinctiveness is driven by isolation by distance rather than life history variation within a population is not negated by the new genomic information presented by Prince et al. (2017).

There are still areas of uncertainty surrounding the genetics behind run timing expression. Waples and Lindley (2018) note that we do not know if the genes identified by Prince et al. (2017) are actually responsible for migration timing differences. It has generally been accepted that run timing variation is strongly associated with genetic sequence variation in the GREB1L/ROCK1 region of chromosome 28; however, all studies to date have established only correlation, not causation. Although the causal variants are likely contained within the GREB1L/ROCK1 genomic region, the underlying biochemical mechanisms responsible for phenotypic variation are still unknown (Ford et al. 2020). It is now well established in the peer-reviewed literature that the DNA sequence of the GREB1L/ROCK1 region is highly conserved within, and in many cases, across *Oncorhynchus* species. Thus, if a particular variant is lost from a population, it can be reintroduced from another population where it is still present.

Waples and Lindley (2018) note several issues with using adaptive genetic markers for defining conservation units and suggest that ESUs incorporate conservation of genetic diversity including variation in migration timing. The summer steelhead ecotype is an important diversity component of the steelhead species that should be preserved as part of the NC steelhead DPS through the use of existing fishery management options. The strong genomic association of GREB1L/ROCK1 and associated regions with adult migration timing (e.g., Prince et al. 2017) is an important result that sheds light on the genetic underpinnings of early run timing in steelhead and other salmonids. However, the Department finds that this genomic association is only one part of the total evolutionary heritage of summer steelhead and, by itself, is not sufficient or appropriate differentiation to consider NC summer steelhead a separate subspecies under CESA at this time.

3. STATUS AND TREND

3.1 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). Within the NC steelhead DPS there is some level of genetic exchange that makes the populations more related to each other than they are to populations outside of the DPS.

The Department has defined and managed runs of anadromous salmonids based on genetic distinctiveness, run-timing differences, juvenile outmigration timing, and watershed (CDFG 1998). For this review, we consider NC summer steelhead to be one of two regional ecotypes (summer and winter) within the NC steelhead DPS. The ecotype concept was first introduced by Turesson (1922) and was used to define subspecies units resulting from genotypical responses, or phenotypic expression, of a population to particular habitats (Turesson 1922). Salmonids have substantial genetic structuring linked to local adaptations that manifest in various life history strategies including those that represent sympatric ecotypes such as winter and summer steelhead runs (Keeley et al. 2007; Pearse et al. 2009; Wollebaek et al. 2017; Collins et al. 2020).

Summer and winter steelhead ecotypes arrive at their respective spawning grounds during different times of year but can overlap somewhat in spawning distribution and timing. Access to various stream reaches can change from year to year, so degree of reproductive isolation may be somewhat inconsistent and fluctuate over the long-term. Generally, winter and summer steelhead within the same Northern California streams have been treated as two different populations, although some assemblages of juveniles likely contain hybrid offspring from parents of both run types.

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) also introduced four criteria for assessing viability of salmonid populations: abundance, productivity (over the entire life cycle), population spatial structure, and diversity. In this chapter, we evaluate, to the best of our ability, these four criteria for NC summer steelhead independent populations.

3.2 Sources of Information

We reviewed all sources available to us for this status review. Sources include literature review, the CESA listing petition, previous federal status reviews, Department reports and documents, newspaper articles, annual reports from existing NC summer steelhead surveys, and historical reports. Agency staff with knowledge of watersheds supporting NC summer steelhead were also consulted for information. Further study and comprehensive monitoring would help elucidate current population status and trends; however, due to lack of historical data determining true historical abundance is likely not possible. Historical information on distribution and abundance was limited to mostly anecdotal reports, personal observations, and a few historical stream surveys. 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.

3.3 Historical and Current Distribution

The NC steelhead DPS includes ten NMFS-defined summer steelhead populations from Redwood Creek in the north to the Mattole River in the south (NMFS 2016b). All summer steelhead populations in the NC steelhead DPS were considered to be historically functionally independent, which means they had a high probability of persisting over 100-year time scales and their risk of extinction was not significantly influenced by exchanges of individuals with other populations (McElhany et al. 2000; Bjorkstedt 2005). Populations in the North Fork, South Fork, upper mainstem, and upper middle mainstem of the Eel River, as well as Larabee Creek, are either very small and inconsistently present or locally extirpated and have been classified by NMFS as data deficient (Spence et al. 2008). There is no pre-industrial documentation of summer steelhead presence in these areas; all historical information begins post-1900. See Table 3.1 for a list of all historical NC summer steelhead populations. Figure 3.1 provides a visual representation of the historical and current distribution of NC summer steelhead within the NC steelhead DPS.

Table 3.1. *Summer Steelhead Populations in the Northern California Steelhead DPS (from Bjorkstedt et al. 2005). Parentheses indicate the population is extirpated (NMFS 2016b).*

Summer Steelhead Population	Historical Population Status
Redwood Creek	Functionally Independent
Mad River	Functionally Independent
Van Duzen River	Functionally Independent
South Fork Eel River	Functionally Independent
Larabee Creek	Functionally Independent
North Fork Eel River	Functionally Independent
(Upper Middle Mainstem Eel River)	(Functionally Independent)
Middle Fork Eel River	Functionally Independent
(Upper Mainstem Eel River)	(Functionally Independent)
Mattole River	Functionally Independent

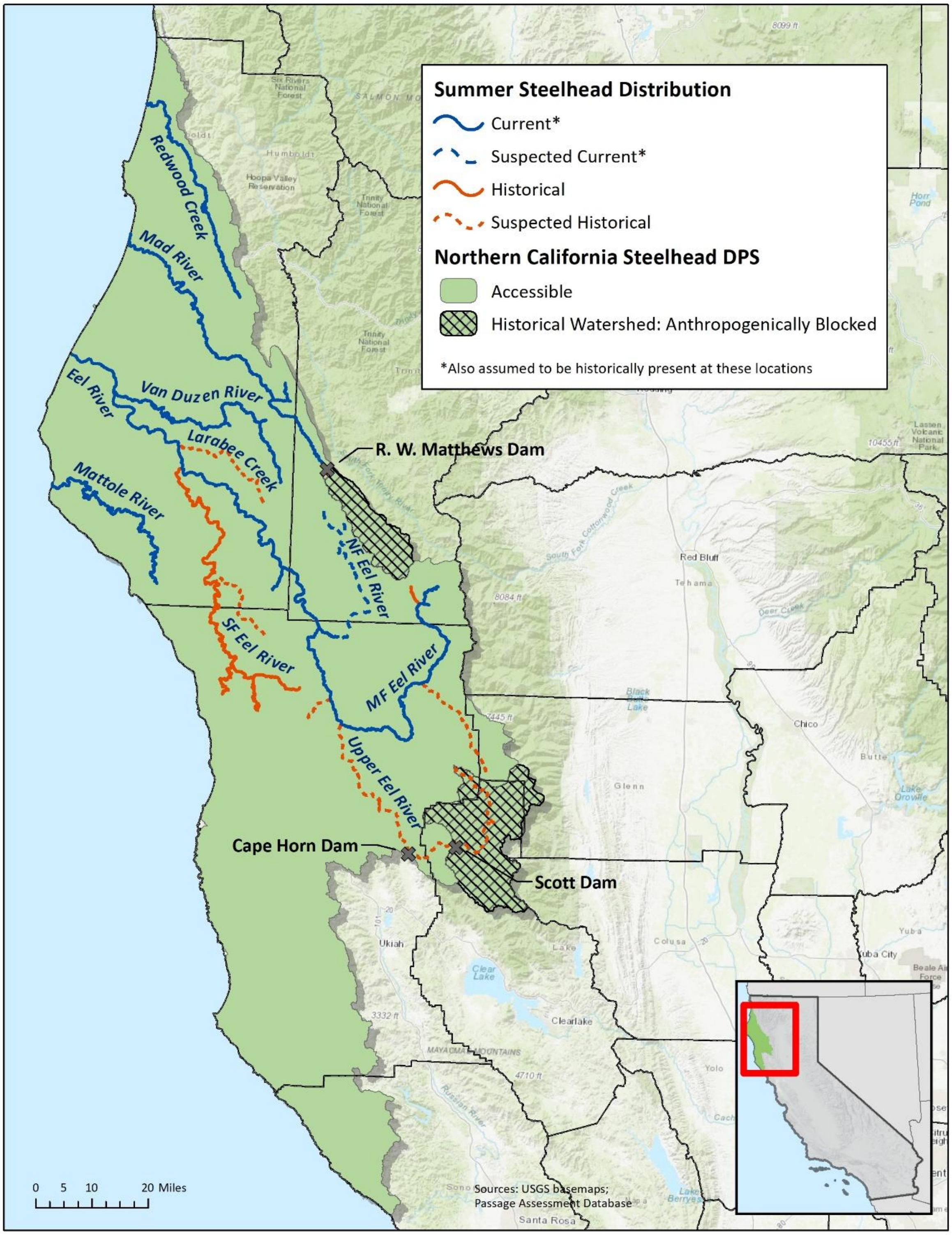


Figure 3.1. *Map of historical and current distribution of NC summer steelhead. (Note: There is likely additional tributary usage by summer steelhead that is unknown. Current surveys focus on mainstem habitat and include little to no tributary monitoring.)*

3.3.1 Redwood Creek

The Redwood Creek watershed is in northern Humboldt County and drains an area of 285 square miles. The mainstem encompasses approximately 65 river miles from Board Camp Mountain to the Pacific Ocean near Orick, CA. Prairie Creek is the largest tributary in the Redwood Creek system, entering Redwood Creek about three miles upstream from the mouth (Figure 3.2). Although winter steelhead are productive in this tributary, it is not used by summer steelhead (CDFG 2006b). Lower Redwood Creek is bordered by public lands up to and through Redwood National Park (RNP), which ends at the confluence of Coyote Creek. Almost all areas upstream of RNP on Redwood Creek are privately owned (CDFG 2006b).

The only permanent dams in the Redwood Creek watershed are the Prairie Creek Hatchery dams on Lost Man Creek (Brown 1988). The mainstem is mostly free flowing except for two temporary gravel dams that were raised each summer to impound water for recreation (CDFG 2006b). Chezem Dam was the larger of the two dams and was located below Captain Creek, while Burton Dam was located downstream between Lacks and Stover creeks (CDFG 2006b). These recreational impoundments likely blocked upstream movement of summer steelhead, which were observed using the pools to hold over summer (Brown 1988; CDFG 2006b). Both seasonal dams have been removed and are no longer annually constructed.

Very little is known about historical numbers of summer steelhead in Redwood Creek. USFWS estimated a steelhead run of about 10,000 based on extrapolations of data from similarly sized streams with like characteristics, though they did not report summer and winter runs separately (USFWS 1960 as cited in CDFG 2006b). Some limited historical information comes from anecdotal accounts from residents of the watershed and local articles from magazines and newspapers, though none of these provide a total estimate of abundance. An article in *American Angler* in 1920 contained an account of summer steelhead in Redwood Creek stating there were ten to twenty-five in each pool between 20 and 36 inches in length (Blackwell 1920). This description contrasts with more recent observations of a maximum of four to five summer steelhead per pool in that same area. In 1994, a local resident, Bill Stover, indicated that there used to be 40 – 50 summer steelhead per pool, but currently only a handful in some pools (CDFG 2006b). Local resident, Joe Massei, whose family owned property in the upper watershed at Ayres Cabin since the early 1940s, remembers seeing holes with 30 – 40 fish in April and May (CDFG 2006b). Given these accounts, it is likely that summer steelhead were historically much more abundant than they are now, though concrete numbers cannot be compared.

Since the mid-1900s, catastrophic flood events, timber harvest practices, and road construction have had a combined negative effect on erosion of already steep and highly erodible hill slopes in the watershed (CDFG 2006b). These anthropogenic effects and natural disasters, especially the floods of 1955 and 1964, have caused mass wasting and geomorphic changes, most notably aggradation of the stream channel and sediment filling in crucial summer steelhead holding pools (CDFG 2006b). In a 1966 Department survey of Redwood Creek, it was noted that the entire channel was extremely degraded and provided insufficient nursery habitat for young salmon and steelhead (Fisk et al. 1996 as cited in CDFG 2006b). In 1972, the Department surveyed the entire length of the Redwood Creek mainstem and stated that pools were essentially absent due to extreme sediment filling with a riffle to pool ratio of about 50 to one (CDFG 2006b). In the middle and lower sections there were multiple miles without any salmonids present. The upper section held small numbers of juvenile salmon, but the uppermost 10 miles of the mainstem were inundated with logging debris, rubble, and sediment. A 1977 USGS survey of 16 miles of Redwood Creek from the mouth to the gorge area above Slide Creek did not observe any pools greater than three feet in depth, though numbers of pools over three feet deep have increased slightly since then. Sediment flushing from the upper watershed to the lower watershed has improved habitat condition in the upper reaches, although to the detriment of downstream areas (CDFG 2006b). Following the catastrophic flood events in the 1950s and 1960s that raised the streambed as much as 30 feet in some areas (Redwood Creek Landowners Association 2000). Bed elevation has recovered in upper and middle areas of Redwood Creek, which have experienced degradation since these large floods, but other areas continue to aggrade particularly in the lower watershed (Madej and Ozaki 1996).

Following reports of increased summer steelhead numbers resulting from improved habitat condition, the National Park Service (NPS) implemented annual adult summer steelhead snorkel surveys beginning with a 32-mile section in 1981 from Hayes Creek upstream to about 1/8 mile above Beaver Creek (NPS 1981 – 2018). The NPS continued to survey the lower half of Redwood Creek annually beginning at Lacks Creek at Stover Ranch and proceeding downstream through Green Diamond Resource Company (GDRCO [formerly Simpson Timber Company]) lands into RNP to a location about 0.75 miles below the confluence of Hayes Creek. This survey area encompassed about 26 miles (38% of total mainstem). Originally, the survey was from Lacks Creek to Tom McDonald Creek, but was expanded to downstream of Hayes Creek in 1992. Upstream of Lacks Creek, within private landowner jurisdiction, some voluntary dive surveys were conducted during the 1990s. During survey years 1993 – 1997 and 2008, California Trout, Inc. (CalTrout) surveyed an area above the NPS survey reach from Chezem Road Bridge down to Stover Creek. During the years 1993 – 1998, an organization called North Coast Fisheries Restoration (NCFR) surveyed variable stream reaches above Chezem Road Bridge. NCFR surveys began at Ayres Cabin in 1993 and started even further upstream at Bradford Creek for all subsequent years through 1998. During this time, numbers of summer steelhead above Lacks Creek were greater than those below (NPS 1981 – 2018).

The NPS established an index reach of about 16 miles from Lacks Creek down to Tom McDonald Creek that has been surveyed annually since 1981 (NPS 1981 – 2018). A maximum of 44 summer steelhead have been observed in this reach. Total numbers accounting for all survey efforts in a year have not exceeded 59 summer steelhead. It is important to note that surveys on Redwood Creek have not resulted in summer steelhead total population estimates, but rather census counts for specific stream sections. We would also like to note that the criteria for identifying adult steelhead changed in 2006 from >16.5 inches to >16.0 inches, though this minor adjustment likely did not have a significant effect on accounting of summer steelhead. See *Appendix C* for numbers of summer steelhead from 1981 through 2019 within the index survey reach from Lacks Creek to Tom McDonald Creek and within the total survey area, annual survey dates, and annual survey reaches.

NPS summer steelhead surveys have shown that fish generally congregate in pools at or near the confluence of tributaries. This is likely due to cooler water temperatures from tributary outflow (NPS 1981 – 2018). One place summer steelhead have commonly been found is the pool at the confluence of Devils Creek, which may indicate their preference for lower water temperature over pool cover since Devils Creek pool has only adequate canopy cover. Fish were generally found in higher numbers in the middle or upper portions of the NPS survey above the gorge area due to better habitat quality. The lower half of the survey area was mostly characterized by limited canopy cover, low gradient riffles, and large exposed gravel bars causing the water to reach higher temperatures (NPS 1981 – 2018). One noteworthy exception to this trend was in 2013 when most fish were observed in the lower reaches of the survey area below Tom McDonald Creek. During this year fish were also mostly found in groups, whereas in previous years fish were generally observed as solitary (NPS 2013). These deviations from the norm could be effects of the California drought, limiting holding habitat to the lower mainstem due to low flows.

Half-pounders have also been documented and enumerated most years in Redwood Creek since NPS surveys began. Other surveys during the 1990s documented half-pounders, though their length criteria were inconsistent between years and different from NPS half-pounder criteria. NPS defined half-pounders as fish less than 12 inches in length, whereas CalTrout and NCFR defined them as between 12 and 16 inches in length.

The limit of anadromy in Redwood Creek is a 45-foot cascade above Snow Camp Creek, about 60 miles upstream of the mouth, though summer steelhead have not been seen holding anywhere upstream of Bradford Creek, which is about 4.5 miles downstream of the cascade (CDFG 2006b). There are 15 pools used by summer steelhead between Hayes and Lacks creeks and 20 pools between Lacks and Bradford creeks (CDFG 2006b). Summer steelhead are thought to spawn in the mainstem of Redwood Creek and in downstream areas of Coyote, Panther, Garrett, Lacks, Mill, Molasses, Minor, Sweathouse, Captain, Lupton, Noisy, Minon, and Bradford creeks (Van Kirk 1994 as cited in CDFG 2006b).

Figure 3.2. *Reference map of locations in the Redwood Creek watershed.*

3.3.2 Mad River

The Mad River originates at Horsehead Ridge near the Yolla Bolly-Middle Eel Wilderness and flows northwest through southern Trinity County into Humboldt County. The watershed drains an area of about 500 square miles, emptying into the ocean in McKinleyville, CA. The upstream half of the drainage is surrounded by Six Rivers National Forest, United States Forest Service (USFS) land, and the lower half is mostly owned by private ranches and timber companies, namely GDRCO (CDFG 2006a). The Mad River riparian canopy is comprised mostly of Douglas fir with a portion of redwood forest, and grass lands and oak woodlands in upland zones and some riparian areas, though much of the canopy has been removed by logging operations. The watershed is characterized by unstable geology that has caused frequent landslides and extensive sediment production (CDFG 2006a).

Two major dams have been constructed on the mainstem Mad River. Sweasey Dam was erected in 1938 and furnished with a fish ladder to allow upstream fish passage (Figure 3.3) (CDFG 2006a). The fish ladder was destroyed during the flood of 1964 (CDFG 2006a). The 45-foot dam was later blasted with dynamite and removed in 1970 (Stillwater Sciences 2010). In 1961, the 150-foot R.W. Matthews Dam was built about 60 river miles upstream (CDFG 2006a; Stillwater Sciences 2010). Matthews Dam, previously known as Ruth Dam, created 49,000-acre-foot Ruth Reservoir and is impassable to fish (Stillwater Sciences 2010). There are also natural barriers that impede fish passage on Mad River below Matthews Dam. Just below Bug Creek there is a section of roughs extending for about two miles that some steelhead can navigate but that salmon are unable to pass. These roughs end with a 25-foot falls that used to be passable by steelhead during higher flows (Bailey 1952 as cited in CDFG 2006a). These falls are well known and have been referred to by a few different names including Mad River Falls, Leadstone Falls, and the Bug Creek fish barrier (CDFG 2006a). Around the same time that Sweasey Dam was constructed, a boulder shifted within the falls and made it much more difficult for steelhead to jump over (Murphy 1950). This obstruction was blasted by the Department in 1980 to improve fish passage (CDFG 2006a). Migration was obstructed again in 1995 and summer steelhead numbers began to decline over the next few years. There is less suitable rearing habitat below the falls than above, which likely contributed to the decline in abundance (CDFG 2006a). Roughs just below Simpson Creek may also act as a flow-dependent barrier (J. Pounds, Blue Lake Rancheria and Mad River Alliance, pers. comm., April 6, 2020).

There are multiple anecdotal accounts of summer steelhead historical presence in the Mad River. Accounts from the 1930s of summer steelhead in the Bug Creek roughs came from Chet Schartzkopf, an outdoor writer for the *Humboldt Standard*. Several long-term residents talked about numerous summer steelhead they would catch both above and below the falls below Bug Creek (Bailey 1952 as cited in CDFG 2006a). A resident who grew up near the Bug Creek fish barrier thought that there were only about a quarter of the number of summer steelhead as there were before Sweasey Dam was built (Murphy 1950). Anecdotal information indicates there may have been up to 500 – 1,000 summer steelhead adults in the Mad River during the early 1900s, however, true numbers cannot be determined (CDFG 2006a). Sweasey Dam ladder counts from 1938 to 1962 suggest summer steelhead numbers were between 200 and 500 adults, a decrease from the decades before, but still somewhat abundant. Opportunistic observations in the 1960s and 1970s, including those made by Department staff, illustrate another decline to less than 50 – 100 summer steelhead adults. Likely causes for this decrease were cumulative effects of the 1964 storm and logging operations, which resulted in decimation of steelhead habitat (CDFG 2006a).

Summer steelhead surveys have been conducted in the Mad River since 1980 by the USFS Mad River Ranger District. Their survey reach fell within the bounds of Six Rivers National Forest and included areas from Matthews Dam downstream to Deer Creek, though not every section was surveyed annually. The Ranger District surveyed a 10.8-mile index reach annually until 1998 from Anderson Ford down to Deer Creek, which was thought to be the most productive reach in the survey area (CDFG 2006a). Surveys were terminated following 1998 due to the very low numbers of steelhead seen above Bug Creek (CDFG 2006a). In 1994, GDRCO and CalTrout began surveying from Deer Creek down to the Mad River Hatchery, and the Department began surveying from Mad River Hatchery down to Kadle Hole. Numbers recorded above Bug Creek indicated extremely low abundance. From 1994 through 2005 and in 2008, an additional 40 miles of river below Bug Creek were surveyed revealing an area with relatively greater numbers of summer steelhead. All surveys discontinued from 2006 – 2007 and then again from 2009 –

2012. In 2008, only the GDRCO/CalTrout area was surveyed. The Department began surveying again in 2013 from R.W. Matthews Dam all the way down to Kadle Hole, though not every reach is surveyed each year. Since 2013, numbers have ranged from 117 to 336 summer steelhead adults and represent the most consistent and reliable data within the Mad River summer steelhead time series (M. Sparkman, CDFW, pers. comm., February 7, 2020).

The “hotspot” for summer steelhead holding habitat has generally been in the nine miles of river between Deer Creek and Cowan Creek (CDFW 2019; M. Sparkman, CDFW, pers. comm., February 7, 2020). This reach was not surveyed between 1980 and 1993, with the exception of some portions in 1982 and 1991, which could explain why numbers were much lower in those years. Additionally, the Bug Creek roughs, depending on water year, are sometimes impassable to summer steelhead during their time of migration in the spring and summer. As mentioned earlier, the Department considers most of the prime juvenile rearing and adult holding habitat for steelhead to be above the Bug Creek roughs, and generally fish are able to pass through them during higher winter flows, allowing their juvenile offspring to rear in the reaches above. There is general concern, however, that the Bug Creek roughs may become impassable year-round, which would have a detrimental effect on production of both summer and winter steelhead in the Mad River (CDFG 2006a). Large schools of summer steelhead were also historically counted in Renfroe Hole as part of earlier surveys, however, since the Mad River Alliance reinitiated surveys in 2013, this area has not been consistently accessible (J. Pounds, Blue Lake Rancheria and Mad River Alliance, pers. comm., April 10, 2020).

In addition to mainstem habitat, Pilot Creek, which is upstream of Deer Creek, is known to be one of the larger tributaries used by summer steelhead (CDFG 2006a). An assessment of the Mad River watershed by Stillwater Sciences stated that a barrier near Deer Creek is impassable at all but the highest flows, but that steelhead are occasionally found farther upstream in Pilot Creek (Stillwater Sciences 2010). The Mad River is different from typical summer steelhead streams in that its flows are mainly dependent on rainfall rather than snowmelt. Snowmelt, and less occasionally rain on snow, does occur. Because the height of seasonal flows occurs earlier than in snowmelt-driven streams, summer steelhead may not always be able to pass flow-dependent barriers and ascend further upstream than winter steelhead (M. Sparkman, CDFW, pers. comm., February 7, 2020). As a result, summer and winter steelhead spawning habitat may overlap to a higher degree than in other summer steelhead streams.

Mad River Salmon and Steelhead Hatchery (MRH) was built about 15 miles upstream from the river mouth to produce salmon and steelhead as a supplement to natural production in the Mad River as well as to provide juvenile fish for releases in other rivers (CDFG 2006a). Additionally, MRH raised Rainbow Trout for fisheries in local lakes (Stillwater Sciences 2010; CDFW 2016). This is the only hatchery that has produced summer steelhead in California. Operations commenced in 1971 with summer steelhead eggs obtained from Skamania Hatchery on the Washougal River in the State of Washington. Yearling smolts of the Washougal strain were planted in the Mad River in the spring of 1972 along with Eel River stock summer steelhead smolts from Trinity River Hatchery (CDFG 2006a; CDFW 2016). Originally, the intent of this program was to maintain a run of summer steelhead through an artificial propagation program at MRH in anticipation of the Butler Valley Project, which would dam the river about 13 miles upstream of the hatchery (CDFG 2006a). The Butler Valley Project was never carried out due to local opposition and lack of need. Production of summer steelhead continued at MRH nonetheless using eggs from returning adults, although they never achieved the success intended by the program. Annual returns were minimal, numbering about 100 – 200 fish (CDFG 2006a), and ultimately the program was discontinued after the 1996 release as it did not provide sufficient angling opportunity (CDFW 2016).

Figure 3.3. *Reference map of locations in the Mad River watershed.*

3.3.3 Eel River

The Eel River watershed is one of the largest river systems in California. The Eel River mouth is located a little over 13 miles south of Eureka, California, and drains an area of about 3,684 square miles making it the third largest river system in California (CDFG 1997a). There are two major dams on the upper mainstem Eel River, Cape Horn Dam located about 156 miles upstream of the mouth and Scott Dam located 12 miles further upstream. These dams are the main components in the hydroelectric Potter Valley Project (PVP) operated by Pacific Gas & Electric Company (PG&E) (Becker and Reining 2009). There is water transport from the upper

Eel River to Lake Mendocino for agricultural and municipal use. Cape Horn and Scott dams were built in 1908 and 1921, respectively. Cape Horn Dam is equipped with a fish ladder, allowing steelhead access to Van Arsdale Reservoir and stream sections below Scott Dam, which forms Lake Pillsbury and blocks up to 288 miles of upstream habitat to anadromous steelhead (Cooper 2020). Van Arsdale Fish Station is located at Cape Horn Dam and is operated to enumerate salmon and steelhead migrating upstream (Becker and Reining 2009).

Though there are no defined populations on the lower mainstem of the Eel River, there have been summer steelhead sighted in this area in some years and upstream migrating summer steelhead need to migrate through lower sections of the river to reach the upper Eel River. Despite extensive sediment problems in the lower river, there is some juvenile rearing habitat and adult holding habitat upstream of Rio Dell, California (Jones 1992; CDFG 1997a). Summer steelhead were also seen by local landowners in Woodman Creek, which is a tributary entering the Eel River around river mile 114, prior to the formation of a barrier in 1964 (CDFG 1981 as cited in Becker and Reining 2009; Jones 1992), though this barrier was modified with explosives in 1984 and again in 2018 or 2019 to improve fish passage (S. Thompson, CDFW, pers. comm., July 8, 2020). No summer steelhead have been observed in Woodman Creek in recent history (Jones 1992), though there have not been any efforts to find them, so it is possible some summer steelhead remain. This may also be the case for the middle mainstem Eel River (S. Thompson, CDFW, pers. comm., July 8, 2020). There were 6 – 10 summer steelhead sighted in the upper Caniveri Pool below the confluence of the Van Duzen River on August 19, 1977 (CDFG 1977). There was also documentation of adult summer steelhead holding on the mainstem Eel River near Spy Rock, about 17 miles downstream of Dos Rios, in the summer of 1994 (CDFG 1994).

3.3.3.1 North Fork Eel River

The North Fork Eel River branches off from the mainstem Eel River at river mile 96.4 and drains an area of 286 square miles (CDFG 1997a). There are no manmade barriers on this river, however, a large boulder deposited in the channel by a landslide has been at least a partial migration impediment over the years. This barrier, called Split Rock, is located about 5 miles upstream from the confluence with the mainstem Eel River and is thought to restrict anadromous fish to the lower river sections especially during low flow seasons (Becker and Reining 2009; NMFS 2016b).

The North Fork Eel River has high temperatures, especially during the summer and fall, and the limited number of salmonids encountered have generally been found only in cold water refugia such as at the bottom of thermally stratified pools or areas with cooler tributary inflows (Becker and Reining 2009). There were a few surveys during the 1970s and 1980s that reported some adequate spawning habitat for winter and spring seasons, but even these areas have had very limited use by salmonids (Becker and Reining 2009). In general, habitat conditions and steelhead abundances have steadily declined since the beginning of the 20th century, especially as a result of human land and water uses, as well as the severe floods in 1955 and 1964 (CDFG 1997a).

There is very little documentation of summer steelhead in the North Fork Eel River and no quantitative estimates of historical abundance exist. There is a slightly more prominent winter steelhead run in this river, however it has declined dramatically since human settlement of the watershed (NMFS 2016b). A few sporadic surveys have been conducted over the past five decades enumerating only a handful of steelhead, though the run types have not always been specified, and conducting surveys during winter months can be difficult due to stream discharge and high turbidities.

The earliest documented report of summer steelhead in the North Fork Eel River was above Soldiers Basin in the 1920s (Jones 1992). Summer steelhead were known to hold in the North Fork Eel River below Hulls Creek during the late 1950s and early 1960s according to Bill Townsend of Ukiah and his fishing partner, Tony Garhart, who regularly fished for summer steelhead in late August to early September. Townsend recounted one instance when there were 50 summer steelhead in a deep, clear pool below Hulls Creek (Jones 1992). Jim Gilman, CDFG, reported seeing summer steelhead in the headwaters of the North Fork Eel River during his childhood (Jones 2000). Following the great flood of 1964, all deep pools used by fish for holding were filled with sediment, gravel, and boulders, greatly decreasing habitat suitability. The 1964 storm was also what created the barrier at Split Rock (Jones 1992).

Sporadic surveys have been conducted since the 1964 flood. In 1968 a survey of 32 miles of the North Fork Eel River documented steelhead presence in the lower and upper reaches, but not in a long section of the middle river (CDFG 1968 as cited in Becker and Reining 2009). This survey did not specify whether these fish were summer steelhead or winter steelhead. Isolated observations of summer steelhead have occurred in more recent decades. A multi-agency cooperative effort to survey the North Fork Eel River for Chinook Salmon, Sacramento Pikeminnow (*Ptychocheilus grandis*), and steelhead, occurred in May 2002 (BLM 2002). The survey covered about 7.7 miles of the stream and two summer steelhead adults were encountered. A few additional steelhead adults and juveniles, as well as 12 steelhead redds, were also documented (BLM 2002). In 2019, the Bureau of Land Management (BLM) organized a snorkel survey of 34 miles of the North Fork Eel River mainstem and tributaries above Split Rock barrier to document the distribution of Sacramento Pikeminnow. Their efforts resulted in observations of two adult summer steelhead as well as a notable assemblage of about 150 steelhead 8-14 inches long in one pool fed by a cold-water seep. They also found upwards of 3,000 juvenile steelhead throughout the mainstem and in Salt Creek (BLM 2019). These findings support the idea that Split Rock may not be a complete barrier to anadromy. A four-year study of the Eel River watershed, which was completed by University of California at Davis in 1991, reported that the barrier had been modified to improve fish passage upstream (Becker and Reining 2009). There are also some historical reports of summer steelhead and Chinook Salmon above the Split Rock barrier, which was classified in the Department's Eel River Salmon and Steelhead Restoration Action Plan (CDFG 1997a) as a barrier to winter steelhead and Sacramento Pikeminnow. Split Rock is also likely a barrier to Chinook Salmon (USDA and USDI 1996).

Given the lack of information on summer steelhead in the North Fork Eel River, even in the form of anecdotal reports, consensus has been that the run is effectively extirpated or severely depleted (Puckett 1975; Spence et al. 2008; Becker and Reining 2009; NMFS 2016b). However, scientific studies designed to enumerate possible summer steelhead abundances are warranted.

3.3.3.2 Middle Fork Eel River

The Middle Fork Eel River is the largest sub-basin of the Eel River watershed and currently supports the most robust population of summer steelhead within the NC steelhead DPS. The headwaters are in the Yolla Bolly Wilderness Area at about 6,000 feet of elevation. The Middle Fork Eel River drains an area of about 750 square miles of mountainous terrain and intersects with the mainstem Eel River near the town of Dos Rios around river mile 119.3 (Jones 1980; Becker and Reining 2009). The watershed is characterized by naturally unstable topography. Previously high levels of livestock grazing and timber harvest operations have exacerbated these issues especially as they relate to sediment deposition (Jones 1980; CDFG 1998).

There are no manmade dams on the Middle Fork Eel River; however, there is a regional zone containing a series of eight sizable roughs caused by landslides that occupy about 3.4 miles of channel (Jones 1980). Some of the roughs have acted as barriers to upstream migration including those at Maple Creek, Asa Bean Crossing, Hoxie Crossing, and Pothole Creek, though the roughs near Pothole Creek were blasted in 1966 and are no longer a barrier (Jones 1980; S. Thompson, CDFW, pers. comm., July 8, 2020). The Osborn Roughs area near Fly Creek have also been an impediment to summer steelhead migration in some years (Figure 3.4). Both Asa Bean and Osborn Roughs have been blasted with dynamite multiple times in the past to allow for upstream movement of anadromous fish. Asa Bean Roughs continues to be an obstacle for summer steelhead, especially since 1996 when boulders within the roughs naturally rearranged and many jump pools utilized by summer steelhead were eliminated (CDFG 1966 – 2018).

Historical information on summer steelhead in the Middle Fork Eel River has been sourced from angler stories, accounts given by local watershed residents, and occasional studies. CDFG (1997a) states that steelhead populations in the Middle Fork Eel River have steadily declined since 1900 as a result of the major flood events as well as increased sedimentation, cattle grazing, logging, and poaching (CDFG 1997a). A 1920 volume of the California Fish and Game journal documented summer steelhead on the North Fork of the Middle Fork Eel River that held all summer in large, deep holes, undisturbed (CDFG 1920). Weldon Jones describes summer steelhead encounters dating back to 1929 in his 1992 report on Eel River summer steelhead historical distribution (Jones 1992). Anecdotal reports of summer steelhead sightings have led to speculations of about 6,000 summer steelhead in the canyon between Asa Bean and Hellhole during the 1930s, and about 3,500 in the canyon above the Eel River Ranger Station during the 1940s (Jones 1992). These counts were made by members of the public and are unconfirmed but provide some gage of population size. In the early 1950s, a widely recognized fishery developed during the spring steelhead migration period as fish moved up into the river system to hold over summer, though angler catch and effort declined in the latter part of the decade (Jones 1992). A 1957 CDFG and USFS joint survey report characterized the Middle Fork Eel River as an important spawning and rearing area for both summer and winter steelhead (CDFG 1957b as cited in Becker and Reining 2009). This survey estimated 360 to 420 summer steelhead adults between Asa Bean and the Eel River Ranger Station (Jones 1992).

In 1973, a study on water temperature conditions in the Eel River system found abundant juvenile *O. mykiss* in the Middle Fork Eel River from Osborn Station to Buck Creek. Additionally, 24 adult summer steelhead were observed between Osborn Station and Hellhole Canyon (Becker and Reining 2009). USFS surveyed 12 miles of the upper river in 1979 and noted that steelhead migration was impeded by a barrier upstream of Balm of Gilead Creek, above which there was mostly dry streambed with high amounts of sediment (Becker and Reining 2009).

The catastrophic flood of 1955 had major effects on fish habitat, most notably erosion and subsequent filling of pools used by summer steelhead for holding. The 1964 flood perpetuated the destruction of habitat, wiping out nearly all summer holding areas due to in-filling of sediments (Jones 1992). In June 1957, 20 adult steelhead were observed in a large, deep pool about a third of a mile below the mouth of the North Fork Middle Fork Eel River. This pool was documented to be at least 30 feet deep at the time (CDFG 1957a). This same area is now surveyed annually as part of the summer steelhead dive surveys on the Middle Fork Eel River and pools have averaged only 11 feet in depth over the past 20 years (S. Thompson, CDFW, pers. comm. August 4, 2020). The Department closed the Middle Fork Eel River to summer steelhead fishing in response to the detrimental effects of these floods. Prior to the 1955 and 1964 floods, it is likely that summer steelhead refugia extended much further downstream of the confluence of the Black Butte River. Historical photos and old anecdotal accounts support this belief (CDFG 1966 – 2018; Becker and Reining 2009). It is also probable that the Black Butte River historically supported summer steelhead. Jones (1992) states that hunters caught summer steelhead in the upper drainage during the 1930s. A 2006 USFS report documented fire crew observations of 18 to 24-inch trout in large, deep pools in the Black Butte River that may have been summer steelhead (Becker and Reining 2009); however, partial surveys conducted by the Department in the summers of 2013 and 2019 did not find any summer steelhead holding in the Black Butte River (S. Thompson, CDFW, pers. comm., August 6, 2020). Anadromous habitat in the Black Butte is accessible to all steelhead run types, though this may not have always been the case. Prior to historic flood events there may have been deeper pools and cascades that provided a greater advantage to summer steelhead over winter steelhead.

Presently, there are no flow-dependent barriers that would impede winter steelhead migration except for one near Jumpoff Creek that may act as a barrier only during very low flows (S. Thompson, CDFW, pers. comm., August 6, 2020). Currently, pools are quite shallow, ranging from three to six feet in depth (CDFW 2013), which is not ideal habitat for adult summer holding.

The difference between present and pre-1955/1964 spawning and rearing habitat availability in terms of total river miles is considerable. Accounting for mainstem habitat, from the confluence of the Middle Fork Eel River with the mainstem Eel River to the current upper limits of anadromy, and counting the Black Butte River as a possible historical summer steelhead stream, about 62 miles were lost from the historic 85 miles. That leaves only about 23 miles of spawning and rearing habitat, or approximately 27% of the historical habitat available (CDFG 1966 – 2018).

Annual summer steelhead snorkel surveys conducted by CDFG and USFS began in 1966 and continued through 1999. CDFG took sole responsibility for surveys from 2000 to the present. The only year in which a survey was not conducted was 1969, and in 1972 only a partial survey was completed. All other years of the survey have consistently covered about 27 river miles including the mainstem from Bar Creek up to Wrights Valley, the North Fork of the Middle Fork Eel River up to the confluence with Willow Creek, and Balm of Gilead Creek up to the natural falls that are considered a barrier to anadromy (S. Thompson, CDFW, pers. comm., February 21, 2020). During the years that the Asa Bean roughs acts as a passage barrier, the survey is reduced to about 19.4 miles total. In the 1970s, surveyors found steelhead on the mainstem above Wrights Valley up to Uhl Creek, about five miles upstream. However, following some shifting of material in a waterfall that was previously passable to fish and seeing zero fish upstream for multiple years, the survey extent was changed to terminate at Wrights Valley in the early 1980s (S. Thompson, CDFW, pers. comm., February 21, 2020). Due to this natural shift in summer steelhead distribution, it is believed that curtailment of the survey to Wrights Valley has not resulted in underestimating summer steelhead abundance. Subsequently, the falls near Wrights Valley have been considered the upper extent of anadromy since this time. Three sequential waterfalls form a complete barrier to anadromy on Balm of Gilead Creek, and a waterfall at the confluence of Willow Creek is the uppermost extent of anadromy on the North Fork of the Middle Fork Eel River (S. Thompson, CDFW, pers. comm., April 7, 2020).

Counts of summer steelhead in the Middle Fork Eel river have ranged from 198 – 1,601 with the lowest abundances occurring in 1966 and 1967, which were the first two years of the survey. Following the 1995 survey, the boulders within Asa Bean Roughs rearranged, severely limiting fish passage upstream of the roughs (CDFG 1966 – 2018). Jump pools were filled in and pathways through the boulders were blocked with sediment. In the following years, there seemed to be a positive correlation between the number of adults able to pass Asa Bean and the number of adults returning from that brood in subsequent years. Asa Bean Roughs generally dries out during summer months and any fish holding within the reach perish before they can spawn (CDFG 1966 – 2018). In the most recent years of the survey, Asa Bean has been a complete barrier to summer steelhead. With Asa Bean blocking upstream migration, over 12 miles of prime spawning habitat are inaccessible and summer steelhead are forced to distribute further downstream where the habitat is less suitable for spawning and rearing (CDFG 1966 – 2018).

Presently, the majority of summer steelhead encountered in Department surveys have been between Asa Bean Roughs and Osborn Roughs below Fly Creek. Summer steelhead are generally not found downstream of Fly Creek or upstream of Fern Point (CDFG 1966 – 2018). A small percentage (about 5%) are found from Fly Creek down to Bar Creek (S. Thompson, CDFW, pers. comm. July 15, 2020). Osborn and Fern Point are the two sites used for juvenile standing crop electrofishing surveys that have been conducted since the 1980s. Distribution seems to be shifting lower in the system, especially as a result of the Asa Bean barrier in conjunction with recent drought conditions that have prevented fish from accessing upstream spawning grounds during low flows. Tributaries that have supported summer steelhead in the recent past include Beaver Creek, North Fork of the Middle Fork Eel River, and Balm of Gilead Creek (Becker and Reining 2009), although there could be more due to a lack of reporting of all possible tributaries. North Fork of the Middle Fork Eel River and Balm of Gilead Creek are both above Fern Point, and summer steelhead have not been found in these tributaries since the mid-1990s (CDFG 1966 – 2018). Summer steelhead were found in the lower 0.3 miles of Beaver Creek in 1995 but could not migrate further upstream due to a barrier fall (Jones 2000). Many tributaries of the Middle Fork Eel River have natural barrier falls at their confluences with the mainstem river, thus, summer steelhead generally hold in the main channel (S. Thompson, CDFW, pers. comm., April 8, 2020).

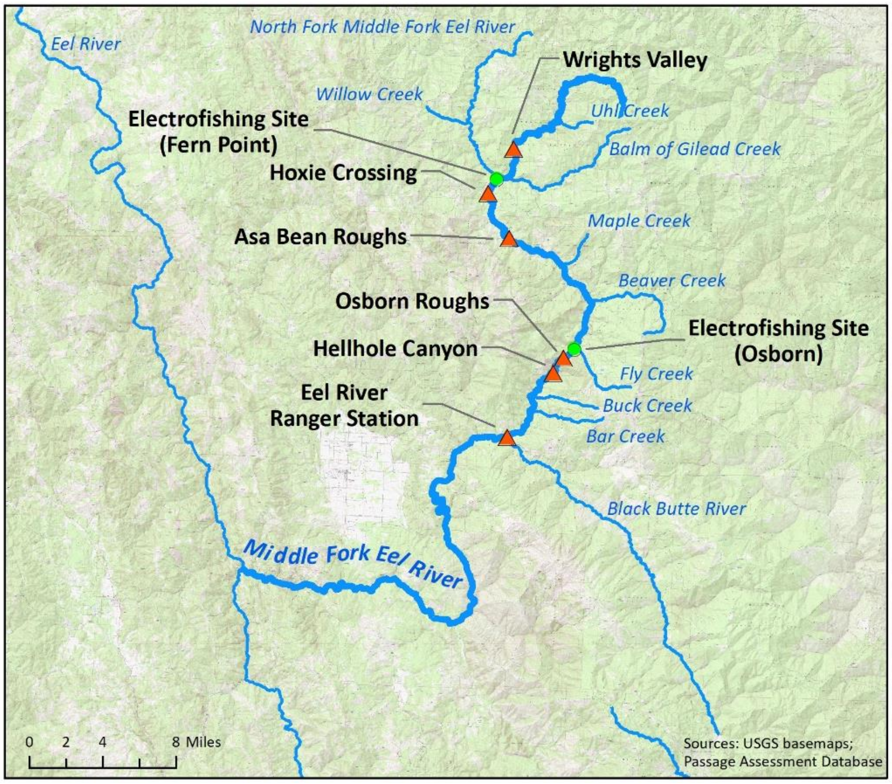


***Figure 3.4**. Reference map locations in the of Middle Fork Eel River watershed.*

3.3.3.3 South Fork Eel River

The South Fork Eel River is a major tributary to the Eel River encompassing 105 river miles and drains a watershed of about 689 square miles (CDFG 1997a). The headwaters originate south of Laytonville and flow through both Mendocino and Humboldt counties, intersecting with the mainstem Eel River at about river mile 40. Benbow Dam was built about 40 miles upstream of the confluence with the mainstem Eel River in 1931 (Figure 3.5). Benbow Dam had a fish ladder to allow upstream passage though the dam was demolished in 2017 (*Benbow Dam removal completed*, 2017 web article).

There is little documentation of summer steelhead in the South Fork Eel River. One anecdotal report was documented from Wayne Calder, who recalled seeing big silver steelhead holding in deep pools of the South Fork during June and July near Leggett in the 1930s (Jones 1992). He claims they also were in Rattlesnake Creek congregating in pools, though only a few would survive through the end of summer. Old CDFG correspondence files document a South Fork Eel River summer steelhead run through the late 1950s. Staff from Cedar Creek Hatchery fished for summer steelhead in the late 1950s and early 1960s (Jones 1992). However, Leo Shapovalov noted that the South Fork Eel River has very little snowfall feeding the river, a characteristic that is generally conditional to presence of summer steelhead (CDFG 1953). He explained that due to the lack of snowmelt and holding pools, at that time there was no summer steelhead run in the South Fork Eel River. Jones (2000) reports a personal observation from 1982 of possible summer steelhead in pools of the headwaters of the East Branch of the South Fork Eel River (Jones 2000). Counts of steelhead were made at the Benbow Dam fish ladder from 1938 to 1976, though these are assumed to only enumerate winter steelhead based upon time of capture (NMFS 2016b).

If a summer steelhead run did historically exist in the South Fork Eel River, it probably would not have been large. Primary summer steelhead holding habitat was thought to be in the canyon between the Wilderness Lodge and Rattlesnake Creek and there are still occasional stories of summer steelhead appearing in the canyon (Jones 1992). In their 2016 Multispecies Recovery Plan, NMFS stated that summer steelhead might be extirpated from the South Fork Eel River (NMFS 2016b). Future studies could confirm this assertion.

Figure 3.5. *Reference map of locations in the South Fork Eel River watershed.*

3.3.3.4 Upper Mainstem Eel River and Upper Middle Mainstem Eel River

The upper mainstem Eel River encompasses the area from the confluence of Soda Creek, about 1.3 miles downstream of Scott Dam, up to and including the Lake Pillsbury sub-basin and its tributaries (NMFS 2016b). Though there is suitable steelhead habitat in the tributaries of the Lake Pillsbury sub-basin, only resident Rainbow Trout exist in these streams since Scott Dam is a total barrier to anadromy. Passage of summer steelhead above Van Arsdale Fish Station is limited due to the closure of the station at the end of the winter steelhead run. Most summer steelhead, however, were probably lost with the construction of Scott Dam in 1921 (NMFS

2016b). Counts of steelhead have occurred at Van Arsdale since 1933, but only during the winter steelhead run (NMFS 2016b). It is possible that summer steelhead adults use habitat between Van Arsdale Station and Scott Dam (S. Thompson, CDFW, pers. comm., July 8, 2020).

There is very little documentation of any summer steelhead in the upper middle mainstem Eel River below Soda Creek, however, some streams that drain the north side of the Eel River basin are ecologically similar to Larabee Creek and these major sub-basins may have historically supported populations of steelhead (Bjorkstedt et al. 2005). A few observations of summer steelhead in the upper mainstem Eel River above Soda Creek were made by local residents. There are multiple accounts of anglers in the area that is now Lake Pillsbury who caught large summer steelhead prior to the construction of Scott Dam. In the mid-1920s, anglers caught large summer steelhead in the pool just below Scott Dam (Jones 1992). CDFG conducted a snorkel survey in 1985 as a response to anglers catching steelhead in the summer and found 21 summer steelhead between Van Arsdale and Soda Creek, 19 of which were found in the Van Arsdale reservoir (Jones 1992). There were likely more summer steelhead than could be seen due to poor visibility. Summer steelhead may sporadically use this area for summer holding (Jones 1992).

3.3.3.5 Van Duzen River

The Van Duzen River is an Eel River tributary draining an area of about 429 square miles. The river originates in Six Rivers National Forest at an elevation of about 4,300 feet and flows northwesterly for 73 miles, meeting the mainstem Eel around river mile 13.5 near Fortuna, CA (CDFG 2006c; Becker and Reining 2009). The lower basin mostly consists of redwood forest, while the upper basin is predominantly oak-conifer woodlands and grasslands (CDFG 2006c). Gravel mining occurs in the lower basin, as well. The Van Duzen River does not have any manmade dams. A proposal to install a large dam on the upper Van Duzen River at Eaton Falls was terminated as a result of the California Wild and Scenic Rivers Act of 1972, which prohibits dam construction on a number of Northern California river basins (CDFG 2006c).

There is limited historical information about summer steelhead in the Van Duzen River. The Department estimated that there used to be 2,000 summer steelhead adults (CDFG 1966 as cited in Jones 1992), though this was likely an overestimate given that pre-1964 counts in the Middle Fork Eel River, which contains double the amount of holding and rearing habitat as the Van Duzen River, numbered around 2,000 fish (CDFG 2006c). The Mad River has similar holding and rearing habitat availability as the Van Duzen River and had a pre-flood estimate of about 500 summer steelhead (USACOE 1973 as cited in CDFG 2006c). The Van Duzen River summer steelhead run likely did not exceed 1,000 fish historically and was probably closer to the pre-flood Mad River estimate of 500 fish (CDFG 2006c).

An outdoor writer for the *Humboldt Standard,* Chet Schwarzkopf, wrote in 1941 that “hundreds” of spring (summer) run steelhead congregated in deep holes of Eaton Roughs in the

Van Duzen River above Bridgeville (Van Kirk 1998b). Eaton Roughs is an 18-mile stretch between Eaton Falls, at river mile 46 near the town of Dinsmore, and Bridgeville (Figure 3.6). Additional articles by Shwarzkopf from 1938 to 1941 documented angler catches of summer steelhead and observations of summer steelhead in large pools or at tributary confluences (Van Kirk 1998b). Leo Shapovalov also documented steelhead resting in the deep pools within Eaton Roughs during the summer in a letter he wrote from CDFG to the Bureau of Fish Conservation on July 6, 1945 (Jones 1992). Murphy and De Witt (1951 as cited in Jones 1992) also described "spring-run steelhead" in the headwaters of the Van Duzen River. Summer steelhead are thought to have occupied the mainstem from Eaton Falls down to or past Baker Creek during their summer holding period (Jones 1992).

The Little Van Duzen River, also known as the South Fork Van Duzen River, historically provided substantial summer steelhead holding, spawning, and juvenile rearing habitat. The sub-basin contains over 30 miles of habitat in comparison with about 18 miles within the Eaton Roughs (CDFG 2006c). Schwarzkopf's 1938-1941 articles in the *Humboldt Standard* mention summer steelhead in the Little Van Duzen River and state that summer steelhead are able to get far upstream in the mainstem but choose not to (Van Kirk 1998b), though they may just not have been seen. CDFG biologist, Eric Gerstung, reported seeing summer steelhead in the Little Van Duzen River in 1960 (CDFG 1991) and estimated there to be over 100 there in the early 1960s, but the 1964 flood drastically changed the channel morphology and created migration barriers between Bridgeville and the Little Van Duzen River (Roelofs 1983). The flood inflicted much more damage to the Little Van Duzen River than to the mainstem (CDFG 2006c). Landslides, in combination with logging activity and forest roads, caused extreme aggradation of the channel and filled in all summer steelhead holding pools (Jones 1992; CDFG 2006c). During the summers of 1980 and 1982 CDFG surveyed the Little Van Duzen River and found zero summer steelhead (Roelofs 1983); however, two summer steelhead were found in the Little Van Duzen River in 1987 (CDFG 1987). In 1997, CDFG and USFS looked extensively for summer steelhead in the Little Van Duzen River and found only one (CDFG 1997b). Over the past decade, personal observations by the Department's Shaun Thompson in the Van Duzen watershed indicate that most adult holding occurs in the mainstem Van Duzen while most spawning likely occurs in the Little Van Duzen. Juvenile rearing has been observed in both the mainstem and the Little Van Duzen River (S. Thompson, CDFW, pers. comm., July 8, 2020).

There is also one historical report of summer steelhead in Yager Creek, another tributary of the Van Duzen River. In a 1917 article from the *Ferndale Enterprise*, a local newspaper from Ferndale, CA, steelhead were documented trying to make their way upstream in Yager Creek and having difficulty passing a barrier falls (Van Kirk 1998b). "Holdover steelhead" were observed by CDFG surveyors in Yager Creek during the summer of 1964 (Van Kirk 1998b).

The Van Duzen River is thought to be accessible to steelhead up to Eaton Falls. The main areas used by summer steelhead for holding are the 18-mile stretch within Eaton Roughs and the

Little Van Duzen River, which contains spawning and rearing habitat for summer steelhead, although as mentioned above, the Little Van Duzen River has had limited summer steelhead use since the 1964 flood (CDFG 2006c). Eaton Roughs is comprised of unstable topography with 58 active slides within 11 miles of the roughs. The constant shifting of material may be both a blessing and a curse to summer steelhead in that boulders and rocks often create barriers to migration, but they can also provide habitat in the form of deep, scoured out pools with cover provided by boulder ledges. Since 1974, about 86% of summer steelhead adults have assembled in the 6.8 miles between Eaton Falls and Baker Creek (CDFG 2006c). During the most recent surveys of the Van Duzen, which cover the majority of holding habitat, over 60% of summer steelhead have been found in two main pools, Big Hole and Salmon Hole (S. Thompson, CDFW, pers. comm., March 23, 2020). A cascade at Salmon Hole, about 10 miles downstream, sometimes impedes steelhead upstream migration (CDFG 2006c). A Department report from 1965 notes a barrier about 12 miles downstream from Eaton Falls that is only passable to some summer steelhead at high enough flows (Becker and Reining 2009). This barrier is likely the cascade at Salmon Hole, which is generally considered to be a migration barrier to salmon and winter steelhead (S. Thompson, CDFW, pers. comm., July 8, 2020).

It is possible that Eaton Falls is not a complete barrier to anadromy. In 2018, Department scientist, Shaun Thompson, observed one adult steelhead above Eaton Falls and in 2019 he conducted a series of snorkel surveys and found 15 fish that may have been anadromous based on their size (14-24 inches). This is not definitive evidence of anadromous fish surpassing the falls, but resident fish typically do not grow that large (S. Thompson, CDFW, pers. comm., March 23, 2020). Additionally, Kannry et al. (2020) found evidence of the anadromous genotype at the genomic region associated with the migratory life history (OMY05) in fish sampled above Eaton Falls as well as a lack of population structure between populations above and below the falls.

Substantial information on summer steelhead in the mainstem Van Duzen River has been obtained since 1967. In 1967, the run was held up for many months below the Salmon Hole barrier and 58 adults were counted in a one-mile stream reach downstream (CDFG 2006c). A weir operating at Bridgeville about five miles downstream in 1968 caught 82 adults. That same year in August, a diving survey found that some of those fish remained in the river mainly downstream of the Salmon Hole barrier. From 1964 to 1973 most steelhead were not able to pass this barrier until higher winter flows arrived. The barrier was blasted in 1973 and most holding fish relocated upstream to the stream section between Baker Creek and the South Fork Van Duzen River (CDFG 2006c).

Summer steelhead snorkel surveys were first conducted in the Van Duzen River in 1979. Until recently, most of these surveys were limited in scope and did not enumerate the entire run. The full 18-mile reach of Eaton Roughs was surveyed in 1979, 1980, 1982, 1984, 1987, and 1997 and resulted in the most dependable data. Because these full Eaton Roughs surveys indicated that low numbers of fish were holding downstream of Baker Creek and most fish upstream of Baker Creek appeared to be holding around Big Hole, the 1988, 1989, 1992, and 1995 surveys were limited to a 1.5-mile stream section between Big Hole and the South Fork Van Duzen River (CDFG 2006c). This alteration was made with the assumption that Big Hole represented most of the adult distribution in the river; however, it may have resulted in substantial under counts. For example, a 1991 spot survey of Salmon Hole, which is downstream of Big Hole, counted 31 adults in an area where assemblages of adults had not been observed since 1972 (CDFG 2006c). This irregularity may have been a result of very low spring flows that year, but this could not be verified because no counts were made above Salmon Hole to see if fish were able to pass upstream (CDFG 2006c). Big Hole can be up to 20 feet deep, depending on streamflow, and is heavily shaded, thus it is not always possible to see all fish holding in this area (CDFG 2006c). This can be a potential concern of all snorkel surveys. Additionally, adult steelhead do not necessarily hold in the same place throughout the summer. Bob Wotherspoon, the owner of the river front property at Big Hole saw varying numbers of summer steelhead between July and September of 1998 and did not believe he was double counting any of the fish (CDFG 2006c), though this assumption cannot be confirmed. If the Department had conducted a single one-day survey, depending on when the survey was conducted, the run could have either looked decent or poor because of the different numbers of fish occupying Big Hole throughout the summer.

Figure 3.6. *Reference map of locations in the Van Duzen River watershed.*

In 2011, the Department's summer dive surveys on the Van Duzen River started covering a much broader extent of summer steelhead habitat. The annual surveys now cover the reach between Little Larabee Creek and Eaton Falls (generally considered the upper extent of anadromy on the mainstem Van Duzen River), and are thought to encompass the majority of available holding pools in the river (Williams et al. 2016).

3.3.3.6 Larabee Creek

Larabee Creek is a tributary of the Eel River, flowing west and intersecting with the mainstem at river mile 36.4. Larabee Creek drains an area of 81.5 square miles (CDFG 2000 as cited in Becker and Reining 2009). All life stages of steelhead have been found in Larabee Creek and high-quality spawning habitat has been reported by Department surveys in 1938 and 1950, and a BLM survey in 1977 (Becker and Reining 2009). There are no documented accounts verifying the presence of summer steelhead in Larabee Creek historically or currently. NMFS (2016b) states that given the amount of available habitat historically within Larabee Creek, steelhead numbers (run unspecified) may have been in the thousands prior to the overfishing and habitat destruction of the 19th century. Bjorkstedt et al. (2005) mentions anecdotal indications that Larabee Creek supported historically summer steelhead based off the fact that it drains a relatively high elevation watershed that is fed by considerable snowmelt. This characteristic is more typical of summer steelhead streams within the Eel River basin, as opposed to lower elevation streams with warmer temperatures that only support winter steelhead.

3.3.4 Mattole River

The Mattole River is the southernmost summer steelhead stream within the Northern California steelhead DPS. The headwaters are located in the Chemise Mountains of the King Range National Conservation Area. The river flows northwesterly and then turns abruptly to the west at the town of Petrolia, emptying into the Pacific Ocean near Punta Gorda. The Mattole River drains an area of about 304 square miles in northern Mendocino and Humboldt counties (CDFG 2007). Coastal areas of the basin are dominated by redwood forest and much of the watershed used to be heavily wooded with Douglas fir and hardwoods like tan oak and madrone. Average annual rainfall is remarkably heavy, though the amount of precipitation can vary greatly from year to year (Downie et al. 2003). There are significant water diversions in the lower river for agricultural purposes, which have caused an increasing number of fish kills (CDFG 2007). Like the Mad River and South Fork Eel River, the Mattole River is unusual as a summer steelhead stream in that it is not driven by snowmelt (NMFS 2016b). Thus, a potentially greater spatial overlap exists between summer and winter steelhead habitat, as was suggested for Mad River steelhead.

Summer steelhead are known to have occupied the Mattole River historically. The first reported sighting of summer steelhead in the Mattole River was by Albert Etter, who was a resident of the watershed beginning in 1894 (Etter 1943 as cited in CDFG 2007). In a letter Etter wrote to the Department in 1943, he described great numbers of steelhead migrating upstream to spawn from January through April when he first settled in Ettersburg, including 300 in one pool alone. It is possible that some of these fish seen in April were early summer steelhead. Etter also mentions two of his guests observing 100-200 large trout mostly between 16 and 24 inches long that were holding in one pool near his farm in October 1942. This was similar to a sighting a few years before about three miles downstream. Etter characterized these fish as deep-bodied and healthy in contrast with how a decrepit winter steelhead would appear (Etter 1943 as cited in CDFG 2007).

Multiple articles from the *Ferndale Enterprise* in the early to mid-1900s mention fishermen targeting large steelhead during summer months, though it is not always clear if those fish were Rainbow Trout or summer steelhead (Van Kirk 1998a). A *Ferndale Enterprise* article from May 1937 described a run of steelhead that had "the appearance of fish from the sea" and the *Humboldt Standard* reported "a good run of spring steelhead" on the Mattole in late April 1939 (Van Kirk 1998a). In May 1949, an article in the *Humboldt Times* talked of "big trout in from the sea for the summer" in the pools of the upper Mattole River (Van Kirk 1998a). Articles in these local newspapers indicate that there were both resident Rainbow Trout and summer steelhead present in the Mattole at that time.

In 1980, 44 adult summer steelhead were observed by a local game warden close to the town of Ettersburg, where Albert Etter had lived. Four possible summer steelhead were also seen during the same year in Honeydew and Bear creeks, two tributaries of the Mattole River (CDFG 2007). The Department surveyed the full length of the Mattole River in 1982 and 1991. Only three fish large enough (16 inches) to be considered summer steelhead were found near Ettersburg in 1982 and zero summer steelhead were observed in 1991 (CDFG 2007).

Consistent annual summer steelhead snorkel surveys began in 1996 and have continued each year through the present day. Between 1996 and 2000, surveys were conducted by the Mattole Salmon Group in conjunction with CalTrout, AmeriCorps, the Humboldt Fish Action Council, and Petrolia School (CDFG 2007). The Department did not participate in the 1996 – 2000 surveys. Though the 1996 – 2000 surveys extended from the headwaters all the way to the mouth, only between 40% and 75% of the habitat was surveyed during these five years (CDFG 2007). Also, no tributaries of the Mattole River were surveyed. Most adult summer steelhead were found to congregate in a reach of 11.6 miles between McKee and Woods creeks (CDFG 2007). Half-pounders were also enumerated and exceeded the number of adults in all but two years (2013 and 2014). It is important to note, however, that size was the main factor used to identify half-pounders (12 – 16 inches), so it is possible that these numbers include some resident Rainbow Trout (MSG 1999 – 2016). See *Appendix D* for numbers of adult and half-pounder steelhead from 1996 to 2019.

Since 2005, the survey has covered almost the entire mainstem Mattole River. The uppermost five miles have been excluded since 2013 due to the lack of suitable adult holding habitat (N.

Queener, Mattole Salmon Group, pers. comm., February 4, 2020). A large proportion of summer steelhead have been found to hold in a 19.4-mile stretch of the upper Mattole River from McKee Creek to about a mile above Grasshopper Hill Creek (MSG 1999 – 2016). However, summer steelhead are rarely found in the headwaters. The number of river miles surveyed has varied each year likely due to fluctuations in availability of volunteers. The highest number of summer steelhead observed since the survey began in 1996 was 56 in 2013.

Figure 3.7*. Reference map of locations in the Mattole River watershed.*

3.4 Abundance and Trends

The Petitioner requested an evaluation of NC summer steelhead to evaluate potential listing under CESA. To provide the best scientific information in our evaluation of the candidate species as required by Fish and Game Code Section 2074.6, we have analyzed status and trends for NC summer steelhead populations with available annual abundance data.

Five NC steelhead DPS streams with extant summer steelhead population segments are surveyed annually to enumerate adult summer steelhead. These streams include Redwood Creek and the Mad, Middle Fork Eel, Van Duzen, and Mattole rivers. It is important to note that for all NC summer steelhead surveys, the method used has been direct observation of fish holding over summer, which likely underestimates abundance to some degree, especially when there is poor visibility, substantial cover habitat, and greater pool depth (Portt et al. 2006). These direct counts are not statistically expanded in any way to account for fish that are not observed within the survey reaches or were outside of the extent of the survey. These adult surveys are conducted months prior to spawning, so there is also potential for overestimating the number of spawning adults due to pre-spawn mortality. Thus, the counts obtained cannot be equated to spawning fish estimates, which are often used as a metric of salmonid population abundance. The analyses below should be viewed and interpreted with an understanding of the limitations surrounding these data. These analyses are merely tools to help assess and contextualize what the available data suggest in terms of general status and trends of each NC summer steelhead population. Without comprehensive monitoring data, it is not possible to produce total estimates of abundance of these summer steelhead populations.

Of the five surveys, the most robust dataset is from the Middle Fork Eel River. Snorkel survey efforts have been relatively consistent since the first year of the survey in 1966; with the reaches from Bar Creek up to Wright's Valley surveyed every year along with the North Fork of the Middle Fork Eel River and Balm of Gilead Creek (S. Thompson, CDFW, pers. comm., January 16, 2020). There were two years where the survey was not conducted or was incomplete, 1969 and 1972, respectively. Long-term trend analyses of Middle Fork Eel River summer steelhead in this report have been conducted using data from 1970 onward to account for the year of missing survey data (1969). Middle Fork Eel River summer steelhead numbers have exceeded numbers of all other surveys each year on the order of hundreds of fish.

Summer steelhead snorkel surveys conducted on Redwood Creek, Mad River, Van Duzen River, and Mattole River have had varying levels of inconsistency in the stream reaches or years surveyed. These four surveys generally did not cover the entirety of summer steelhead holding habitat, though most have improved in more recent years and cover the majority of the mainstem habitat. Most years of the Redwood Creek survey covered only an index reach of about 24 miles. Mad River surveys spanned anywhere from three and a half to 72 miles with the most recent years covering about 50 miles of habitat. Until recently, the Van Duzen River survey was also inconsistent, with only 15 years constituting what would be considered a "full" survey from the Eaton Falls barrier down to the confluence with the mainstem Eel River (S. Thompson, CDFW, pers. comm., March 23, 2020). These full survey efforts include the most recent surveys beginning in 2011.

We contemplated using index reaches that were surveyed consistently each year in trend analyses in attempts to standardize these datasets but there are several necessary considerations regarding this approach. The intent of an index reach is to represent relative annual summer steelhead abundance by selecting a reach that has consistent use by summer steelhead and is surveyed every year. Although an index reach would provide a consistent survey area across years, the index reach selected may not represent the same level of utilization by summer steelhead each year. The selected reach could be a summer steelhead hotspot one year but unutilized the next year due to shifts in distribution as a response to flows, temperature, or other environmental factors that can vary annually. Additionally, an index reach would not provide an idea of total abundance, unless the reach covered most of the known habitat. Index reach numbers could only show possible trends in population growth or decline and would need to be expanded to achieve reliable abundance estimates. Because of this, we decided to use total counts from most datasets with the caveat that these numbers likely do not represent total abundance each year. It was decided to use an index reach only for the Redwood Creek trend analysis (*Section 3.4.2 Trend Analysis*), since the annual survey included a consistent index reach of that stream. For analysis of Redwood Creek data to determine the trend in population abundance, we eliminated survey efforts by CalTrout and NCFR from 1993-1998 and 2008 because they were only conducted for six out of the 39 total years of the survey and could skew the analysis, producing an inaccurate trend value for the population.

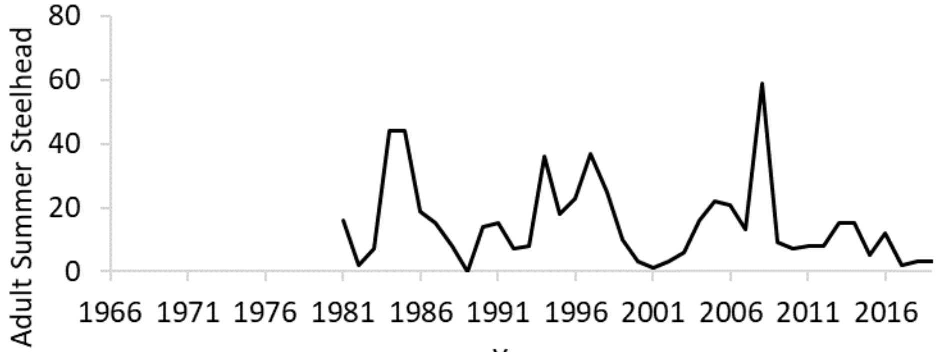
We also note that the Redwood Creek, Mad River, and Mattole River surveys enumerated half-pounders for some or all years the surveys were conducted. We attempted to remove half-pounder counts from our analyses as we do not feel they are representative of true population abundance. Half-pounders ultimately return to the ocean where they are exposed to further threats of mortality. Additionally, the origin of these half-pounders is uncertain as some could be strays from northern Oregon streams. There were four years (1992, 1995, 1997, and 2004) where half-pounders were known to be included in summer steelhead counts for the Van Duzen River. These were removed to the best of our ability by excluding fish less than 16 inches based off information in original survey reports. During a few years of the Redwood Creek survey, fish less than 16 inches (the typical length cutoff for adult summer steelhead) were counted towards adult abundance. These few fish were not removed because they were not designated as half-pounders and were often grouped in with larger fish as a total count rather than enumerated separately. Mad River half-pounders have been enumerated separately each year and, in most years, exceed the number of adults counted. Mad River half-pounders were not included in analyses.

We used data from 1966 through 2019 to analyze the status and trend of the five summer steelhead populations mentioned above. Because annual surveys did not begin until 1966 for the Middle Fork Eel River, and at least a decade later for all other surveys, we want to emphasize that these datasets likely begin after the most devastating declines in abundance had already occurred probably as a result of human land use beginning in the mid-1800s and effects of the catastrophic floods in 1955 and 1964. Thus, any trends found to be significant in this current time series of abundance should be taken in the context that historical numbers were probably higher than the current data suggest. Some of the analyses below were not possible to conduct for all five populations due to missing years of data. For example, the Mad River dataset is missing six years between 2006 and 2019 and the Van Duzen River dataset is missing 11 years over the entire time series of the survey since 1979. Figure 3.8 shows full time series summer steelhead counts for the five populations. *Appendix B* provides annual summer steelhead counts used in Figure 3.8 and identifies data sources.

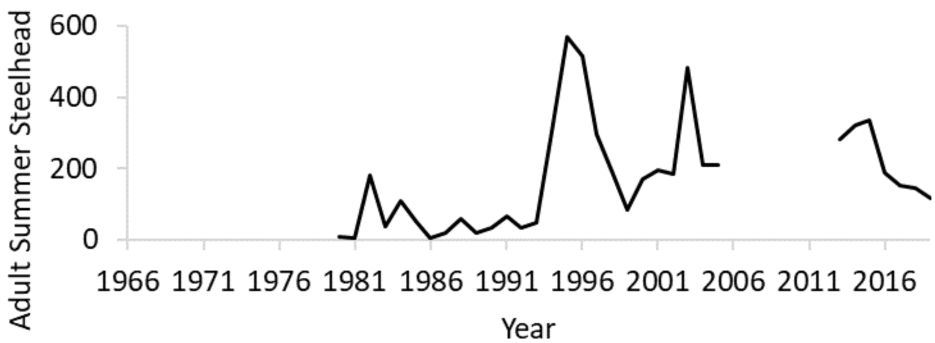
Age-specific estimates of returning adults and subsequent cohort reconstructions would be the best way to analyze population size per generation and population growth rates. Currently, age-specific return estimates are not available for NC summer steelhead, so we decided to use a proxy of mean generation time to calculate total population size per generation (*Section 3.4.1.3 Total Population Size per Generation [Ng*]) and cohort replacement rates (*Section 3.5.1 Cohort Replacement Rate, ln[CRR]*). For these analyses we used a mean generation length of four years. Although Puckett (1975) showed the majority of summer steelhead return to the Middle Fork Eel and Van Duzen rivers at the age-3, a study conducted by Ward (n.d.) in the late 1980's showed an almost even split between age-3 and age-4 summer steelhead returning to the Middle Fork Eel River. Additionally, in an intraoffice correspondence from Leo Shapovalov, a Senior Fisheries Biologist for the Department, he notes scale age work he conducted that showed Middle Fork Eel River summer steelhead spending two years in fresh water and two years at sea (4 years total) before returning for their first spawning event (CDFG 1953). Spence et al. (2008) states that without population-specific age information, the mean generation time for steelhead is assumed to be four years; the most common age for steelhead to spawn within the North-Central California Coast Recovery Domain, which includes the NC steelhead DPS (Spence et al. 2008). Given the available age information and the guidance from Spence et al. (2008) we proceeded with a generation time of four years. Using a standard generation time or age of return does not account for fish returning to the river to spawn multiple times during their life nor fish that return at multiple ages in any given year. This means that one fish may be included in multiple annual adult counts and contribute to more than one generation in our analyses.

It is important to note that we did not use the metric of spawning fish abundance as recommended by Spence et al. (2008), rather we used counts of summer holding fish, which were the most comparable available data. Estimates of adult summer steelhead holding in pool habitat can provide measures of general trends in abundance, though many of these fish succumb to mortality before they are able to spawn (Moyle et al. 2008). Uncertainty surrounding these numbers also comes from factors discussed above like inconsistent survey frame and potential underestimation in snorkel surveys. Due to these uncertainties, summer steelhead counts do not represent estimates of total spawning fish abundance.

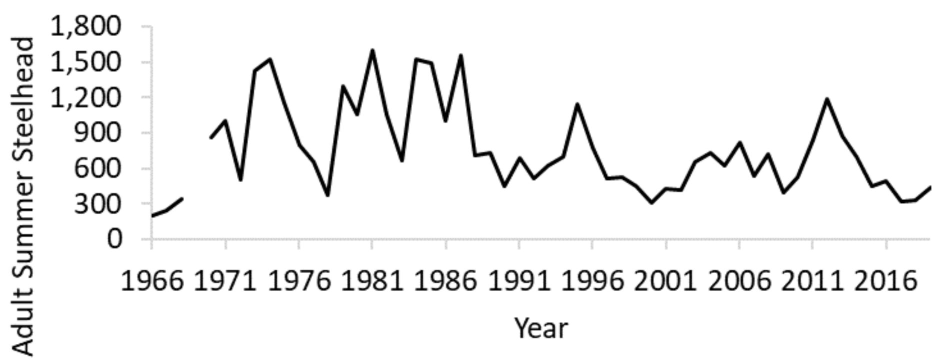
A. Redwood Creek

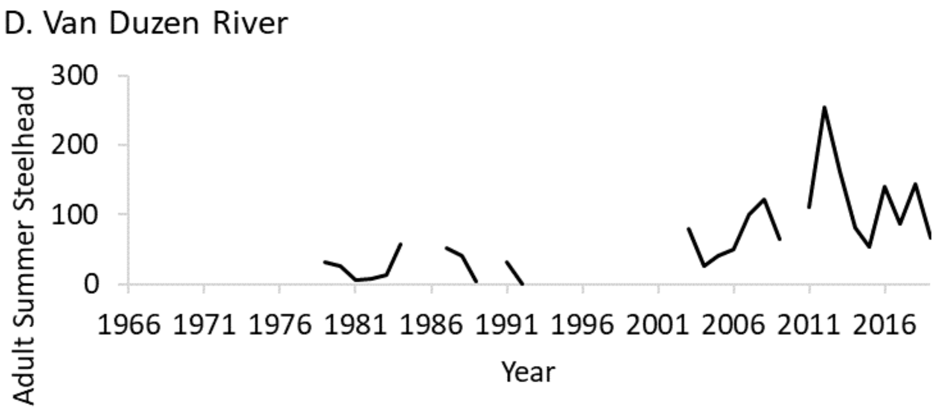


B. Mad River



C. Middle Fork Eel River





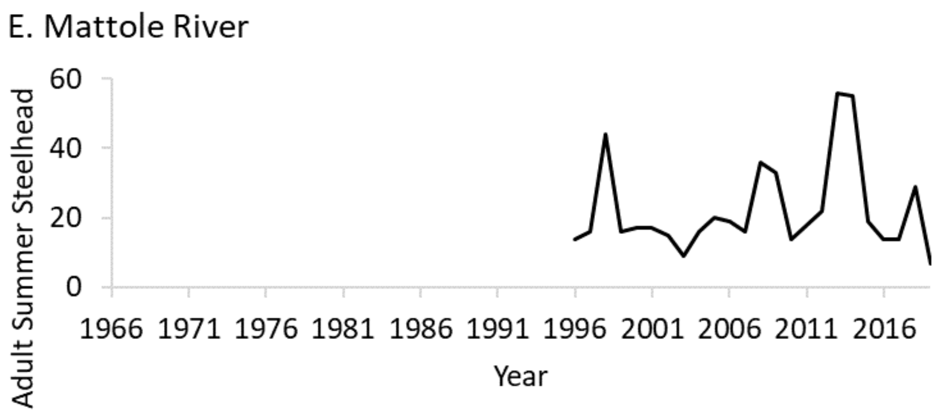


Figure 3.8. *Adult abundance for Northern California summer steelhead populations. A) Redwood Creek. B) Mad River; no data 2006, 2007, 2009-2012. C) Middle Fork Eel River; no data 1969. D) Van Duzen River; no data 1985, 1986, 1990, 1993, 1994, 1996, 1998, 2000-2002. E) Mattole River. Note different scales on the Y axis.*

3.4.1 Abundance Analysis

3.4.1.1 Geometric Mean Abundance

Geometric mean can be an important indicator of status, thus we looked at geometric mean of annual adult summer steelhead abundance (N_a) in the context of three different time frames for the five available population datasets of Northern California summer steelhead. Geometric mean is a useful metric for status evaluation because it calculates central tendency of abundance while minimizing the effect of outliers in the data. It is thought to more effectively characterize the time series of abundance based on counts than the arithmetic average (Good et al. 2005; Spence et al. 2008). Spence et al. (2008) recommend using the geometric mean of spawning fish abundance for the most recent 3–4 generations as an estimator for the mean annual population abundance (Spence et al. 2008). We did not use the arithmetic average because it is known to be overly sensitive to a few large counts and can result in incorrect depiction of central tendency with typically highly variable salmon population data. A range of minimum and maximum abundances are provided for scale (Table 3.2).

In addition to looking at long-term geometric mean abundance using the total available time series, we also chose to analyze a medium-term geometric mean using 12 years to represent three generations, and a short-term geometric mean of 5 years, which has been standard in NMFS reviews to evaluate status (NMFS 2011; NMFS 2016a). There were missing data in some of the time series noted in the following tables. Only the available data were used in the calculations, with no effort to interpolate or otherwise fill in missing data. Using methods from Spence et al. (2008), we defined the geometric mean of adult summer steelhead abundance as:

$$\bar{N}_{a(geom)} = \left(\prod N_{a(i)}\right)^{1/n}$$

where $N_{a(i)}$ is the total number of adult summer steelhead in year *i*, and *n* is the number of years of data available.

Although we evaluated status using the best available long-term data sets, we note that what amounts to "historical abundance" several decades ago may have limited use for predicting future abundance as conditions in the past may be less relevant to current and future abundance. For this reason, we rely on recent 12- and 5-year geometric means as the best indicators of current abundance status (Table 3.3). We also note that the longest time series we have for NC summer steelhead (Middle Fork Eel River) extends only back to 1966 and the most drastic declines in NC summer steelhead likely occurred prior to this time. Pre-1966 declines are not accounted for in the analysis below.

***Table 3.2**. Minimum and maximum abundance for long-term, medium-term, and short-term time frames for the five Northern California summer steelhead populations.*

	Redwood Creek	Mad River	Middle Fork Eel River	Van Duzen River	Mattole River
Long-term					
Years	1981-2019	1980-2019	1966-2019	1979-2019	1996-2019
Min	0	5	198	0	7
Max	59	569	1,601	255	56
12-year					
Years	2008-2019	2008-2019	2008-2019	2008-2019	2008-2019
Min	2	110	323	54	7
Max	59	336	1,191	255	56
5-year					
Years	2015-2019	2015-2019	2015-2019	2015-2019	2015-2019
Min	2	117	323	54	7
Max	12	336	493	144	29

Table 3.3. *Long-term, medium-term, and short-term geometric mean abundance for five Northern California summer steelhead populations.*

Population	Years	Long-term Geometric Mean	Years	12-year geometric mean	Years	5-year geometric mean
Redwood Creek	1981-2019	10	2008-2019	8	2015-2019	4
Mad River[1]	1980-2019	99	2008-2019	189	2015-2019	175
Middle Fork Eel River[2]	1966-2019	667	2008-2019	559	2015-2019	399
Van Duzen River[3]	1979-2019	36	2008-2019	106	2015-2019	91
Mattole River	1996-2019	20	2008-2019	22	2015-2019	15

[1] No data long-term 2006, 2007, 2009-2012; 12-year 2009-2012.

[2] No data long-term 1969.

[3] No data long-term 1985, 1986, 1990, 1993, 1994, 1996, 1998, 2000-2002, 2010; 12-year 2010.

Three populations had lower 5-year geometric mean abundances than their long-term geometric mean abundances (Redwood Creek, Middle Fork Eel River, and Mattole River). All geometric mean abundances for NC summer steelhead are below 1,000 and most are below 500 except for the long-term and 12-year Middle Fork Eel River geometric mean abundances. However, there is a clear decline from the long-term geometric mean abundance of the Middle Fork Eel River population to the 5-year geometric mean abundance, indicating moderate risk of extinction based on population size (Spence et al. 2008). Geometric mean abundances for Redwood Creek and the Mattole River are critically low across all time frames.

3.4.1.2 Changes in Geometric Mean Over Time

We calculated the running 12-year (3-generation) geometric mean abundance for the populations with at least ten data points available, which include Redwood Creek, the Middle Fork Eel River, and the Mattole River (Figure 3.9). Both Redwood Creek and the Mattole River, as stated before, have remained at critically low numbers (under 50 adults per year) across all years. The 3-generation geometric mean of annual adult abundance in the Middle Fork Eel River has declined from over 900 to just over 550. Running 12-year geometric mean abundances could not be calculated for the Mad River or the Van Duzen River due to multiple years of missing data. It is important to note the different scales of abundance between the Middle Fork Eel River and the other two populations, with the Middle Fork Eel River in the hundreds of adults annually and the Mattole River and Redwood Creek in the tens. However, even though the Middle Fork Eel River population has remained in the hundreds of fish per year, the long-term decline is cause for concern.

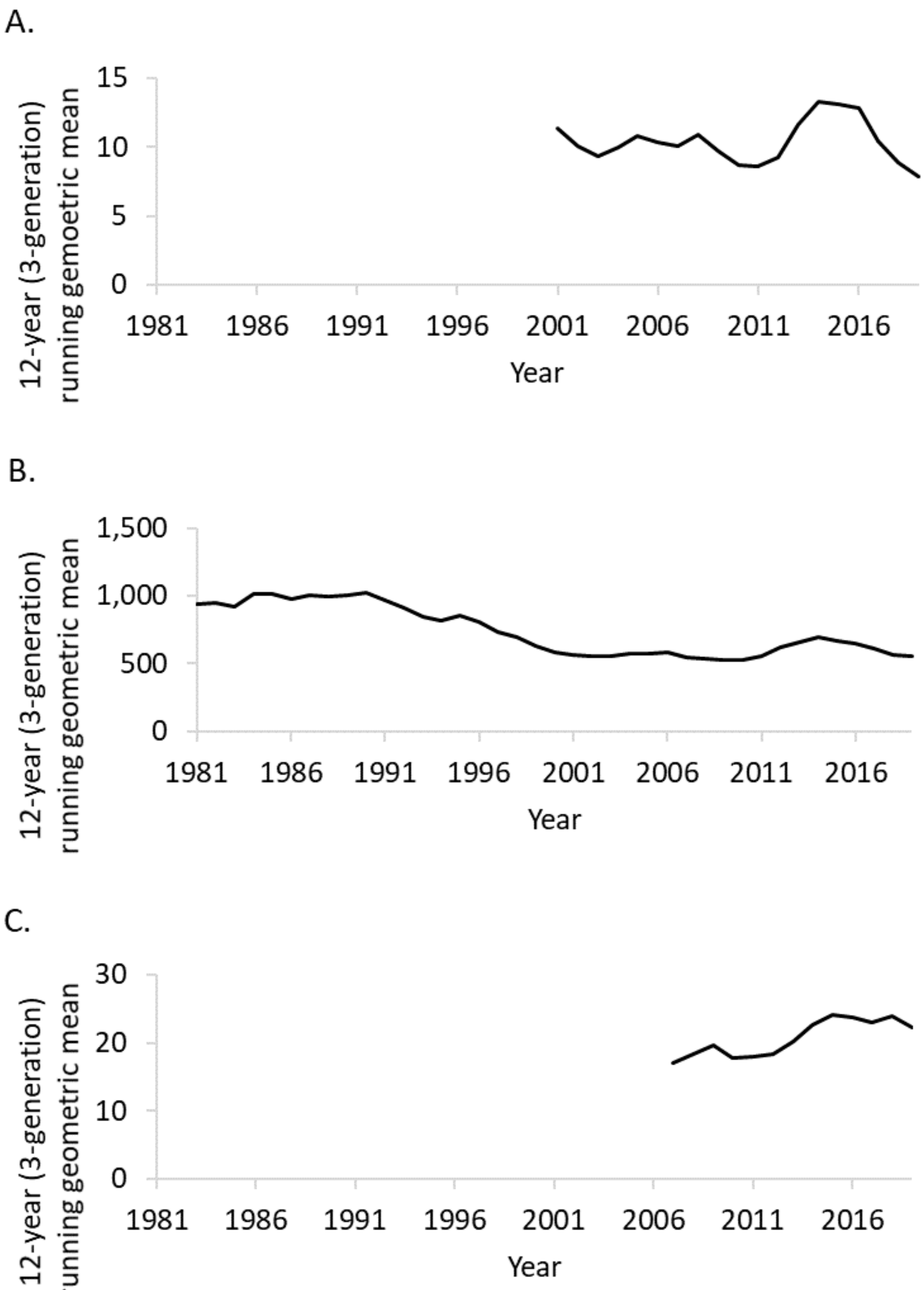


Figure 3.9. *Running 12-year (3-generation) geometric mean of NC steelhead abundance for A) Redwood Creek, B) Middle Fork Eel River, and C) Mattole River. Note different scales on the Y axis.*

3.4.1.3 Total Population Size per Generation (N_g)

Table 3.4 shows the harmonic mean of total population size per generation (N_g) over five years was calculated for five NC summer steelhead populations (Table 3.4). N_g is a criterion used for assessing the level of extinction risk for a population as defined by Spence et al. (2008) (Table 3.5). Generation time is the average interval between birth of an individual and the birth of its progeny. For NC summer steelhead, we do not have age-specific data, so we used a mean generation time of four years (see *Section 3.4 Abundance and Trends* for explanation). To estimate N_g, Spence et al. (2008) suggested using the harmonic mean of the running sum of returning fish abundance for the mean generation time of the species and population over a period representing at least four generations (Spence et al. 2008). Using methods from Spence et al. (2008), the harmonic mean of total population size per generation can be mathematically expressed as:

$$\bar{N}_{g(harm)} = \frac{1}{\frac{1}{n}\sum_{t-1}^{n}\frac{1}{N_{g(t)}}}$$

Where $N_{g(t)}$ is the running sum of adult abundance at time *t* for a period equal to the mean generation time *k* of the population:

$$N_{g(t)} = \sum_{i=t-k}^{t} N_{a(i)}$$

Where $N_{a(i)}$ is the total number of adult summer steelhead in year *i*. It is recommended that a period of four generations be used (Spence et al. 2008); however, for two of our five datasets, we were unable to calculate these values using four generations due to missing years of data. Because of this discrepancy, we decided to show the recent time frame of five years to illustrate current population size per generation, using a mean generation time of four years. For the Mad River, harmonic mean of total population size per generation could only be calculated for the most recent four generations due to missing years of data.

Table 3.4. *Short-term harmonic mean of total population size per generation for Northern California summer steelhead populations.*

Population	Years	Short-term harmonic mean of total population size per generation
Redwood Creek	2015-2019	30
Mad River[1]	2016-2019	838
Middle Fork Eel River	2015-2019	2,016
Van Duzen River	2015-2019	434
Mattole River	2015-2019	96

[1] Only calculated for four generations due to missing years of data

Given the risk extinction criteria from Spence et al. (2008) and Allendorf et al. (1997) (Table 3.5), all Northern California summer steelhead populations with available population data are at least at moderate risk of extinction, with two populations, Redwood Creek and the Mattole River, at high risk of extinction. The NMFS viability target for all NC summer steelhead populations is defined to be $\overline{N}_{g(harm)}$ = 2,500 (Williams et al. 2016). The Department has not established abundance targets for NC summer steelhead populations. NMFS viability targets were developed based on what would constitute viable population sizes and do not necessarily reflect estimates of historical abundance. None of the five surveyed populations are at or approaching their viability targets. The Middle Fork Eel River population is the closest at around 2,000 adult summer steelhead per generation.

Table 3.5. *Criteria for assessing the level of risk of extinction for populations of Pacific salmonids. Overall risk is determined by the highest risk score for any category.* N_g *= total number of spawning fish per generation*; N_e *= effective population size*; *and* N_a *= annual spawning fish abundance.* (*Adapted from Spence et al. 2008)*

Population Characteristic	Extinction Risk High	Extinction Risk Moderate	Extinction Risk Low
Extinction risk from population viability analysis (PVA)	≥ 20% within 20 yrs	≥ 5% within 100 yrs but < 20% within 20 yrs	< 5% within 100 yrs
	- or any ONE of the following -	- or any ONE of the following -	- or ALL of the following -
Effective population size per generation *or* Total population size per generation	$N_e \leq 50$ *or* $N_g \leq 250$	$50 < N_e < 500$ *or* $250 < N_g < 2500$	$N_e \geq 500$ *or* $N_g \geq 2500$
Population decline	Precipitous decline[a]	Chronic decline or depression[b]	No decline apparent or probable
Catastrophic decline	Order of magnitude decline within one generation	Smaller but significant decline[c]	Not apparent
Spawning fish density	$N_a/IPkm$[d] ≤ 1	$1 < N_a/IPkm <$ MRD[e]	$N_a/IPkm \geq$ MRD[e]
Hatchery influence[f]	Evidence of adverse genetic, demographic, or ecological effects of hatcheries on wild population	Evidence of adverse genetic, demographic, or ecological effects of hatcheries on wild population	No evidence of adverse genetic, demographic, or ecological effects of hatchery fish on wild population

[a] Population has declined within the last two generations or is projected to decline within the next two generations (if current trends continue) to annual run size $N_a \leq 500$ spawning fish (historically small but stable populations not included) *or* $N_a > 500$ but declining at a rate of ≥10% per year over the last two-to-four generations.
[b] Annual run size N_a has declined to ≤ 500 spawning fish, but is now stable *or* run size $N_a > 500$ but continued downward trend is evident.
[c] Annual run size decline in one generation < 90% but biologically significant (e.g., loss of year class).
[d] *IPkm* = the estimated aggregate intrinsic habitat potential for a population inhabiting a particular watershed (i.e., total accessible km weighted by reach-level estimates of intrinsic potential; see Bjorkstedt et al. [2005] for greater elaboration).
[e] MRD = minimum required spawning fish density and is dependent on species and the amount of potential habitat available.
[f] Risk from hatchery interactions depends on multiple factors related to the level of hatchery influence, the origin of hatchery fish, and the specific hatchery practices employed.

3.4.2 Trend Analysis

Population trends were estimated using the methods described by Good et al. (2005) and Spence et al. (2008). Population trend, *T*, was estimated as the slope of the number of natural-area adult summer steelhead (log-transformed) regressed against time. A value of one was added to the number of natural-area adult summer steelhead before the log-transformation to address any zero values if they were present in the dataset [i.e., $ln(\overline{N}_a + 1)$]. Using methods from Good et al. (2005), the linear regression can be expressed as:

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

Where $\overline{N}_a$ is annual adult summer steelhead abundance, β_0 is the intercept, β_1 is the slope of the equation, and ∈ 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:

$$\exp(\beta_1) \pm t_{0.05(2),df} s_{b_1}$$

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_{b_1} is the standard error of the estimate of the slope, b_1 (Good et al. 2005).

Long-term trends are estimated using all available data in the time series. Spence et al. (2008) states that a time series of at least two and up to four generations of adult abundance is necessary to estimate trend (Spence et al. 2008). We evaluated recent trends using what we consider the minimum amount of data (at least 10 data points within the most recent 12-years; three summer steelhead generations) likely to give reliable trend estimates (Table 3.6). Trend values less than one indicate average population component decline, whereas trend values greater than one indicate average growth. Confidence intervals that range from less than one to greater than one indicate a lack of statistical support for the calculated trend.

For Redwood Creek, as discussed in *Section 3.4.1*, we used only the counts collected by the NPS to provide a standardized reach for our trend analysis. During the years 1993 – 1998 and 2008, CalTrout and/or NCFR surveyed upstream portions of Redwood Creek and their counts exceeded those within the NPS sections of the creek. In order to provide a more reliable analysis, CalTrout and NCFR data were excluded from calculations of trend for Redwood Creek and we used only NPS data from approximately 24 miles (37.8 km) of surveyed habitat. CalTrout and NCFR data are included in all of the abundance analyses above.

For the Mad River, survey extent varied drastically from year to year until the Mad River Alliance assumed responsibility in 2012 with the intention of implementing consistent protocols and a comprehensive survey frame. Since 2012, about 50 miles of the mainstem Mad River is covered by the summer steelhead snorkel survey between Matthews Dam and Kadle Hole. Because of the inconsistency in number of miles surveyed until 2012, we did not calculate long-term trend for the Mad River. NMFS' most recent five-year status review of Pacific Salmon and steelhead listed under the ESA also did not calculate for summer steelhead abundance in the Mad River (Williams et al. 2016). We could not complete recent trend analysis for the Mad River due to missing years of data.

The number of miles surveyed on the Van Duzen River also varied greatly between 1979 and 2010 with only six of those years representing what would be considered a comprehensive survey (S. Thompson, CDFW, pers. comm., 28 February 2020). Thus, we did not calculate long-term trend for summer steelhead in the Van Duzen River. In 2011, the Department implemented consistent annual survey efforts that likely cover most of the holding habitat used by summer steelhead. Although the recent trend analysis incorporates survey data from 2008-2009 when only partial surveys were conducted (no survey in 2010), we have reported the trend value because most surveys were roughly equivalent in extent during the recent time frame. Data prior to 2011 was not used by Williams et al. (2016) nor did they calculate trend for Van Duzen River summer steelhead abundance.

Table 3.6. *Long-term and recent trends in adult abundance using slope of ln-transformed time series counts for four Northern California summer steelhead 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**	**Long-term Trend**[1]	**Lower 95% CI**	**Upper 95% CI**	**Years**	**Recent Trend**[1]	**Lower 95% CI**	**Upper 95% CI**
Redwood Creek[2]	1981-2019	0.99	0.97	1.02	2008-2019	**0.87**	0.78	0.97
Mad River[3]	1980-2019	-	-	-	2008-2019	-	-	-
Middle Fork Eel River	1970-2019	**0.98**	0.98	0.99	2008-2019	0.94	0.88	1.01
Van Duzen River[4]	1979-2019	-	-	-	2008-2019	0.98	0.89	1.08
Mattole River	1996-2019	1.01	0.97	1.04	2008-2019	0.93	0.83	1.04

[1] "-" indicates no trend analysis available.

[2] Standardized reach used; represents only a partial population estimate.

[3] No data long-term 2006, 2007, 2009-2012; recent 2009-2012. Survey frame inconsistent until 2012.

[4] No data long-term 1985, 1986, 1990, 1993, 1994, 1996, 1998, 2000-2002, 2010; recent 2010. Survey frame inconsistent until 2011.

Redwood Creek shows a statistically significant recent downward trend (p = 0.016) though no significant long-term trend (p = 0.451). This population has been and remains at critically low abundance (Williams et al. 2016). The Middle Fork Eel River population, even though it is the most stable of all NC summer steelhead populations, has experienced a statistically significant negative long-term trend (p = 0.0001) and a negative recent trend that is approaching statistical significance (p = 0.098). The Van Duzen summer steelhead population showed no statistically significant trend (p = 0.660) for the recent time frame though additional years of data will be useful for future analyses. The Mattole River has the shortest dataset of NC summer steelhead populations. Neither the long-term or recent trends are statistically significant (p = 0.707 and p = 0.166, respectively).

In general, based on the available data, NC summer steelhead seem to be experiencing a slight decline. However, there were only two statistically significant trend values, Redwood Creek recent and Middle Fork Eel River long-term. Trends that were not statistically significant mostly indicate slight declines over both long-term and recent time frames.

3.5 Productivity

3.5.1 Cohort Replacement Rate, ln(CRR)

We evaluated productivity of the Northern California summer steelhead populations by evaluating cohort replacement rate over time. Cohort Replacement Rate (CRR) is:

$$CRR = \ln\left(\frac{N_{t+4}}{N_t}\right)$$

Natural log-transformed CRRs >0 indicate that the cohort increased in size that year in relation to the brood year three years earlier, whereas *ln* CRR <0 indicates that it declined. This analysis assumes a generation time for Northern California summer steelhead of four years, which has been determined to be reasonable based off our best understanding of summer steelhead age structure (CDFG 1953; Spence et al. 2008; Ward, n.d.). CRRs were calculated for four populations of NC summer steelhead (Figure 3.10). The Van Duzen River had too many years of missing data to produce meaningful results.

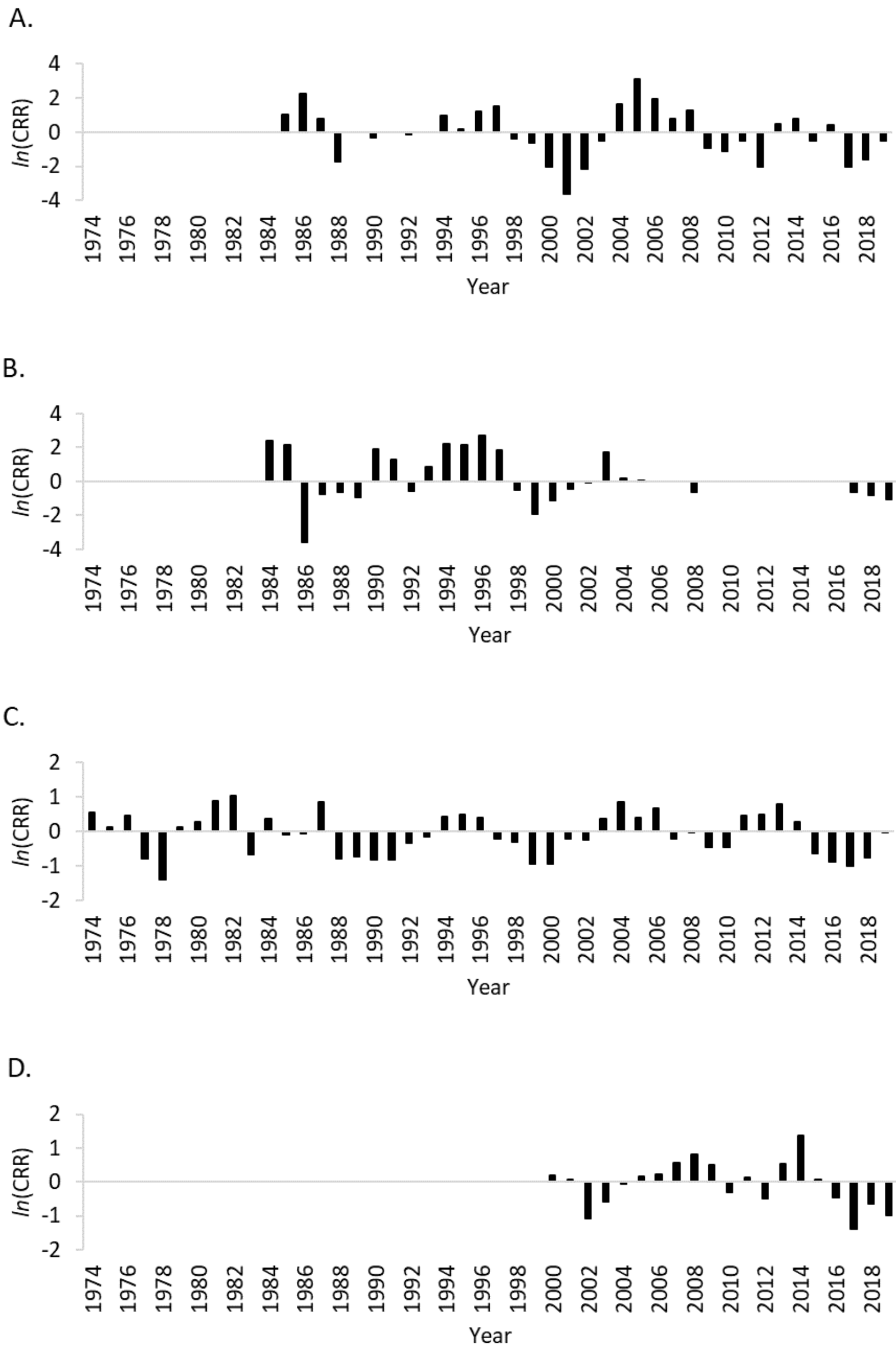


Figure 3.10. Ln-*Cohort Replacement Rates for four NC summer steelhead populations, A) Redwood Creek, B) Mad River, C) Middle Fork Eel River, and D) Mattole River. Gaps are due to years of missing data. Gaps are a result of missing years of data. Note different scales on the Y axis.*

Although not completely consistent, a cyclical pattern of growth and decline can be detected when looking at CRR's over the time series available. Cyclical patterns are seen in other fish populations, though cycles are generally most apparent when there is a strong density-dependent factor involved or when one age-class strongly influences recruitment and spawning occurs only once (Townsend 2006). CRR's for all four populations evaluated were almost evenly divided between positive and negative values. Redwood Creek and the Mad and Middle Fork Eel rivers had slightly more negative CRR's than positive, and the Mattole River had slightly more positive CRR's than negative. The most recent years have seen declines in productivity for all populations.

3.5.2 Middle Fork Eel River Juvenile Density

Surveys have been conducted to determine juvenile standing crop density at two sites on the Middle Fork Eel River during the fall since the 1980s. Collection at Fern Point, the upstream site, began in 1980, and collection at the Osborn site began a few years later in 1984. The Fern Point and Osborn sites were selected because they generally represent the upper and lower extents, respectively, of summer steelhead holding distribution (S. Thompson, CDFW, pers. comm., June 19, 2020). Although neither site likely has spawning activity due to the majority of the substrate being comprised of boulder and cobble, there is suitable spawning habitat about a quarter kilometer away from both (S. Thompson, CDFW, pers. comm., June 19, 2020). Multi-pass depletion surveys are conducted at each site with two backpack electrofishing units along a transect about 30 meters in length. Electrofishing has not been used at Fern Point since 2012 due to the absence of adult steelhead above Asa Bean Roughs. Instead, a snorkel survey has been conducted at Fern Point beginning in 2013 (CDFG 1966 – 2018). Multiple year classes are encountered on these surveys and length frequency analysis is used to separate young-of-the-year (YOY) from yearling plus (Y+) fish (CDFG 1966 – 2018).

Assemblages of juveniles at the Fern Point and Osborn sites are assumed to mostly consist of summer steelhead, though it is possible that some winter steelhead are mixed in at the Osborn site given its location further downstream (S. Thompson, CDFW, pers. comm., June 18, 2020). There is likely a small contribution from resident Rainbow Trout at both sites, as well. Juvenile abundance at Fern Point from 2013-2019 provides an idea of resident trout contribution since there were no summer steelhead in those years above Asa Bean Roughs. Juvenile densities of about 0.01 fish/m^2 as seen at Fern Point in recent years are typical for habitat above barriers to anadromy throughout Mendocino County (S. Thompson, CDFW, pers. comm., June 18, 2020).

McElhany et al. (2000) suggest that estimates of smolt production can provide a measure of productivity by showing a population's potential to increase and its ability to sustain in poorer conditions (McElhany et al. 2000). Although we do not have smolt production estimates, looking at changes in juvenile density over time may provide a general trajectory for juvenile abundance and provide some understanding of summer steelhead productivity. Below are graphs depicting juvenile density (fish/m^2) at the Fern Point and Osborn sites (Figure 3.11). YOY fish tend to be the dominant age class within juvenile assemblages at the collection sites (S. Thompson, CDFW, pers. comm., June 19, 2020). We note that densities at these two sites should not be used as estimates for the entire Middle Fork Eel River. Because only two sites were sampled, fluctuations in density may be somewhat attributable to fish behavior rather than abundance. Interannual changes in spawning distribution or carrying capacity of the two habitats could have a substantial impact on these site-specific juvenile abundances, as well. It is also important to understand that, although a linear relationship has been found between number of juvenile outmigrants and number of returning adults (Ward and Staley 1988), smolt-to-adult survival is highly dependent on environmental conditions in migration corridors and even more so in the ocean especially during the early marine life stage (Kendall et al. 2017).

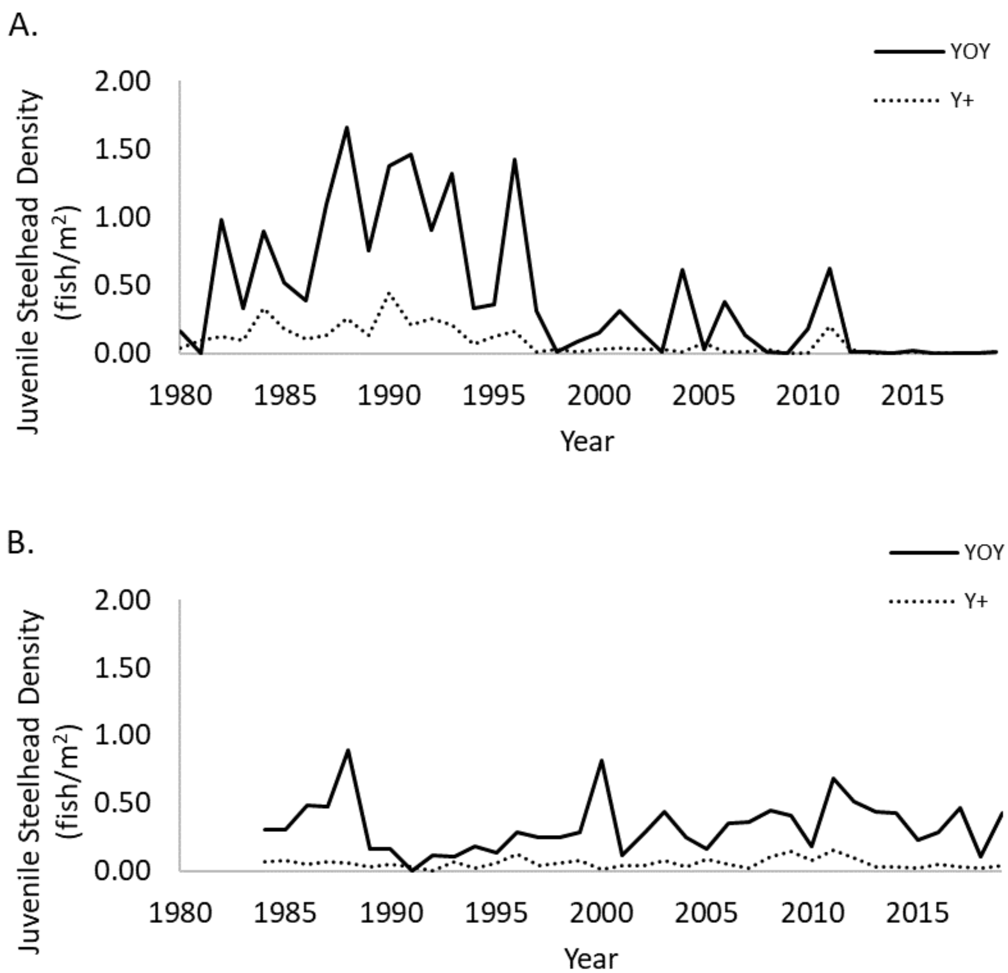


Figure 3.11. *Juvenile (YOY and Y+) summer steelhead densities at A) Fern Point and B) Osborn sites on the Middle Fork Eel River 1980 – 2019. Osborn site sampling began in 1984.*

Fern Point juvenile density is trending negatively across the entire timespan of the survey for both age classes. Densities in the most recent years when summer steelhead have not been observed above Asa Bean Roughs likely have a substantial effect on these trends, although even in years prior there was generally a downward trajectory, especially as compared to densities during the late 1980s through the mid-1990s. The decline in fish densities at Fern Point after 1996 is likely a result of the rearrangement of boulders within Asa Bean Roughs the year prior, which has made it more difficult for adult summer steelhead to reach Fern Point in recent years and indicates a possible shift in summer steelhead distribution to lower areas of the watershed.

In contrast to Fern Point juvenile densities, Osborn juvenile densities seem to be trending in a slightly positive direction. This trend could be influenced by the downstream shift in summer steelhead distribution. Prior to 2013, when summer steelhead were still being observed above Asa Bean Roughs, the Osborn site had lower juvenile summer steelhead abundance on average than Fern Point. Compressing spawning and rearing habitat into a smaller area has the potential to create density-dependence issues such as crowding, increased predation, and competition, which could have detrimental effects on population size and productivity if carrying capacity of the smaller habitat area is exceeded.

3.6 Spatial Structure

Spatial structure of a population encompasses 1) the geographic distribution of individuals within the population and 2) the processes that produce this geographic distribution (McElhany et al. 2000). Salmonid habitat is often inconsistent throughout a river system, so fish will use distinct habitat "patches" that have varying levels of suitability. Scale used to define the size of a habitat "patch" is not defined as they are specific to individual populations. Salmonid population spatial structure is not clearly understood and the configuration of a typical spatial structure for salmonids is unknown; however, McElhany et al. (2000) defined guidelines that are thought to contribute positively to sustaining the spatial structure of a viable salmonid population:

1. Habitat patches should not be destroyed at a faster rate than they develop naturally.
2. Human activities should not affect natural stray rates between subpopulations.
3. Some suitable habitat patches should be preserved even if they are not currently used.
4. Source subpopulations should be preserved.
5. Uncertainty and lack of information should be accounted for in analyses of spatial structure. As a default, Historic spatial processes are assumed to have been sustainable, thus, they should be preserved as a default.

Spatial structure within NC summer steelhead populations has changed over time especially as a result of anthropogenic activities, catastrophic weather events, and cumulative effects. Reductions in habitat availability and suitability are likely the primary consequences of these activities. Man-made dams have blocked off substantial summer steelhead habitat on the Mad River and the mainstem Eel River, which has compressed the summer steelhead range in those rivers and likely caused more overlap in summer and winter steelhead spawning habitat use. Although the Matthews Dam likely only removed access to a couple miles of potential habitat on the Mad River (NMFS 2016b), Scott Dam is estimated to have blocked 198-288 miles of potential upstream steelhead habitat on the mainstem Eel River and tributaries of the Lake Pillsbury basin (Cooper 2017). Other human land uses such as timber harvest, agriculture, mining, and road construction have had a myriad of effects on availability and quality of summer steelhead habitat. Though we cannot classify the types of population or subpopulation structures for NC summer steelhead, it is clear they have experienced effects of the changing environment and their historical spatial processes have been substantially altered. Many habitat "patches" that were used historically are no longer available or suitable. The loss of spatial structure within a population, in addition to habitat homogenization and loss of habitat connectivity, removes the mechanism by which steelhead persevere through environmental shifts and catastrophic events (Crozier et al. 2019).

3.7 Diversity

Diversity is the fourth criterion to consider when evaluating viability of salmonid populations (McElhany et al. 2000). Diversity is a key component of viability because it allows a species to use a wider array of environments than they could without it, protects a species against short-term spatial and temporal changes in the environment, and provides the raw genetic material for surviving long-term environmental changes. Waples et al. (2001) emphasize that general conservation goals should include preserving the diverse assemblage of populations necessitated to buffer anthropogenic effects on the natural processes of evolution (Waples et al. 2001).

Diversity has been shown to be adaptive (Taylor 1991) and severely altering or losing some aspects of it can have a detrimental effect on population viability (McElhany et al. 2000). McElhany et al. (2000) also provide guidelines for maximizing diversity in salmonid populations as follows:

1. Human-induced habitat changes, harvest pressures, artificial propagation, and exotic species introduction, or other factors should not considerably modify variation in traits, e.g., migration timing, age structure, size, fecundity, morphology, behavior, and molecular genetic characteristics.
2. Human-caused factors should not significantly change the rate of inter-population gene flow.
3. Natural processes that result in ecological variation should be preserved.
4. Population status evaluations should take uncertainty about requisite levels of diversity into account. Uncertainty regarding necessary levels of diversity should be accounted for in evaluations of population status.

Anthropogenic activities can have a significant impact on straying between populations, and as a result, change local adaptations and patterns of diversity (Keefer and Caudill 2014). Alterations of flow regimes and migration pathways, as well as instream contaminants like pesticides, can inhibit homing of salmonids, causing them to stray from natal spawning grounds (Keefer and Caudill 2014). In addition to affecting inter-basin straying, anthropogenic impacts can have a devastating effect on productivity and survival. When populations dip into low abundances, the risk of genetic bottlenecks is much greater and thus extinction risk becomes higher. At low densities, populations may experience what is called depensation, where per capita growth rate decreases as abundance decreases (Spence et al. 2008). This causes numerous negative effects such as reduced likelihood of finding mates, more detrimental effects of predation, loss of habituation to the local environment, and loss of effective group dynamics (Liermann and Hilborn 2001). Depensation-related consequences can have marked influence on extinction risk.

Stream-maturing (summer) steelhead and (spring) Chinook Salmon have experienced more population losses than the ocean-maturing life history type likely due to significant loss of the high elevation, over-summer, holding habitat that they require as well as higher vulnerability to multiple threats during that holding time (Gustafson et al. 2007). Gustafson et al. (2007) argue that sustaining salmon and steelhead diversity requires conservation of local populations and their habitat. Summer steelhead in the NC steelhead DPS have already experienced population losses, which may have resulted in a loss of local adaptations and genetic diversity. Populations of summer steelhead in the upper mainstem and upper middle mainstem of the Eel River were already lost with the construction of Scott Dam and additional populations of NC summer steelhead are near extirpation. Loss of the summer migration life history type in the NC Steelhead DPS would represent a loss of crucial genetic and phenotypic diversity for the species.

3.8 Conclusions

Our analyses have demonstrated that some NC summer steelhead populations are currently at critically low abundances and face a high risk of extinction, while others at higher abundance levels are at moderate risk of extinction. NMFS set viability targets for all NC summer steelhead populations at 2,500 adult summer steelhead per generation and, at this time, none of the populations with long-term survey data are near or approaching this target. Redwood Creek and the Mattole River have rarely seen over 50 summer steelhead each in a given year. Although long-term population trends have been mixed with both positive and negative trajectories, recent trends for all five summer steelhead populations assessed show slight declines, though most of these trends were not statistically significant. We would also like to emphasize that, although survey numbers may be underestimates as explained earlier, it is likely that comprehensive monitoring data would still show these population segments to be depressed as compared to speculations of what abundances were in the early to mid-20th

century (CDFG 1966 as cited in Jones 1992; USACOE 1973 as cited in CDFG 2006c; Jones 1992; CDFG 2006a, 2006b, 2006c;).

Evaluation of the four viability criteria from McElhany et al. (2000) (i.e., abundance, productivity, spatial structure, diversity) indicate that none of the NC summer steelhead populations are viable, nor are they at or approaching the viability targets identified for them by NMFS. Recent trends in abundance, although not statistically significant, are slightly negative for all populations that were evaluated, indicating that these populations will remain at critically low numbers. Productivity of NC summer steelhead populations is cyclical in nature, but there have been a few more negative CRR's than positive across years that surveys occurred. Juvenile abundance at two sites in the Middle Fork Eel River, which supports the largest summer steelhead population in the NC steelhead DPS, shows changes in distribution and habitat use that could negatively affect productivity. Spatial structure of NC summer steelhead has been markedly altered from historical population structure as a result of reduced habitat availability and suitability in all watersheds. Additionally, select populations have already been extirpated. These population losses and reductions increase the risk of depensation, genetic bottlenecks, and lack of diversity.

Spence et al. (2008) considered population viability from two perspectives. The first objective is to define the minimum viable population size at which a population would likely persist at a specified probability over a specified amount of time. The second objective of Spence et al. (2008) looks at viability as it relates to how a population is currently functioning compared to its historical function. An important part of this second perspective on viability is using historical conditions of abundance, productivity, spatial structure, and diversity as a baseline for populations having high probability of long-term persistence. Unfortunately, we do not have quantitative estimates of historical conditions for NC summer steelhead for this purpose.

Current population abundances in NC summer steelhead watersheds that were assessed are likely much lower than what was suggested by historical accounts, which have led to theories of multiple hundreds to multiple thousands of fish, depending on the drainage (CDFG 1966 as cited in Jones 1992; USACOE 1973 as cited in CDFG 2006c; Jones 1992; CDFG 2006a, 2006b, 2006c). Generally, small populations are much more likely to respond poorly to environmental variation, genetic processes, demographic stochasticity, and catastrophic events, and thus are much higher risk for extinction than large populations (McElhany et al. 2000). Populations with low densities of spawning fish can be at risk for depensation (McElhany et al. 2000; Spence et al. 2008). Spence et al. (2008) suggests that depensation can trigger a positive feedback loop that propels a population towards extinction. Although we do not have evidence of depensation occurring explicitly in NC summer steelhead populations, their low population sizes suggest it could be a concern.

4. HABITAT NECESSARY FOR SURVIVAL

4.1 Adults

4.1.1 Migration

Summer steelhead enter freshwater during the spring and migrate to river headwater reaches. Unlike winter steelhead, summer steelhead enter as immature fish and complete their life cycle after they migrate. Summer steelhead enter the Mad River between April and July, and the more southern Mattole River between March and June (Moyle et al. 2017). Upstream migration and holding areas are contingent upon rainfall and stream discharge because steelhead enter the rivers during the ascending or receding flows of spring depending on if hydrology is influenced more by rainfall or snowmelt. Summer steelhead rely on adequate water depths and velocity to reach the upstream pools they hold in during the summer (Reiser and Bjornn 1979; Moyle 2002; NMFS 2016b). Holding habitats are typically distributed farther inland than winter migration holding areas (Busby et al. 1996; Moyle 2002). Ideal velocity for upstream migration is 40 – 90 cm/s with maximum velocities of 240 cm/s (NMFS 2016b). Upstream migration also requires water levels to be at a minimum appropriate depth for passage of at least 18 cm (Bjornn and Reiser 1991) unless the distance where water is less than 18 cm deep is short enough to pass through (M. Sparkman, CDFW, pers. comm., July 8, 2020). Due to their deeper-bodied, more robust morphology as compared to other California steelhead (Bajjaliya et al. 2014), NC steelhead may require slightly deeper water for passage. Along with high water velocity and low water levels, upstream migration can be physically blocked in many ways including dams, plugged or blocked culverts, fish passage facilities that are improperly sized, or extended sandbar closures (NMFS 2016b). Steelhead are able to jump a maximum of 3.4 m (Reiser and Peacock 1985 as cited in Spence et al. 1996). Typically, pool depth must be at least 25% greater than barrier height to achieve the required swimming velocity to pass the barrier (Stuart 1962 as cited in Spence et al. 1996). Adult salmonids can sometimes use waterfall back currents to propel themselves over obstacles (M. Sparkman, CDFW, pers. comm., July 8, 2020).

4.1.2 Holding

Adult summer steelhead hold for many months in cool, deep pools which are often characterized as thermally stratified and ideally have riparian forest cover (NMFS 2016b). Preferred holding velocities are around 0.28 m/sec for adults with higher velocities requiring additional energy expenditure (Moyle and Baltz 1985). In addition to holding velocities, high temperatures can affect productivity and survival. Temperatures of 23 – 24°C can be lethal for adult steelhead (Moyle 2002). Deeper pools provide cooler water temperatures often due to shade from riparian forest cover, ledges, subsurface flow, and thermal stratification (Moyle et al. 2017). Pool availability varies within each river system; however, depths of pools used by summer steelhead has been documented as ranging from approximately one meter to over three meters, and deeper sections are used when pools are thermally stratified during the summer when surface temperatures are warmer during the day (Nakamoto 1994; Nielsen et al. 1994; Baigún et al. 2000; Baigún 2003). In the New River, California, adult summer steelhead occupied areas with less than 35% substrate embeddedness, mean water depths of about 1.0 – 1.4 m, and riparian shade cover (Nakamoto 1994). Department scientist, Michael Sparkman, notes seeing summer steelhead in even deeper pools in the New River, up to 20 feet in depth (M. Sparkman, CDFW, pers. comm., July 8, 2020). Data collected by the Department through the annual census on the Van Duzen and Middle Fork Eel rivers found a higher percentage of adult summer steelhead were in pools with depths between 4.4 – 5.4 m and 3.5 – 4.4 m, respectively (Tables 4.1 and 4.2) (CDFW internal data, S. Thompson). In both rivers, there was a higher percentage of shallow pools than pools with greater depths and the majority of fish used the deeper pools. In the Van Duzen River, 30% of pools had depths ranging from 1.5 – 2.4 m while in the Middle Fork Eel River, 32% of pools counted had depths of 2.5 – 3.4 m. This suggests that pools were selected based on depth preference rather than on availability. Summer steelhead have been found to use cold pool habitat more so than any other habitat types, despite the fact that pools often make up a smaller proportion of available habitat (Bisson et al. 1988; Baigún et al. 2000; Baigún 2003; Moyle et al. 2017).

Table 4.1. *Maximum depth of pools occupied by adult summer steelhead in the Van Duzen River. Survey years: 2011 – 2019.*

Maximum pool depth (m)	Count of pools at depth	% of pools at depth	Sum of fish at depth	% of fish at depth
0.9-1.5	5	12%	64	7%
1.5-2.4	13	30%	71	7%
2.5-3.4	4	9%	60	6%
3.5-4.4	5	12%	156	16%
4.5-5.4	10	23%	608	63%
5.5-6.4	5	12%	5	1%
6.5-7.4	1	2%	2	0%

Table 4.2. *Maximum depth of pools occupied by adult summer steelhead in the Middle Fork Eel River. Survey years: 2000, 2001, 2004 – 2008, 2010 – 2019.*

Maximum pool depth (m)	Count of pools at depth	% of pools at depth	Sum of fish at depth	% of fish at depth
0.9-1.4	13	2%	44	0.5%
1.5-2.4	199	28%	956	10.3%
2.5-3.4	229	32%	2037	21.9%
3.5-4.4	136	19%	2720	29.3%
4.5-5.4	106	15%	2479	26.7%
5.5-6.4	26	4%	701	7.5%
6.5-7.4	9	1%	349	3.8%

Nielsen et al. (1994) found that during the summer in the Middle Fork Eel River, when midday ambient stream temperatures were between 26 and 29°C, adults were found deep in stratified pools where cold water pockets averaged 3.5°C cooler than surface temperatures (Nielsen et al. 1994). In several Northern California streams such as Redwood Creek, Rancheria Creek, and Middle Fork Eel River, cold water sources included tributary inflow seepage from groundwater source along the channel bed, thermal stratification of deep, still water, and intergravel flow (Nielsen et al. 1994). Intergravel flow through river bars into pools was 3 – 9°C cooler than water that entered through the mainstem and accounted for almost all of the cold-water inflow in pools not associated with a tributary (Nielsen et al. 1994). In the Middle Fork Eel and Van Duzen rivers, large pools maintain temperature stratification simply by remaining undisturbed by surface flows, while smaller pools can have temperature stratification due to hyporheic flows from adjacent gravel bars and tributary inputs (S. Thompson, CDFW, pers. comm., July 14, 2020). Some pools in the Van Duzen River also maintain temperature stratification as a result of seepage from banks with dense riparian vegetation or springs and seepage from upslope sources like fens or ponds. These additional processes observed in the Van Duzen River generally do not occur in the Middle Fork Eel River partly because most of the riparian canopy has been removed by previous floods (S. Thompson, CDFW, pers. comm., July 14, 2020). Annual Department surveys on the Van Duzen and Middle Fork Eel rivers generally found fish in pools with bottom temperatures less than 20°C and surface temperatures ranging up to 23°C (Tables 4.3, 4.4, 4.5, and 4.6) (CDFW unpublished data, S. Thompson). These temperatures align with studies in other California streams. USFS (1986) suggested that water temperatures should not exceed 20°C in holding areas within the Klamath, Mendocino, Shasta-Trinity, and Six Rivers National Forest management areas.

Table 4.3. *Measured bottom temperature of pools occupied by adult summer steelhead in the Van Duzen River. Survey years: 2011 – 2019.*

Temperature (°C)	Count of pools at temperature	% of pools at temperature	Sum of fish at temperature	% of fish at temperature
< = 17	5	11%	209	22%
18	4	9%	90	10%
19	10	23%	340	36%
20	10	23%	135	14%
21	3	7%	39	4%
22	9	20%	65	7%
23	1	2%	20	2%
24	2	5%	40	4%
25	0	0%	0	0%
26	0	0%	0	0%

Table 4.4. *Measured bottom temperature of pools occupied by adult summer steelhead in the Middle fork Eel River. Survey years: 2000, 2001, 2004 – 2008, 2010 – 2019.*

Temperature (°C)	Count of pools at temperature	% of pools at temperature	Sum of fish at temperature	% of fish at temperature
< = 17	85	12%	1422	15%
18	112	16%	2151	22%
19	176	25%	2528	26%
20	222	32%	2515	26%
21	46	7%	388	4%
22	111	16%	468	5%
23	26	4%	165	2%
24	5	1%	4	0%
25	1	0%	1	0%
26	0	0%	0	0%
27	0	0%	0	0%

Table 4.5. *Measured surface temperature of pools occupied by adult summer steelhead in the Van Duzen River. Survey years: 2011 – 2019.*

Temperature (°C)	Count of pools at temperature	% of pools at temperature	Sum of fish at temperature	% of fish at temperature
18	3	7%	88	9%
19	9	20%	88	9%
20	3	7%	187	19%
21	1	2%	26	3%
22	9	20%	134	14%
23	7	16%	272	28%
24	10	22%	155	16%
25	1	2%	1	0%
26	1	2%	5	1%
27	1	2%	10	1%

Table 4.6. *Measured surface temperature of pools occupied by adult summer steelhead in the Middle Fork Eel River. Survey years: 2000, 2001, 2004 – 2008, 2010 – 2019.*

Temperature (°C)	Count of pools at temperature	% of pools at temperature	Sum of fish at temperature	% of fish at temperature
17	38	5%	212	2%
18	56	7%	917	10%
19	127	16%	1403	15%
20	182	23%	2029	21%
21	77	10%	1532	16%
22	203	26%	2675	28%
23	83	11%	803	8%
24	15	2%	40	0%
25	2	0%	26	0%
26	1	0%	5	0%
27	0	0%	0	0%

4.1.3 Spawning

Summer steelhead spawning usually occurs during the late winter or early spring in loose gravel typically at the tail of a pool or in a riffle. Substrate is typically small and medium sized gravel ranging from 0.6 to 13 cm in diameter with low embeddedness (Moyle 2002; NMFS 2016b; Moyle et al. 2017). Although steelhead use smaller substrate than other salmonids, fine sediment or substrate that is more than 20 – 25% material smaller than 6.4 mm in diameter, can reduce survival of embryos by preventing the flow of oxygenated water into the egg (Reiser and Bjornn 1979; Barnhart 1986; NMFS 2016b; Moyle et al. 2017) or cause entombment. Water depth in redds ranges between 10 and 150 cm and velocities are typically 20 to 155 cm/sec (Moyle et al. 2017).

4.2 Juveniles

4.2.1 Egg and Larval Development and Fry Emergence

Steelhead eggs are smaller than those of salmon and are typically spawned in spring when water temperatures are increasing (Quinn 2018). Dissolved oxygen (DO) in water is an important aspect of egg development and insufficient DO can be a source of high mortality. DO requirements increase as embryos grow and peak just prior to hatching (Quinn 2018). Intra-gravel oxygen concentration allows for embryo respiration and concentrations of 8 mg/l or more contribute to high survival of steelhead embryos (Reiser and Bjornn 1979). In addition to negative impacts from high amounts of sediment, extremely cold water can induce mortality (Reiser and Bjornn 1979), though this is uncommon for most NC steelhead populations (M. Sparkman, CDFW, pers. comm., July 8, 2020). Fry emerge in late spring or early summer and incubation time is dependent on water temperature (Moyle et al. 2017; Quinn 2018). Temperatures that are too cold or above 21.1°C, can decrease survival of emerging fry by restricting their ability to obtain oxygen from the water (McEwan and Jackson 1996).

4.2.2 Rearing and Emigration

Juvenile steelhead rely on freshwater for at least their first year, although successful juveniles may rear in streams for two years or more (Moyle et al. 2017). Most steelhead mortality occurs in freshwater during the juvenile rearing stage due to lack of good summer rearing habitat; and larger smolt size at ocean entry enhances marine survival, thus, freshwater and estuarine habitats provide vital resources for the species (NMFS 2016b). Suitable rearing habitats require adequate water temperature, flow velocity, water depth, DO concentrations, and prey items. Juveniles generally occupy cool, clear, higher velocity riffles that have cover from predators (Moyle 2002; NMFS 2016b).

Higher growth rates and increased survival are observed when rearing occurs in temperatures between 10°C and 19°C. Less than 15°C is considered optimal (NMFS 2016b; Zedonis and Newcomb 1997). Juveniles can tolerate temperatures up to 29°C if DO concentrations are high and there is an abundant food supply (Sloat and Osterback 2013). In Rancheria Creek in Northern California, 65% of juvenile steelhead moved into nearby stratified pools during times when ambient stream temperatures reached 23 – 28°C (Nielsen et al. 1994). Sparkman (2007) documented juvenile steelhead using areas with cooler water from groundwater seeps that entered mainstem Redwood Creek during summer months when in-river temperatures were high.

Higher flows increase food resources and allow for elevated metabolic demands (McCarthy et al. 2009), although flood flows may disturb or displace prey resources such as larval aquatic insects (Power et al. 2008). California’s Mediterranean climate can result in low flows during summer and late fall rearing, making flow a limiting factor for survival (NMFS 2016b); however, high velocity also increases metabolic demands. The preferred holding velocity for steelhead is 0.19 m/sec, but critical swimming velocity for juvenile steelhead varies with size and is reported to be 7.7 body lengths/sec (Moyle and Baltz 1985; Hawkins and Quinn 1996). In-river habitat with physical objects including boulders, large woody debris (LWD), and undercut banks creates hydraulic heterogeneity and allow for diverse and abundant invertebrate life (Moyle et al. 2017; NMFS 2016b). During high flow periods, juvenile steelhead choose areas with lower velocities such as those created by physical objects, side channels and backwater habitats, floodplains, and pools formed by debris (NMFS 2016b; Moyle et al. 2017). In the Middle Fork Eel River, yearling-plus and young-of-the-year both sheltered behind large rocks despite some seasonal variation in chosen habitat (Ward 1989). In early summer, YOY steelhead were found in calm shallow cobble and gravel areas near currents and as they aged, they moved towards deeper areas with larger substrates and faster currents (Ward 1989). Y+ steelhead had seasonal shifts as well, moving from the head of pools during the summer to spread out across the pools by later summer, and finally leaving pools during daylight hours in September when the preference for feeding overrides the preference for cooler water.

Between March and June, juvenile steelhead transition from fresh water to estuaries and lagoons or to the ocean, though predominating habitat conditions sometimes prevent ocean entry of smolts until late fall (Moyle et al. 2017). Lagoons, which are formed when sandbars create temporary barriers between rivers and the ocean, provide ideal habitat for rearing when the water column is fresh or slightly brackish (NMFS 2016b). This type of lagoon habitat formation is common during the summer in small estuary environments that support NC summer steelhead (Moyle et al. 2017). Open estuaries, unlike lagoons, generally have suitable water quality conditions throughout the year (NMFS 2016b). When successful formation occurs, sandbar built estuaries may provide diversity of water depths and pools, secondary channels, and marsh plains that provide productive rearing habitat for juvenile steelhead (Clark and O'Connor 2019). These transitional areas allow juveniles to move into salt water and maintain faster growth than they would achieve in upstream habitats prior to ocean entry (NMFS 2016b). Sand bars can sometimes prevent ocean entry if they are not breached soon enough. This has been seen in Redwood Creek where juveniles appear to perish over time with the formation of a sand bar (M. Sparkman, CDFW, pers. comm., July 9, 2020).

In some areas, including Scott Creek watershed, it was found that higher percentages of returning adult steelhead are fish that reared in the lagoon even though they may represent a small proportion of the overall downstream migrating population (Hayes et al. 2008). Lagoon formation depends on environmental conditions, water diversions, and sandbar formation. For example, in Scott Creek (central California) lagoons suitable for rearing are not always present, which leads to variation in the percentages of juveniles rearing in the lagoon versus in the river between years. But, in years when lagoons formed, models comparing juvenile growth in Scott Creek found that growth in the warm (up to 24°C) estuary-lagoon was higher than in the upper-watershed or upper-watershed and estuary-lagoon combined (Hayes et al. 2008). Growth rates of steelhead reared in this more productive habitat were two to six times higher than growth rates of fish in cooler, less productive upstream habitat; however, late formation of sandbars could reduce this productivity. Additionally, streams with estuaries in poor condition may not support accelerated growth rates.

Within the range of the NC steelhead DPS, Redwood Creek, Mad River, and Mattole River have been classified as lagoonal estuaries (Heady et al. 2014), however, steady flows from Matthews Dam prevent sand bar formation in the Mad River estuary (M. Sparkman, CDFW, pers. comm., July 9, 2020). In most of the watersheds in the NC steelhead DPS, stream flow during the dry season is low and sandbar formation occurs later in the summer. In these cases, lagoons don't transition to freshwater or form stratified water columns, which leads to lower productivity in the lagoon (NMFS 2016b). The Redwood Creek estuary is also known to be in poor condition and does not provide suitable habitat for juvenile rearing (M. Sparkman, CDFW, pers. comm., July 9, 2020). The Mattole River, however, where marshes are successfully flooded for longer periods in the summer, allows for increased hydraulic connectivity between the main channel and back water environments provides better habitat for rearing juveniles (NMFS 2016b).

4.3 Ocean Growth

Poor ocean conditions likely affect survival especially during early ocean residence (Kendall et al. 2017). Interdecadal climate variability in the northeast Pacific Ocean can limit biological productivity. Steelhead smolts tend to migrate offshore more rapidly compared to other salmonids, which spend time along the continental shelf (Daly et al. 2014; Quinn 2018). Many California steelhead juveniles spend only a few months feeding in the California Current Large Marine Ecosystem (CCE) before they migrate northwest to cooler waters offshore (Daly et al. 2014). However, in a trawl study conducted off Oregon and Northern California, steelhead were found consistently through August off the Northern California coast (Hayes et al. 2016). This was an unexpected outcome, but the authors found that the temperature regime in that area was cooler for a longer period of time compared to other areas of the CCE and potentially served as thermal refugia for steelhead, which would explain their presence. 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 as a result of their quick movement offshore, and most catches of steelhead in research trawls are in the upper 30 m of the water column (Moyle et al. 2017; Quinn 2018). While in the ocean, steelhead feed on pelagic organisms such as krill, fish, and amphipods (Moyle et al. 2017). While feeding on the coast, steelhead typically stay in temperatures ranging from 8 to 14°C. With the exception of half-pounders, steelhead will stay in the ocean for two to three years before returning to their natal rivers to spawn (Quinn 2018).

5. FACTORS AFFECTING THE ABILITY TO SURVIVE AND REPRODUCE

5.1 Changes in Ocean Conditions

There are various indices that describe the fluctuations in ocean conditions and can help determine years during which Pacific salmonids will experience a productive ocean and those during which they will experience a less productive ocean. These indices include the Ocean Niño Index (ONI), Pacific Decadal Oscillation (PDO), and the North Pacific Gyre Oscillation (NPGO). The ONI, which tracks average sea surface temperature and describes the El Niño/Southern Oscillation (ENSO) climate pattern, and the PDO, which describes annual North Pacific sea surface temperature (SST) anomaly patterns, are closely correlated (Mantua et al. 1997). Positive values of ONI and PDO indicate warmer temperatures in the ocean. The NPGO is highly correlated with decadal variations in salinity, nutrients, chlorophyll, various types of zooplankton, and fish species in the Northeastern Pacific Ocean (Di Lorenzo et al. 2008). Negative NPGO values indicate wind patterns that create conditions unfavorable to upwelling, which leads to reduced nutrients and chlorophyll, decreasing primary productivity. Over the past few years, 2014 – 2019, the NPGO has declined to near historic lows (NOAA 2019, 2020), indicating a recent trend of weak circulation, low influx of nutrient-rich water, and low primary productivity (Figure 5.1) (NOAA 2019). These trends indicate poorer ocean conditions for salmonids, including NC summer steelhead, and can negatively influence steelhead growth and survival to adulthood (Kendall et al. 2017).

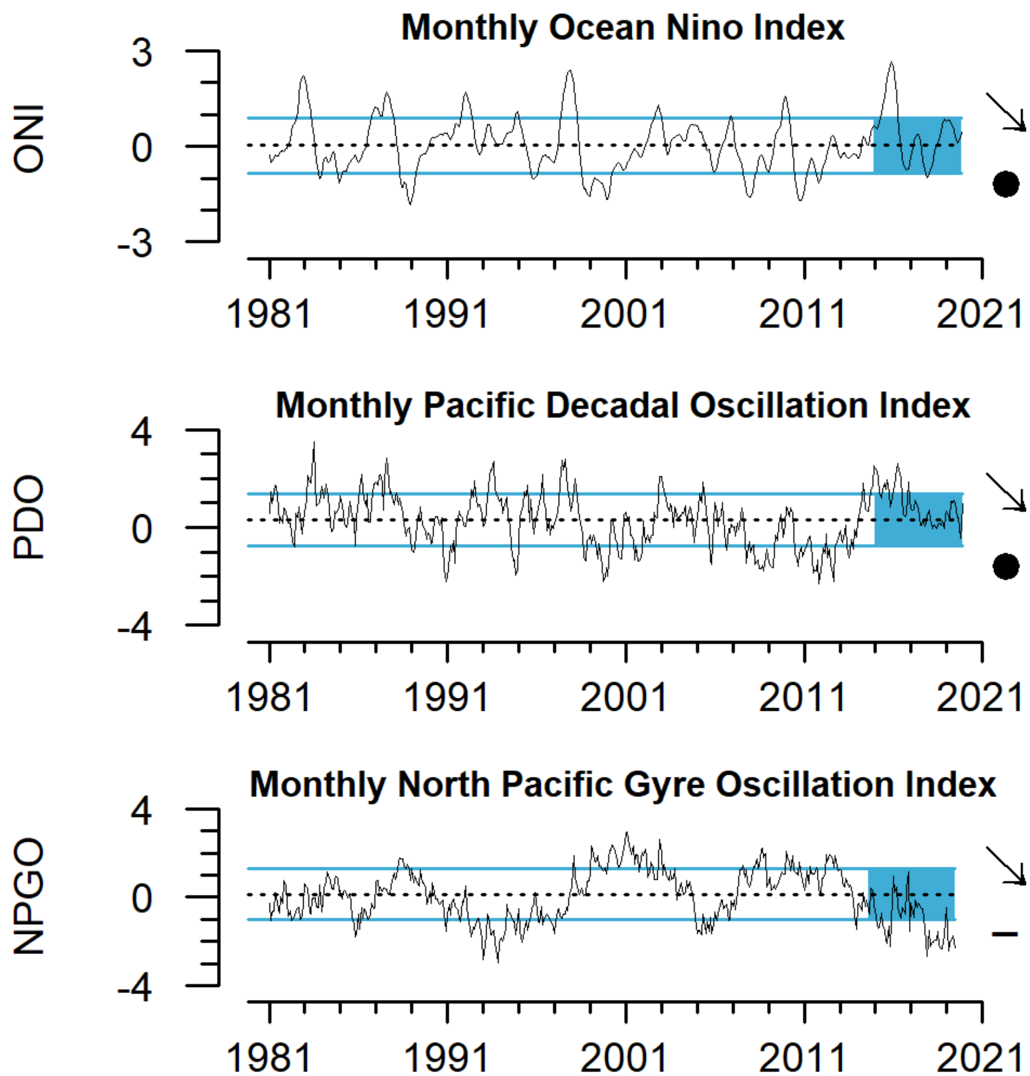


Figure 5.1. *Monthly values of the ONI, PDO, and NPGO from 1981- 2019. Dashed lines represent the mean over the time series, and solid lines indicate one standard deviation. Arrow at the right indicates if the trend over the evaluation period (shaded blue) was positive, negative, or neutral (from NOAA 2020).*

Changes in the thermal regime due to El Niño and La Niña events greatly impact primary and secondary productivity by limiting upwelling and nutrient replacement on the Pacific coast of North America (Pearcy 1992 as cited in Brown et al. 1994). El Niño is characterized by warmer than usual ocean temperatures that affect the central and east central Equatorial Pacific (National Ocean Service 2020). La Niña has the opposite effect of El Niño, with unusually cold ocean temperatures in the Equatorial Pacific (National Ocean Service 2020). Changes in surface currents and upwelling strength will impact temperature, salinity, and nutrient availability, thereby influencing the availability of food for juvenile salmonids (Roesler and Chelton 1987 as cited in Spence et al. 1996), predation rates (Holtby et al. 1990), and the transport of smolts either along-shore or off-shore upon ocean entry (Francis and Sibley 1991 as cited in Spence et al. 1996).

The cyclical nature of changes in ocean condition may mask salmonid population declines due to an unfavorable freshwater environment. The long-term decline of salmonid populations are not linear in nature and thus, a negative trend can be obscured by improved escapement as the result of favorable ocean conditions. Lawson (1993) demonstrated this in a conceptual model looking at freshwater habitat quality, marine environment, and Coho Salmon escapement (Figure 5.2).

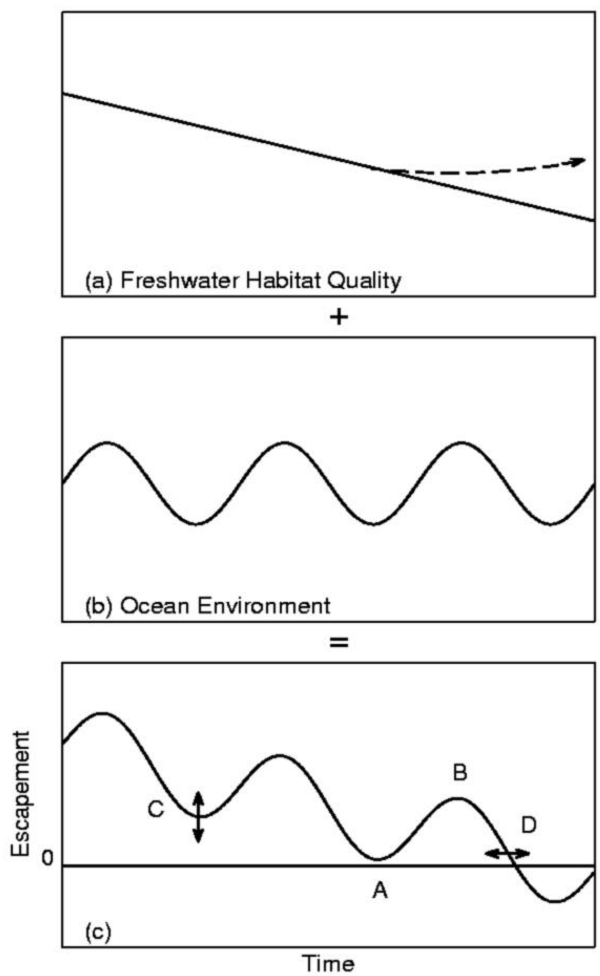


Figure 5.2. *Conceptual model of effects of declining habitat quality and cyclic changes in ocean productivity on the abundance of Oregon's coastal natural Coho Salmon (from Lawson 1993). Top panel shows trajectory of habitat quality over time, with the dotted line representing potential effects of habitat restoration projects. Middle panel shows the cyclic nature of ocean productivity. Bottom panel demonstrates the combination of top two panels with letters that signify the following: A = current situation (at time of publication), B = future situation, C = change in escapement due to increasing or decreasing harvest, and D = change in time of extinction due to increasing or decreasing harvest.*

Regional climate indicators are also important when looking at ocean temperatures and nutrient content in California's coastal waters. Upwelling is a process that drives the seasonal primary productivity that supports the CCE food web by bringing nutrients to surface areas (NOAA 2019). These nutrients promote the production of phytoplankton, which provide food for zooplankton. A novel way of determining coastal upwelling was developed by Jacox et al. (2018) using the Coastal Upwelling Transport Index (CUTI) to estimate vertical water transport, and the Biologically Effective Upwelling Transport Index (BEUTI) to estimate nitrate flux (Jacox et al. 2018). In conjunction, these indices track the volume and quality (in terms of nutrients) of water, respectively, moving through the surface layer of the ocean, which have strong influences on productivity (Jacox et al. 2018). Typically, the peaks in upwelling occur in late July near Newport, OR, mid-June off of Point Arena, CA, and late April near San Diego, CA. Upwelling off of Point Arena tends to be greater than in the other two areas by an order of magnitude (NOAA 2020). In 2019 (Figure 5.3), BEUTI values were generally average or above average coastwide and during all seasons except for some below-average periods of nitrate flux at Point Arena, CA during winter and spring (NOAA 2020). CUTI values were average or above average during winter and spring in the northern CCE and during the summer in the central and southern areas (NOAA 2020). Central and southern CUTI values during the winter and spring, as well as northern CUTI values during the summer, were around or below average (NOAA 2020).

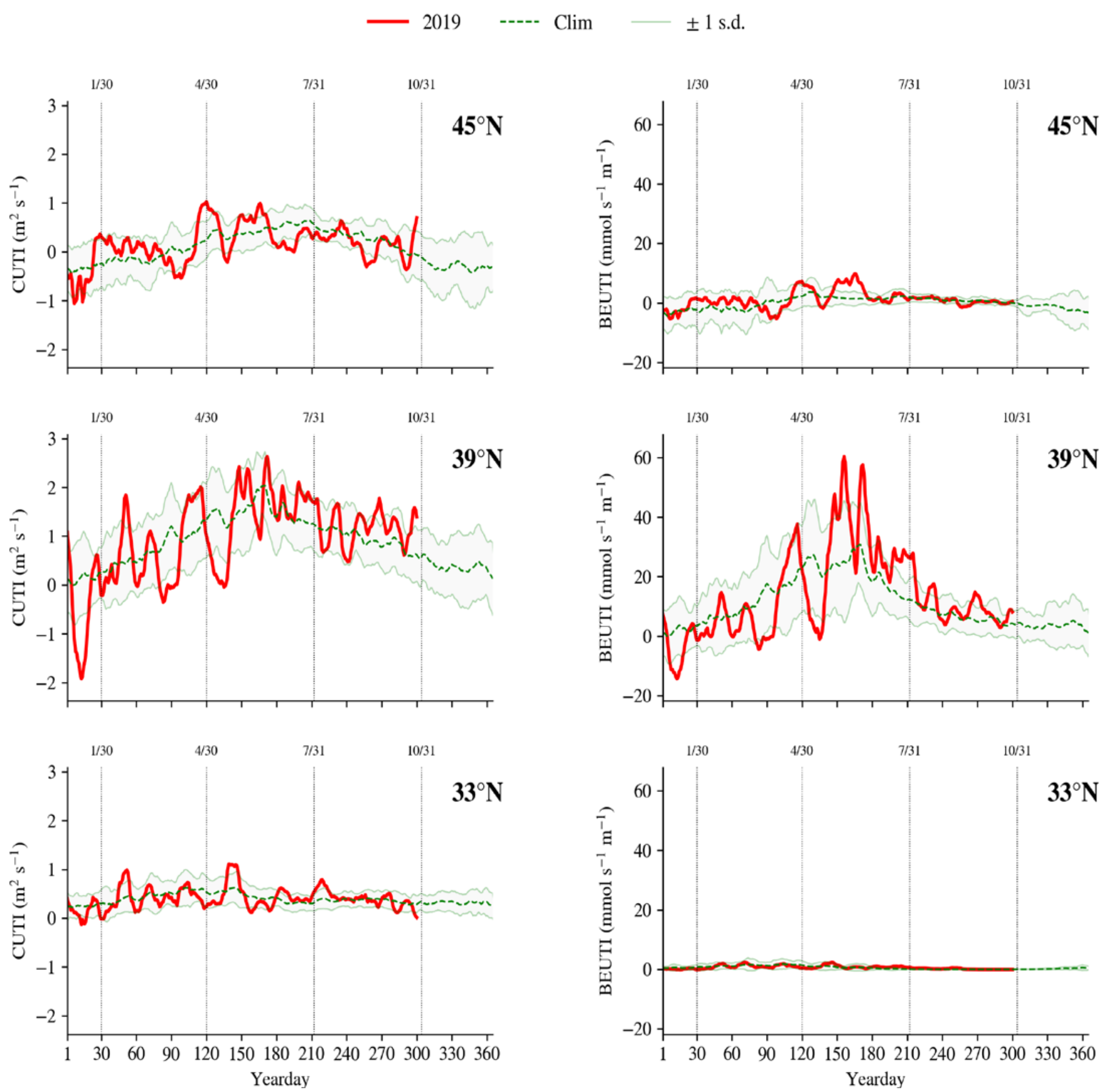


Figure 5.3. *Daily 2019 values of Biologically Effective Upwelling Transport Index (BEUTI; left) and Coastal Upwelling Transport Index (CUTI; right) from Jan. 1 – Sept. 1, relative to the 1988 – 2019 climatology average (green dashed line) ±1 standard deviation (shaded area), at latitudes 33° (San Diego, CA), 39° (Point Arena, CA), and 45°N (Newport, OR). Vertical lines mark the end of January, April, July, and October (from NOAA 2020).*

Copepod biomass and composition are also indicative of productivity. There are two distinct groups of copepod taxa present in the CCE: northern copepods and southern copepods. Northern copepods are found in cold water environments and are high in wax esters and fatty acids, which are essential to the diet of pelagic fish (NOAA 2019). Northern copepods are indicative of La Niña conditions and a more productive ocean environment. Southern copepods are warm water zooplankton that have lower fat content and less nutritional value. Southern copepods are indicative of El Niño events and positive PDO values when productivity decreases.

Forage or bait fish, which feed primarily on marine copepods (Osgood et al. 2016), are a primary source of prey for steelhead at sea, thus copepod biomass and composition have a significant effect on steelhead prey availability, growth, and survival (LeBrasseur 1966; Quinn 2018). During 2014 – 2016, anomalous warm ocean temperatures resulted in a predominance of southern copepod species. From 2017 – 2018, northern copepod biomass rebounded from negative values to neutral and southern copepod biomass decreased to negative values (NOAA 2019). This trend continued in 2019 (Figure 5.4, NOAA 2020), indicating a possible improvement for foraging conditions for pelagic baitfish.

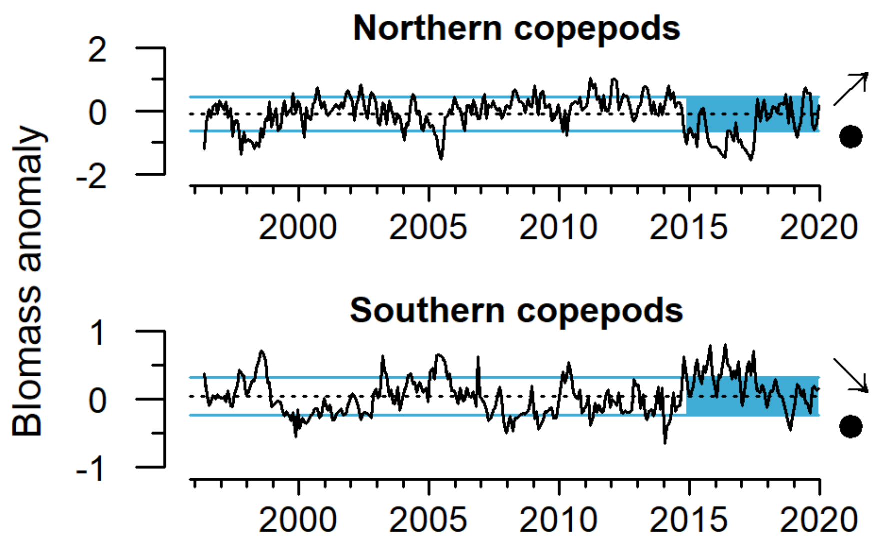


Figure 5.4. *Monthly northern and southern copepod biomass anomalies from 1996-2019. Dashed lines represent the mean over the time series, and solid lines indicate one standard deviation. Arrow at the right indicates if the trend over the evaluation period (shaded blue) was positive, negative, or neutral (from NOAA 2020).*

Over the past few decades there have been greater fluctuations in ocean ecosystem indicators in relation to salmon survival, putting greater strain on populations experiencing low abundance and limited habitat (Lindley et al. 2009). A poor ocean environment in 2005 and 2006 resulted in unusually poor survival for brood years 2004 and 2005 of Sacramento River fall Chinook Salmon and was the proximate cause of the stock collapse in 2008 (Lindley et al. 2009). Following this collapse, salmon populations rebounded for a few years, but in 2018, five west coast salmon stocks, including two of California's most prominent Chinook Salmon stocks, Sacramento River and Klamath River fall Chinook Salmon, were determined to be in peril due to a combination of factors in both fresh and saltwater including a warm, unproductive ocean in 2014 through 2016 that compromised marine survival of brood years 2013 – 2015 (PFMC and NMFS 2019; Fisheries Off West Coast States; West Coast Salmon Fisheries; Rebuilding Chinook Salmon Stocks, 2020). While 2018 saw improved ocean conditions for salmon, 2019 ocean ecosystem indicators again ranked warmer. Although northern copepod biomass remained quite high in 2019, other indicators suggested poorer conditions for juvenile salmon entering the ocean (Peterson et al. 2019). It is likely that effects of these more extreme fluctuations in the marine environment on survival of California's salmon stocks are similarly affecting steelhead survival. Kendall et al. (2017) posited, after examining 48 coastal steelhead populations, that smolt survival rates to adulthood were positively correlated for populations whose river mouths were geographically proximate, which is an indication that processes related to steelhead ocean survival occur early in their marine life. Welch et al. (2000) describe changes in smolt to adult survival that were most likely of result of decreasing marine survival attributable to changing ocean conditions. Friedland et al. (2014) found the return rate of adult steelhead to be negatively correlated with SST in areas which were thought to act as postsmolt rearing habitat, thus, steelhead growth may be directly influenced by ocean warming or indirect effects of ocean warming such as changes to the marine food web.

A new study by Thalmann et al. (2020) discovered significant differences in prey items consumed by Columbia River juvenile steelhead during warm ocean years as compared to during average or cold ocean years. A higher percentage of unidentified or rare fish composed the juvenile steelhead diet during warm years as well as juvenile and larval rockfishes and insects, whereas during cold years polychaete worms and krill were more common. Thalmann et al. (2020) 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. Bioenergetics simulations revealed significant differences in growth rate during warm and cold ocean conditions. Cooler temperatures and higher feeding rates supported the highest juvenile steelhead growth rates. 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. So, although a degree of plasticity was demonstrated in 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).

5.2 Effects of Climate Change

The Earth's climate is warming, and the primary causes are greenhouse gas emissions and deforestation (IPCC 2007; USGCRP 2009; USGCRP 2017). A warming climate is likely to result in poorer future environmental conditions for salmonids in general, and for NC summer steelhead specifically. These impacts include average temperature increases, increased precipitation over a compressed annual time period, increased magnitude and length of drought periods, decreased snowpack, more frequent and severe wildfires, lower dry season stream flows, and greater severity of floods (Grantham 2018). Climate change impacts on steelhead in the ocean may include sea level rise, and more specifically the negative effects of sea level rise on estuary habitat, and ocean acidification (Crozier et al. 2019). A more detailed description of climate change impacts can be found below.

5.2.1 Rising Temperatures

One of the greatest threats imposed by climate change on NC summer steelhead will likely be rising stream temperatures and indirect effects associated with this trend. Isaak et al. (2018) modeled trends in water temperature of rivers in the northwestern United States and found warming trends of mean annual river temperatures over the last two decades. Although annual warming trends over the past four decades were generally milder, especially high rates of warming occurred during the summer and early fall. Isaak et al. (2018) predicted an 8% decrease in the length of thermally suitable stream reaches resulting from a river temperature increase of 1.0°C. Decreases of 18 – 31% of suitable habitat could occur if more drastic temperature increases are realized. The largest decreases in thermally suitable habitat were forecast for Oregon and Northern California streams. Results of this study suggested increasing temperatures may contract ranges of resident trout, forcing these species to find limited cold water refugia further upstream. This range reduction could be exacerbated by a wider distribution of predators that prefer warmer water (Isaak et al. 2018). Summer steelhead may experience similar effects. Over several decades Northern California has experienced rising summer and fall air temperatures resulting in warmer stream temperatures during summer months. Higher summer water temperatures are more stressful for summer-rearing salmonids and over-summering adult summer steelhead (Madej 2011) and could contribute to lower survival and increased mortality of these fish.

5.2.2 Flooding

Floods are typically caused by heavy precipitation over a relatively short period of time and can have detrimental effects on all life stages of salmonids. High flows associated with intense flooding have the potential to scour or bury redds, causing reduced salmonid egg survival (Elwood and Waters 1969; Zimmerman et al. 2015; Sparkman 2017). Severe flooding can also cause high mortality of juvenile life stages of fish (Elwood and Waters 1969; Jowett and Richardson 1989) and geomorphic changes to river channels results in destruction of suitable habitat. Floods can also remove invertebrate populations, eliminating vital food resources (Elwood and Waters 1969). Floods can have long-lasting impacts on watershed conditions and require decades or longer to recover, especially in systems already impaired by other processes or activities.

The 1955 and 1964 floods had a devastating effect on watersheds in California's north coast leaving them highly degraded. Impacts, especially those of the 1964 flood, were exacerbated by anthropogenic factors, most notably logging operations, road construction, and livestock grazing on unstable slopes (Moyle 2002). Massive loads of sediment were deposited into the streams, resulting in the filling of pools, widening of channels, and destruction of riparian vegetation that mitigated rising water temperatures, provided habitat for insects, and was used by summer steelhead for shelter. Although this gravel deposited in Northern California streams has been scoured out over time, it has taken decades to recover and the process is ongoing.

These streams, especially those in the Eel River system, have the potential to be hit by future flooding events that could again devastate the landscape by way of mass wasting (Moyle 2002). Timber harvest activity is still present in this area and logging of steep forested hillsides along with impacts associated with wildfires will further impair the stability of the terrain (Moyle et al. 1995).

Predictions for annual precipitation in California's north coast show a slight increasing trend by the end of this century (Grantham 2018). This region in California is already the location of the highest intensity storms in the state (Grantham 2018). It has also been hypothesized that the duration of California's wet season will be shortened but have greater intensity and total precipitation (Swain et al. 2018). Changes to the natural hydrograph can cause complications for salmonids in all life stages. Rainfall that occurs less spread out over the season and of higher intensity has the potential to result in more catastrophic flooding, which will further damage stream geomorphology and negatively impact aquatic species survival in north coast streams.

5.2.3 Drought

Drought is not a new climate feature in California. Tree ring chronologies show that multi-year droughts have plagued California over the last millennium (Griffin and Anchukaitis 2014). During periods of drought, stream flows may drop, constricting or fragmenting available habitat (Spence et al. 1996). Reduced flows can cause water temperature increases which can result in heat stress to fish, direct mortality to juvenile steelhead (Sparkman 2007), and changes in adult migration behavior (Robards and Quinn 2002; High et al. 2006).

The most recent 2012 – 2016 drought was one of the warmest and driest on record, affecting both aquatic and terrestrial environments across the state (Figure 5.5; CDFW 2018). In the Northern Region of California, which includes Del Norte, Humboldt, Mendocino, Lassen, Modoc, Shasta, Siskiyou, Tehama, and Trinity counties, average water temperatures were found to be much more extreme as compared to climate change projections. The year 2015 was particularly bad. Water temperatures during the drought exceeded survival thresholds of salmon and steelhead in coastal watersheds, likely leading to higher mortality rates of returning adults as well as their offspring (Sparkman 2007; CDFW 2018). Primary impacts in north coast streams were high water temperatures, at or approaching lethal levels for salmonids, and low instream flows (CDFW 2018). Statewide effects of the 2012 – 2016 drought included early drying out of streams, poorer water quality of estuaries and bar-built lagoons, critically high water temperatures, and low streamflow leading to poor water quality and stranding of fish (CDFW 2018).

Several patterns during this most recent extreme drought were documented throughout the state. These patterns included: 1) streams drying earlier in the season and for longer amounts of time, 2) estuaries and bar-built lagoons experiencing poorer water quality, 3) water temperatures often rising to levels detrimental to salmon and steelhead survival, 4) development of winter anchor ice threatening high elevation populations of wild trout, and 5) low streamflow leading to stranding of fish as well as poor water quality (CDFW 2018).

Studies have shown that drought conditions are likely to become more frequent and more intense as a result of rising temperatures (Wang et al. 2014; Diffenbaugh et al. 2015; Wehner et al. 2017). It is likely that NC summer steelhead populations will suffer with more frequent and severe droughts.

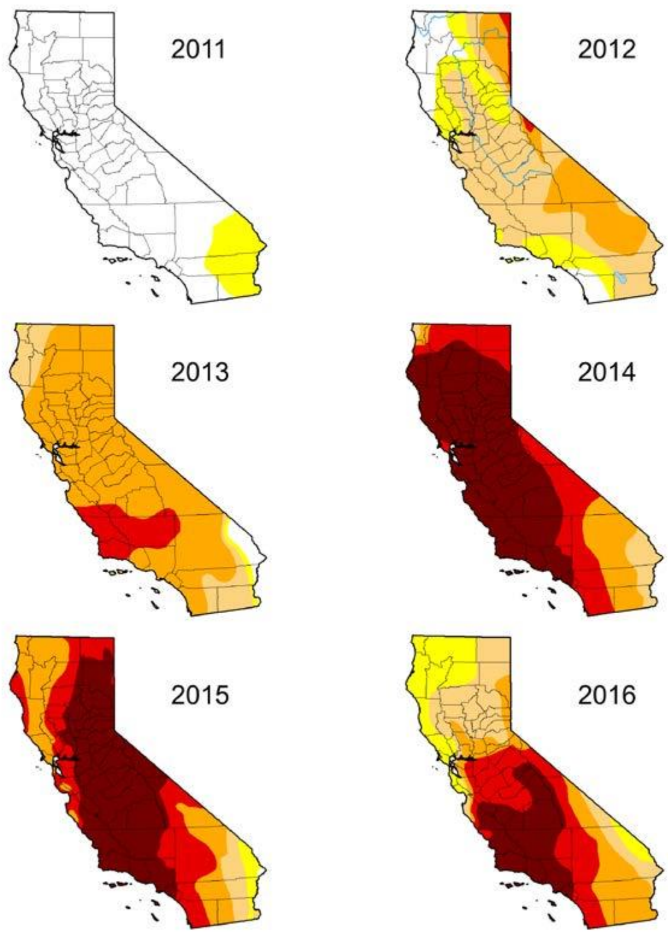


Figure 5.5. *The distribution and progression of drought conditions in California from 2011 to 2016, depicting the level of drought at the beginning of each Water Year (October 1). Dark Red indicates exceptional drought (CDFW 2018 based on U.S. Drought Monitor).*

5.2.4 Reduced Snowpack

Decreased snowpack has been forecast for northern coastal California (Grantham 2018). Battin et al. (2007) investigated the implications of global climate change projections on Snohomish River Chinook Salmon population dynamics. They found that the highest elevation basins that are fed substantially by snowmelt will experience the greatest impacts to their hydrography, and declines of salmon in those areas will be more pronounced than in lower elevation sub-basins (Battin et al. 2007). This finding has implications for summer steelhead, which are generally found in higher elevation streams driven by snowmelt hydrology. Department biologists working in the Eel River system generally agree that the gradual decline of the natural hydrograph during the late spring driven by snowmelt is the key factor that allows summer steelhead to ascend into upstream reaches of the Middle Fork Eel and Van Duzen rivers with adequate temperatures for summer holding. Without this gradual decline, summer steelhead likely would not be able to pass barriers at Osborn Roughs on the Middle Fork Eel River or the cascade at Salmon Hole on the Van Duzen River, forcing them to withstand temperatures that may not be cold enough to sustain them through the summer (S. Thompson, CDFW, pers. comm., July 14, 2020).

5.2.5 Wildfires

With the prospect of reduced snowpack, increasing temperatures, and growing human populations, consensus is that wildfire seasons on the north coast will lengthen and fires will become more frequent (Westerling et al. 2006; Micheli et al. 2018). Wildfires remove riparian vegetation, reduce LWD availability, and increase erosion and soil instability leading to more sedimentation, all of which reduce habitat suitability for steelhead. In August 2020, a fire spread through portions of Mendocino, Six Rivers, and Shasta-Trinity National Forests, including areas of upper Middle Fork Eel River adult summer steelhead holding habitat. Effects on habitat, water quality, and summer steelhead survival are currently unknown (A. Renger, CDFW, pers. comm., September 17, 2020).

5.2.6 Reduced Stream Flows

Regardless of the predicted increase in annual precipitation, summer baseflows in some Northern California streams are decreasing likely as a result of higher rates of evapotranspiration due to climate change (Sawaske and Freyberg 2014 as cited in Grantham 2018). Models developed by Grantham et al. (2018) predicted increases of stream flow during wet seasons and decreases during dry seasons over the next few decades in California. Reductions in streamflow during the dry months can lead to increased water temperatures, drying of streambeds, increased primary production and algal blooms, off-channel pond disconnection and subsequent fish stranding, and the inability of migrating adults to pass flow-dependent barriers. Consequences could be severe for adult NC summer steelhead, which rely on deep, temperature stratified pools to hold during the summer prior to spawning.

5.2.7 Ocean Acidification

Ocean acidification is a result of increased carbon dioxide absorption from the atmosphere, which lowers pH of the water. Global ocean pH is projected to increase due to current and future carbon emissions (Caldeira and Wickett 2005). Ocean acidification could alter marine food webs and impact steelhead food resources. 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. 2013). 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 steelhead. Increased carbon dioxide in freshwater may also have detrimental effects on growth, olfaction, and predator avoidance in juvenile Pink Salmon (*O. gorbuscha*)(Ou et al. 2015). Similarly, higher levels of carbon dioxide dull olfactory-mediated behavioral responses of Coho Salmon due to changes in neural signaling (Williams et al. 2019).

5.2.8 Sea Level Rise

Sea level rise is predicted to change the dynamics of estuaries and near shore environments. Rising sea levels may also affect estuary hydrodynamics with saltwater intrusion, and as a result, increased salinity in estuaries is likely to become more of an issue (Glick et al. 2007). Loss of estuary habitat due to sea level rise may affect salmonid survival and productivity, as estuaries serve as important nursery habitat for juveniles (Glick et al. 2007; Crozier et al. 2019). Rates of sea level rise for Humboldt Bay are projected to be the highest in California according to recent estimates (Patton et al. 2017). This means that the Humboldt Bay area will likely be affected by sea level rise sooner than other areas of the California coast (Anderson 2018). Sea level rise may increase the area of estuarine habitats, which would increase habitat availability for rearing in some places.

5.2.9 Conclusions

Climate change impacts have many implications for salmonids including NC summer steelhead and are likely to exacerbate other threats that are already present. The cumulative effects of both natural and anthropogenic factors that contribute to severe sedimentation, altered flow regimes, pollution, changes to stream morphology, decreased habitat suitability, and habitat destruction, will be amplified by the consequences of a changing global climate. As a result, NC summer steelhead will likely experience accelerated habitat loss and degradation, reducing the availability of suitable habitat and limiting their ability survive and reproduce throughout their range.

5.3 Disease

A myriad of diseases caused by bacterial, protozoan, viral, and parasitic organisms can infect steelhead in both the juvenile and adult life stages. These diseases include bacterial kidney disease (BKD), *Ceratonova shasta*, columnaris, furunculosis, infectious hematopoietic necrosis virus (IHNV), redmouth and black spot disease, erythrocytic inclusion body syndrome, and whirling disease (NMFS 2011). Though steelhead have coevolved with some of these microorganisms, the introduction of non-native hatchery stocks has exposed native populations to foreign pathogens not historically found in the watershed (NMFS 2011). Natural-origin steelhead have been shown to be less vulnerable to infection by pathogens than hatcherypropagated steelhead (Buchanan et al. 1983 as cited in NMFS 2011), thus reducing the likelihood of disease outbreaks in wild steelhead populations (NMFS 2016b). *C. shasta*, a myxosporean protozoan, has posed substantial issues in the Klamath River basin particularly for Chinook Salmon (Fujiwara et al. 2011); however, steelhead have been shown to have an innate resistance to *C. shasta* and are generally not infected by the parasite (Stone et al. 2008). Warming stream temperatures can also increase infection rates as a result of several factors (Crozier et al. 2008).

Although Department pathologists have observed several types of endemic bacterial infections and external parasites in MRH steelhead, treatment of water entering the hatchery with ultraviolet light has significantly diminished disease and pathogen issues (CDFW 2016). Yearling fish are also inspected prior to their release to ensure they are healthy before entering the natural environment. (CDFW 2016). Given the limited hatchery production of steelhead in Northern California, disease attributed to hatcheries likely has a negligible effect on summer steelhead populations and is not thought to be a major threat to steelhead in the Northern California DPS (NMFS 2016b).

5.4 Hatcheries

5.4.1 Mad River Hatchery Winter Steelhead

The Mad River Hatchery winter steelhead program has been in operation since 1971 and using broodstock collected from the South Fork Eel River through 1973. The hatchery also introduced 37,000 and 20,000 Russian River steelhead fry in 1984 and 1985, respectively (CDFW 2016). Given how few fish were planted, it is doubtful these more recent events affected the Mad River steelhead population structure in the long term (Spence et al. 2008). By 1974, returns to MRH were enough to supply the hatchery's production needs and all subsequent broods have been produced from steelhead returning to the hatchery (Cramer et al. 1995). However, beginning in 2017, the Department integrates natural-origin winter run steelhead into the breeding program at a 1:1 ratio with adipose marked hatchery fish (CDFW 2016).

In addition to releases into the Mad River, MRH steelhead were outplanted as part of a "Coastal Steelhead Planting and Release Program" in the mid-1970s (Will 1976 – 1978 as cited in Cramer et al. 1995). This program included plantings in the Smith, Eel, Garcia, Gualala, Trinity, Klamath, and Van Duzen rivers and Lagunitas Creek (Will 1976 – 1978, Kelley et al. 1987 as cited in Cramer et al. 1995; Yoshiyama and Moyle 2010). Steelhead from the Eel River and Redwood Creek were also planted in Lagunitas Creek (CDFG 1983 as cited in Cramer et al. 1995). Plantings after broodyear 1980 were only allowed in the basin where the hatchery was located (Cramer et al. 1995). The discontinuation of outplanting should have curtailed impacts of out-of-basin hatchery fish on wild populations. McEwan and Jackson (1996) noted that steelhead eggs have been collected at Van Arsdale Fish Station, reared at MRH, and released as yearlings back into the upper Eel River. According to Steiner Environmental Consulting (1998) as quoted in

Yoshiyama and Moyle (2010), planting of Van Arsdale steelhead stock raised at MRH continued from 1965 through 1995.

Mad River Hatchery has released about 150,000 winter steelhead smolts annually since 2009 (Moyle et al. 2017). MRH steelhead could have detrimental genetic effects on natural-origin NC steelhead DPS range as a result of inbreeding (NMFS 2008 as cited in CDFW 2016). Interbreeding of hatchery- and natural-origin stocks can reduce fitness and productivity of the natural stock. Hatchery fish are not exposed to the same natural selection pressures that control fitness in the wild and are subjected to a different selective environment in the hatchery setting (CDFW 2016). Altered genetic composition and phenotype, affecting characteristics such as adult size, smolt age, emigration timing and size, fecundity, and egg size, are consequences of hybridization of hatchery fish with natural-origin salmonids, especially when they are genetically dissimilar (Spence et al. 2008). These effects can eliminate local adaptation of a wild population, which is key to the viability of that population (Spence et al. 2008). As reviewed and analyzed by Christie et al. (2014), multiple studies have found that hatchery-produced salmon and steelhead have lower reproductive success and reduced fitness due to hatchery rearing even when predominantly wild-origin broodstock was used (Araki, Ardren, Olsen, Cooper, Blouin 2007; Araki, Cooper, and Blouin 2007; Berntson et al. 2011; Thériault et al. 2011; Hess et al. 2012; Ford et al. 2013; Milot et al. 2013). Araki et al. (2009) found that wild-born Hood River steelhead with two hatchery-origin parents had lower reproductive fitness than steelhead with two wild parents and fish with one hatchery-origin and one wild parent had intermediate reproductive fitness. These results suggest that hatchery supplementation could have cumulative negative effects on wild population abundance and fitness that are not only environmental but also genetic (Christie et al. 2014).

Recent genetic analysis shows that historical Mad River winter steelhead collections from the 1970s clustered with Eel River steelhead due to the use of Eel River broodstock in early hatchery operations (Fong 2020). It seems; however, that the Eel River steelhead were never permanently established since contemporary Mad River steelhead populations are more closely related to steelhead from Redwood Creek than those from the Eel River, following the isolation by distance model. Contemporary MRH broodstock are somewhat diverged from historical Mad River collections possibly as a result of hatchery management practices like having few effective breeders and, until recently, not incorporating natural-origin broodstock (Fong 2020).

Through the new Hatchery and Genetic Management Plan (HGMP), MRH has implemented measures to counteract potential negative genetic effects. The hatchery now aims to incorporate natural-origin winter steelhead in their broodstock at a 1:1 ratio with marked hatchery fish to achieve a proportionate natural influence (PNI) of at least 0.5 (CDFW 2016). Implementing a requirement of 0.5 PNI for hatchery broodstock is a significant improvement to hatchery operations; however, it does not immediately reverse genetic impacts already accrued through previous hatchery propagation.

Hatchery winter steelhead likely do not interact frequently with summer steelhead on the Mad River. The MRH HGMP states that because yearlings are planted directly in the river during flow surges, it is unlikely that the fish travel back upstream, rather, they rapidly make their way to the ocean in large schools (Flagg et al. 2000; CDFW 2016). They concluded that any mixed-stock interactions likely only occur downstream of the hatchery; however, the possibility of interbreeding between summer steelhead and MRH-produced fish remains. Leider et al. (1984) suggests that although observed reproductive overlap was low between wild summer and winter steelhead in the Kalama River basin, Washington, there was likely a small amount of gene flow due to spatial and temporal overlap of distribution of spawning fish (Leider et al. 1984). Evidence of heterozygosity at the GREB1L locus associated with salmonid migration timing, as discussed in *Section 2.6, Genetics and Genomics* suggests that winter and summer steelhead within the same river basin do interbreed frequently enough to be reflected in their genotypes (Prince et al. 2017; Pearse et al. 2019; Kannry et al. 2020).

MRH steelhead likely have a greater impact on natural-origin winter steelhead than they do on summer steelhead in the Mad River given their spatial and temporal overlap during spawning. The same is likely to be the case in other NC Steelhead DPS streams that may receive MRH steelhead strays. The concept of adult salmonid straying has become a widespread concern of salmonid biologists and managers. The magnitude of the effects of hatchery-origin strays on natural populations depend on the number of donor strays and the size of the natural-origin recipient population (Grant 2012; Bett et al. 2017). Straying of hatchery fish into non-natal streams, especially those with small populations, can have the potential to significantly affect the genetic composition of recipient populations and loss of local adaptations (Grant 2012).

Direct genetic effects of hatchery production on wild fish can include: 1) reduction in between-population genetic variance, and 2) outbreeding depression (Waples 1991a). Hybridization, though it can increase average genetic diversity within the hybridizing populations, decreases genetic diversity between those populations. Maintaining genetic and phenotypic diversity between different populations is essential in buffering the effects of unexpected or periodic changes in the environment on steelhead productivity and survival (Riggs 1990 as cited in Waples 1991a). In addition to the risk of homogenization and domestication, outbreeding depression can also be a consequence of hybridization between populations. Outbreeding depression is a decrease in fitness within a population as generations of offspring become more genetically distant from the parental stocks in the initial hybridization event (Waples 1991a). This occurs when a locally adapted wild gene pool becomes swamped by genes from divergent hatchery fish. NMFS (2016b) had classified the MRH program as a high threat to both winter and summer steelhead due to concerns over outbreeding depression and reduced productivity (NMFS 2016b). The recently issued HGMP attempts to mitigate negative impacts from the hatchery.

Indirect genetic effects of hatchery production on wild fish can include: 1) reduced population size, and 2) altered selection regimes (Waples 1991a). Artificial propagation can lead to decreases in natural-origin population sizes as a result of increased competition, predation, and disease (Steward and Bjornn 1990), although sometimes direct negative effects of hatchery propagation on natural-origin population size may be difficult to prove (Courter et al. 2019). Significant reductions in population size over a short period of time can pose a high risk of extinction from serious environmental changes. Long-term effects of reduced population size can include genetic bottlenecks, although these must last for multiple generations for significant inbreeding to transpire (Waples 1991a). Factors affecting abundance can also modify selective pressures causing direction genetic changes in natural-origin populations. These can include both ecological and management-induced pressures, i.e., selective predation on specific sizes or phenotypes or flow allocations that are based on timing of hatchery juvenile releases (Waples 1991a). The recent HGMP attempts to minimize any adverse consequences to natural origin fish in streams where hatcheries are present.

Even without significant genetic effects incurred by interbreeding, other ecological effects of MRH-produced fish should be considered when assessing the impact on NC summer steelhead. As mentioned above, hatchery releases of juvenile fish can increase competition for food or space within shared habitat and when there are limited resources. Based on where smolts are released from the hatchery as compared to where natural-origin summer steelhead smolts begin their migration, it is unlikely there is competition between the two. If any competition does exist, it would occur downstream of MRH. There are conflicting studies regarding whether hatchery juvenile salmonids following release are more successful than wild fish or less, though generally wild fish have been found to have higher survival rates during outmigration (Jokikokko et al. 2006; Melnychuk et al. 2009; Johnson et al. 2010). Hatchery release strategies, such as fish size and release date, can have a significant effect on success of hatchery-origin juveniles (Irvine et al. 2013) and the degree to which they may compete with natural-origin juveniles in the system (McCubbing et al. 2008), though success of these release strategies differ between streams and are dependent on local conditions.

The new HGMP offers a number of risk aversion measures aimed at reducing the ecological and genetic impacts of hatchery fish on wild stocks throughout the propagation process. As a means of maintaining the natural run timing, a representative proportion of all returning steelhead will be selected for spawning based on arrival time and sexual maturity. The HGMP proposes using a minimum of 50% natural-origin broodstock annually to minimize genetic drift, inbreeding, and domestication. Also proposed is a 1:1 natural-origin to hatchery-origin spawning ratio to avoid hatchery stock divergence. Parentage of each egg lot will be tracked by separately incubating each family group. This also allows culling of eggs to equalize the contribution of each mated pair to the brood. The HGMP proposes reduced production numbers as well as release of yearling steelhead in turbid, high flows to promote rapid downstream migration (CDFW 2016).

5.4.2 Mad River Hatchery Summer Steelhead

It is unknown what affects, if any, linger from the past summer steelhead propagation program, which ended a quarter century ago with the release of broodyear 1995 summer steelhead. Eggs from Skamania Hatchery on the Washougal River in Washington were used to start the propagation program in 1971 (Cramer et al. 1995; CDFW 2016). In 1972 and 1973, 100,000 summer steelhead fingerlings of Eel River stock were obtained from Trinity River Hatchery due to broodstock shortages at Washington State hatcheries (CDFW 2016). The Mad River Hatchery collected and spawned its own broodstock for the years 1974 – 1977 and then in 1978 augmented with Washougal strain yearlings from Silverado Fish Station in Yountville, CA (CDFW 2016). In 1976 a total of 222 summer steelhead returned as adults to MRH, though returns ultimately decreased over time (CDFW 2016). Table 5.1 shows total summer steelhead releases from MRH from 1972 through 1996.

Hybridization of Washougal strain summer steelhead with native Mad River summer steelhead could have resulted in decreased fitness of the native stock due to the genetic and ecological effects mentioned previously in this section; however, there is little evidence to support or refute definitive interbreeding between these two stocks. Following the termination of the summer steelhead artificial propagation program, it is unknown whether the introduced Washougal summer steelhead persisted in the system.

Table 5.1*. Total summer steelhead releases from MRH during the period of 1972 through 1996 (CDFW 2016).*

Year	Number Planted	Size (fish/lb)
1972	67,030	64.8-67.7
1972	10,400	87.2
1976	17,897	11.0
1976	59,893	32.2-32.4
1978	35,034	13.4
1979	56,335	6.5
1979	96,000	10.0
1979	14,200	7.1
1980	21,000	8.0
1980	128,500	10.0
1981	52,355	10.0-11.4
1981	33,750	12.5
1982	60,000	15.0
1983	30,015	6.0
1983	28,060	4.0
1986	102,384	4.8
1987	21,655	6.1

Year	Number Planted	Size (fish/lb)
1987	20,075	5.5
1987	37,260	6.9
1987	24,790	6.7
1987	21,760	3.4
1989	79,205	7.3
1990	147,395	8.2
1990	2,205	3.0
1991	79,002	6.3
1992	74,500	5.0
1992	40,380	12.0
1993	96,000	6.0
1994	75,000	100.0
1994	96,990	5.3
1995	51,600	4.3
1996	54,900	6.1
1996	72,600	4.4

5.4.3 Eel River Steelhead Planting

There were also significant steelhead planting efforts in the Eel River system throughout the 1900s. Steelhead egg collection began on Price Creek in 1902. Another egg collecting station was constructed on Howe Creek for steelhead propagation (CFC 1902 as cited in Yoshiyama and Moyle 2010). Over nine million steelhead (run unspecified) have been planted in the Eel River since 1900 (SEC 1998 as cited in Yoshiyama and Moyle 2010). Many steelhead were planted in the South Fork Eel River between 1956 and 1965, though plantings continued through 1995. Most plantings on the mainstem Eel River were released at Van Arsdale Fish Station between 1965 and 1995 and were predominantly Eel River stock, though some Mad River stock was used for the 1974 and 1978 plantings (SEC 1998 as cited in Yoshiyama and Moyle 2010). Steelhead egg collection at Van Arsdale, which was established by Snow Mountain Light and Power Company, occurred from 1907 to 1997 (NMFS 2002). Rearing of Eel River steelhead eggs was integrated into other hatchery programs, MRH and Don Clausen Hatchery on the Russian River, by 1975 (NMFS 2002).

Steelhead egg collection has also occurred on Yager Creek, a tributary of the Van Duzen River, since 1972. The Yager Creek Hatchery facility, owned by Pacific Lumber Company (PALCO), was constructed in 1976 to collect Chinook salmon and steelhead eggs mainly for release on PALCO property. In 1993, satellite facilities on the South Fork Yager Creek and Corner Creek were constructed (CDFG 1999 as cited in NMFS 2002). The hatchery on Yager Creek was operated from 1977 through 2000 (HRC 2009).

5.5 Predation

5.5.1 Freshwater Predation

Predation is not thought to be a major threat to summer steelhead or a primary cause for their decline, with the exception of predation by Sacramento pikeminnow in the Eel River system. Steelhead have coevolved with natural, native predators and have been subject to natural rates of predation in all life stages. Predators in the freshwater environment can be both native and non-native and include aquatic, terrestrial, and avian species. Avian predators can be American Dippers (*Cinclus mexicanis*), gulls (*Larus* spp.), Belted Kingfishers (*Megaceryle alcyon*), herons (*Ardea* spp.), Common Mergansers *(Mergus merganse*r), and Osprey (*Pandion haliaetus*) (Spence et al. 1996). As the quality of river and estuary habitat decreases, avian predation likely will increase. River otters are also salmonid predators and have been documented to consume steelhead in Northern California streams (Moyle et al. 1995). Otter predation may fluctuate year to year but has been documented to be most substantial when their preferred prey sources are less available (Moyle et al. 1995). Although otter predation has not been studied, it may be intensified in areas that have experienced decreased pool depths, loss of pools due to sedimentation, lack of LWD, and increased stream temperatures (M. Sparkman, CDFW, pers. comm., July 10, 2020). Lonzarich and Quinn (1995) found that juvenile steelhead preferred habitat with more structure and greater depth, and they speculated that selection of these habitats could be to seek refuge from terrestrial predators. River otter predation was mentioned in 1980 as a potential problem on the Middle Fork Eel River that may need to be addressed (Jones and Ekman 1980).

The predator species mentioned above that has caused special concern over its effects on steelhead populations in the Northern California DPS is the Sacramento Pikeminnow, which is native to California's Central Valley, but was illegally introduced into the Eel River Basin around 1979 or 1980 (Moyle 2002; Yoshiyama and Moyle 2010). Sacramento Pikeminnow currently inhabit all sub-basins of the Eel River system (Moyle 2002). Warmer water temperatures have made the Eel River more suitable for Sacramento Pikeminnow than native salmonids, allowing the non-native species to spread throughout the system within about a decade of introduction (Higgins et al. 1992). Although salmonids are not a typical prey item under normal conditions, Sacramento Pikeminnow can consume large numbers of juveniles especially in higher temperatures when their metabolic rate increases (Moyle 2002).

Nakamoto and Harvey (2003) found that in the mainstem Eel and South Fork Eel rivers, Sacramento Pikeminnow exhibited non-selective feeding behavior, though they became increasingly piscivorous at larger sizes (Nakamoto and Harvey 2003). Salmonids only accounted for a small proportion of their diet overall, especially for small Sacramento Pikeminnow. However, steelhead made up 23% of the diet of large Sacramento Pikeminnow (>250 mm standard length) in the South Fork Eel River from late season sampling in August. One sample site on the South Fork Eel River showed particularly high rates of late season predation on steelhead and was thought to have a high concentration of steelhead in the presence of large Sacramento Pikeminnow (Nakamoto and Harvey 2003). Results from this study suggest that Sacramento Pikeminnow will consume steelhead if they are present and areas where relative abundance of steelhead is high compared to other prey items may be hotspots for predation.

Steelhead declines between Scott and Cape Horn dams have been documented and attributed to the widespread presence of Sacramento Pikeminnow (SEC 1998). The species tends to congregate in main channels as well as near tributary confluences during the spring presumably feeding on young salmonids that are emigrating to the ocean during that time of year (SEC 1998). In lower water years, Sacramento Pikeminnow likely pose a more severe threat to juvenile salmonids due to higher, more suitable temperatures for the non-native fish and lower abundances of salmonids (SEC 1998). Further study of Sacramento Pikeminnow foraging could elucidate predation threats to NC summer steelhead throughout the Eel River drainage.

5.5.2 Marine Predation

Marine predation specifically targeting steelhead is not well documented or understood. Primary predators of salmonids in the marine environment are pinnipeds such as harbor seals (*Phoca vitulina*), California sea lions (*Zalophus californianus*), and Stellar sea lions (*Eumetopia jubatus*) (Spence et al. 1996). Seals and sea lions are known to hunt mostly in the saltwater environment and estuary but can also travel into rivers to pursue migrating fish (M. Sparkman, CDFW, pers. comm. July 10, 2020; Spence et al. 1996). Though steelhead have been found to compose less than 20% of the pinniped diet (Jameson and Kenyon 1977; Roffe and Mate 1984), the recent increases in pinniped populations in Oregon and California (most notably harbor seals and California sea lions) have caused concern in regard to declining numbers of salmonids (NMFS 2011). Hanson (1993 as cited in NMFS 2011; NMFS 2016b) specifically mentioned Humboldt Bay and the Mad River as areas where sea lions may feed on salmon and steelhead. Pinniped predation has not been considered to be a factor contributing to the declines of NC steelhead populations (NMFS 2011; NMFS 2016b). However, predatory wounds from otters, seals, and sea lions have been documented on up to 50% of first time returning adult steelhead at MRH (M. Sparkman, CDFW, pers. comm., July 10, 2020).

5.6 Competition

Steelhead generally do not compete with non-salmonid species. One exception may be competition between juvenile steelhead and juvenile Sacramento pikeminnow in the Eel River system (Reese and Harvey 2002). When it comes to interaction with other salmonids, *O. mykiss* tend to be very aggressive and defend their freshwater feeding territories (Moyle 2002). Juvenile steelhead are also more equipped to occupy fast water habitats than other salmonids due to their more cylindrical body type, smaller median fin size, and larger paired fins (Bisson et al. 1988). These physical characteristics are ideal for holding in faster moving water (Bisson et al. 1988).

Segregation of steelhead from other salmonids cooccurring in streams into different microhabitats has been observed during spring and summer due to competition for space (Hartman 1965; Everest and Chapman 1972). Hartman (1965) showed habitat segregation was most pronounced between Coho Salmon and steelhead during the spring and summer with steelhead occupying and defending riffle habitat, whereas Coho Salmon occupied and defended pool habitat. Hartman (1965) observed no habitat segregation between Coho Salmon and steelhead during the winter when they were found coexisting in pools. This is likely because fish exhibited less aggressive behavior and environmental demands were different between the species (Hartman 1965). Nilsson (1956) postulated that spatial segregation occurs in sympatric populations of species that are closely related due to the tendency of a species to compete under conditions to which it is best adapted, giving them an advantage over the other species (Nilsson 1956 as cited in Hartman 1965). However, *O. mykiss* have been observed to experience competitive pressures from other salmonid species, including from non-native Brown Trout. When adult Brown Trout have been found in streams with steelhead, they tend to edge steelhead into faster moving water lacking cover, exposing steelhead to greater predation and angling threats (Moyle 2002). Juvenile Brown Trout have been found to displace juvenile Rainbow Trout into deeper, faster water than they would otherwise prefer when Brown Trout are absent (Gatz et al. 1987). A similar pattern was observed by Young (2004) who reported that juvenile Coho Salmon with a natural size advantage pushed steelhead into riffle habitat. This asymmetric competition, therefore, was found to reduce niche overlap between Coho Salmon and steelhead. Young (2004) also determined that steelhead tended to be more generalist in terms of habitat preference, using riffle habitat even in the absence of Coho Salmon.

Intraspecific competition among steelhead can also occur, especially in shallow, degraded waterways. Abundant smaller steelhead in these environments may outcompete and stunt the growth of larger steelhead; however, Everest and Chapman (1972) observed inter- and intraspecific size groupings that minimized interaction of different sized pre-smolts of steelhead and Chinook Salmon (Everest and Chapman 1972). As they increased in size, juvenile fish were shown to move into microhabitats with deeper, faster-moving water (Everest and Chapman 1972).

5.7 Genetic Diversity

Genetic diversity provides a species with the raw materials that allow flexibility in responding to and surviving long-term environmental changes and short-term random events (Waples et al. 1990; Allendorf et al. 1997; McElhany et al. 2000). Genetic variation of a species is characterized at two levels: 1) the quantity and type of variability within populations, and 2) the degree of differences among populations (Waples et al. 1990). Within population diversity includes measures such as heterozygosity and number of alleles per locus. Diversity among different wild populations is mainly a result of the effects of reproductive isolation, genetic drift, gene flow, and local adaptation through selection.

Loss of genetic variation can translate to loss of alleles, loss of heterozygosity, or changes in allele frequencies. All potential genetic variation has a chance of being adaptive, non-adaptive, or maladaptive, and can positively or negatively impact the nature and persistence of breeding populations. Risks correlated with loss of genetic diversity have been investigated in several published papers including Waples (1991a), Currens and Busack (1995), Utter (1998), and McElhany et al. (2000).

Biodiversity and its genetic foundations should be preserved for three main reasons (McElhany et al. 2000):

1. It provides the flexibility to make use of a wider variety of environments, i.e., differential run and spawn timing allowing for the use of a greater range of spawning habitat than would otherwise be possible without this variation.
2. It shields a species from short-term environmental changes that would otherwise wipe out or severely diminish abundance; the more diverse a population is, the more likely that some individuals would survive and reproduce even if others perished.
3. Genetic diversity provides species with the raw material for persisting through long-term ecological variations allowing them to adapt to changes in the freshwater, estuarine, and ocean environments caused by natural processes or human-induced effects.

Loss of genetic diversity can be a consequence of inbreeding depression, which is the same as outbreeding depression, except that it involves individuals of the same population breeding with each other rather than hybridization of a wild stock with an outplanted hatchery stock. Inbreeding depression can result in a greater proportion of homozygotes with deleterious, recessive alleles, which reduces the fitness of a population and increases local extinction risk (Waples 1991a).

Decrease in population size, and associated reductions in effective population size, is one of the main causes of genetic diversity loss. Loss of genetic diversity per generation is related to the effective population size (N_e), which is the number of effective breeders (Allendorf et al. 1997) and is generally much smaller than the total population size (N_t). The ratio of N_e/ N_t was estimated by Allendorf et al. (1997) to be around 0.2 for natural populations of salmonids. Effective population size is often used to estimate population size targets for conservation (McElhany et al. 2000), but the minimum effective population size necessary to maintain enough genetic variation and avoid inbreeding depression has been debated among scientists. Franklin (1980) and Soulé (1980) proposed that at least 500 effective breeders is necessary to avoid long-term loss of genetic variation (Franklin 1980, Soulé 1980 as cited in Lande 1995).

Lande (1995) suggested that an effective population size of 5,000 may be necessary to maintain adaptive genetic variation in a population.

Applying the N_e/N_t estimate from Allendorf et al. (1997) to the minimum effective population size from the literature cited above (500), the target minimum population size per generation sufficient to maintain long-term genetic variation is 2,500. Assuming an average generation time of four years for NC summer steelhead, the rough number of NC summer steelhead breeders per year needed to maintain acceptable amounts of genetic diversity is 625. Given that most NC summer steelhead total population sizes, except that of the Middle Fork Eel River, fall below this number (Table 3.2), none of their effective population sizes would be close to 625 breeders using the N_e/N_t ratio from Allendorf et al. (1997). Thus, all NC summer steelhead populations likely have a high potential for the loss of genetic variation. Summer steelhead possess important genetic variation that contributes to overall diversity of steelhead in the NC DPS and should be conserved.

5.8 Habitat Condition

Habitat condition is a major factor in the persistence of NC summer steelhead and has been affected by many anthropogenic factors. These activities in conjunction with natural processes and catastrophic events have impaired habitat quality and availability in all NC summer steelhead streams. One of the biggest issues that is common among all watersheds that support summer steelhead in the NC steelhead DPS is sedimentation. Timber harvest activities and road building, in particular, have contributed greatly to this problem. The combination of inherently unstable terrain with road construction and deforestation has caused severe erosion and sediment loading. Turbidity levels are high in these systems and much of the pool habitat has been eliminated (NMFS 2016b). Loss of habitat complexity from removal of riparian flora, lack of instream wood, filling of pools, and channel aggradation and widening impairs these watersheds as well. Low flow conditions are common to all NC summer steelhead streams and seasonal low flows are further reduced by water diversions including major dams on the Eel Rivers (NMFS 2016b). In contrast, Matthews Dam on the Mad River supplies water during summer months, which would not otherwise occur (M. Sparkman, CDFW, pers. comm., July 10, 2020). Though temperature can be an issue in areas of all NC summer steelhead watersheds, it affects some streams more than others (NMFS 2016b). Redwood Creek specifically is highly temperature-impaired especially in the middle and upper mainstem (Sparkman 2017) while in the Mad River, water temperature is only problematic in some areas of the watershed with some tributaries and mainstem areas having temperatures that likely sustain steelhead rearing all year (NMFS 2016b).

General impacts on NC summer steelhead streams are listed below with primary causal factors in parentheses:

- Obstruction and sedimentation of spawning and rearing habitat (dams, artificial barriers, road construction, and landslides)
- Alterations to natural flow regimes (dams, water diversions, artificial barriers, and road construction)
- Altered sediment transport and increased gravel embeddedness, and reduction of gravel particle sizes suitable for spawning (dams, artificial barriers, and road construction)
- Destabilization of inherently unstable topography (timber harvest and road construction)
- Reductions of riparian cover habitat (timber harvest, road construction, and livestock grazing)
- Increased erosion and mass wasting resulting in sediment loading (timber harvest, road construction, gravel mining, and livestock grazing)
- Reduced water quality from nutrient loading and suspended fines (livestock grazing and marijuana grows)
- Higher stream temperatures (timber harvest and water diversions)
- Reduced habitat complexity (timber harvest and road construction)

5.8.1 Timber Harvest

Timber harvest began in the northwestern region of California in the mid-nineteenth century. Large-scale harvest ramped up in the mid-twentieth century and continues today in watersheds both historically and currently inhabited by NC summer steelhead. Lands used for timber harvest, both public and private, make up a large proportion of the NC summer steelhead range. In Humboldt County, Timber Production Zones, which are areas devoted solely to growing and harvesting timber, make up almost half of the total land area, or about 1,009,000 acres (Humboldt County 2012). In Mendocino County, there are over 850,000 acres of Timber Production Zone lands, making up 38% of the total land area (Mendocino County 2009). Consequences of logging that directly affect steelhead are sedimentation, increased water temperatures, removal of canopy cover, instream habitat destruction, reduced woody debris instream, and changes in flow regimes (Moyle et al. 2017). For example, the Eel River and its tributaries experienced widespread channel aggradation and braiding, which degraded steelhead spawning and rearing habitat, created more obstructions to juveniles migrating downstream, and reduced productivity of aquatic invertebrates, which are a primary prey source for juvenile steelhead (Moyle et al. 2017).

Much of the summer steelhead range in California is characterized by severely unstable geology, which, in conjunction with significant seasonal precipitation in these areas, has caused massive sediment loads to be flushed into the stream channels (Roelofs 1983; Madej and Ozaki 1996). Sedimentation typically occurs as a result of mass wasting processes (Swantson and Swanson 1976). Although these mass wasting events sometimes occur naturally, often they are triggered or worsened by timber harvest activity. Sediment loading can fill pool habitat used by summer steelhead for holding and rearing, embed spawning habitat with fines, and raise the streambed creating shallower runs and riffles (Madej and Ozaki 1996; Moyle et al. 2017).

Deforestation can also substantially reduce slope stability and soil strength, as well as alter rainfall and snowmelt hydrology and increase debris in stream channels (Swantson and Swanson 1976). Canopy cover regulates the amount of precipitation that reaches the forest floor through interception and evapotranspiration. Controlling moisture absorption helps maintain resistance of the land to mass sliding, which can occur via mobilization of clay or surface soil movement when highly saturated (Swantson and Swanson 1976). Tree canopy also controls the rate at which precipitation infiltrates the soil and mitigates severe rain-on-snow runoff (Swantson and Swanson 1976). Peak winter flows enhanced by vegetation removal have higher potential to scour redds or suffocate them in sediment (Lisle 1989 as cited in Nakamoto 1998). Root systems of trees also stabilize hillslopes through vertical anchoring and, especially in areas of highly unstable terrain, root structures can be key in sustaining slope stability (Swantson and Swanson 1976). Removal of riparian vegetation can also have a substantial impact on stream temperatures. Riparian canopy serves as shade and helps mitigate rising stream temperatures during the summer. It also provides cover habitat for fish and can act as thermal refugia. Timber harvest also results in less available LWD, a key habitat feature for salmonids.

In addition to the logging activity itself, methods used to transport harvested timber have had a significant effect on salmonid habitat. Historically, log driving was the main method for lumber transportation. Log driving uses stream flow to float logs downstream in loose aggregations (Sedell et al. 1991). As timber adjacent to streams was logged and timber harvesters had to haul wood from farther distances, instream alterations became more extensive including construction of splash dams and sluiceways as well as dredging of canals (Sedell et al. 1991). Other alterations to “improve” streams for better log transportation included blocking off of sloughs, swamps, low meadows, and wide banks to keep wood in the main channel, and blasting of instream obstacles like large boulders or accumulations of LWD (Sedell et al. 1991). Alterations of this nature disrupt the normal flow patterns of a stream and pose threats to salmonid spawning and egg incubation (Sedell et al. 1991).

Modern log transportation now relies on trucks and log driving via streams is no longer used in California; however, construction of forest roads for use in the harvest of timber can also be severely damaging most notably due to erosion or failure and subsequent sediment loading in streams (Yee and Roelofs 1980; Reid 1998). A 1973 comprehensive review of literature on the impacts of timber harvest on streams concluded that forest roads are the largest human cause of erosion (Gibbons and Salo 1973). Sediment transport from roads into streams most often occurs through mass soil movement and surface erosion (Yee and Roelofs 1980), though even small amounts of erosion can have cumulative impacts. Construction of roads over steep terrain can also cause or aggravate slope failure (Yee and Roelofs 1980). In addition to erosion, road culverts can also be barriers to anadromous migration (Yee and Roelofs 1980). There are numerous culverts within the NC summer steelhead range that do not provide suitable access for adult and juvenile salmonids (M. Sparkman, CDFW, pers. comm., July 10, 2020). Culvert failure can also add sediment to streams and rivers.

Other effects of logging can be indirect, such as reduced resistance to wildfires. In coastal areas of Northern California, forest management focused on tree production has led to more frequent fires with increased severity (Moyle et al. 2017). A practice called “salvage logging,” which is the removal of dead trees following a fire, can increase erosion and reduce the amount of LWD available for salmonid habitat (Moyle et al. 2017). Timber harvest activity can also increase access to characteristically remote summer steelhead habitat and increase the potential for poaching via the construction of logging roads (Roelofs 1983).

During the last century of timber harvest in coastal Northern California, harvest practices have evolved dramatically, primarily due to changes in technology, decreasing availability of larger or higher quality logs, and state and federal legislation. There are also many more restrictions in place to help maintain aquatic habitat and water quality. Current forest practices in California, in fact, have been shown to sometimes result in favorable habitat modification, such as increased water yield (Keppeler 1998 as cited in CDFG 2002), increased insect productivity (Hicks et al. 1991), and increased salmonid productivity (Graves and Burns 1970; Nakamato 1998). However, increased smolt yield observed by Graves and Burns (1970) may have been a result of premature emigration due to decreased favorable rearing habitat after forest road construction, and Nakamoto (1998) states that benefits of logging activity to steelhead in the short-term may not last. Burns (1972) found that if proper measures were implemented to protect stream habitat and the watershed, logging practices could enhance anadromous fish production. See Table 5.2 for an overview of potential positive and negative effects of forest practices on the stream environment and salmonid habitat and biology.

Regardless of improved techniques and regulation, legacy effects from a long history of timber harvest remain in the coastal watersheds of Northern California. Legacy effects such as sedimentation and slope instability continue to impact salmonid habitat long after logging activity has ended since recovery from these issues can take many decades if not longer (Murphy 1995).

***Table 5.2**. Forestry activities and potential changes to stream environment, salmonid habitat, and salmonid biology (CDFG 2002 adapted from Hicks et al. 1991).*

Forest Practice	Potential effects to stream environment	Potential effects to salmonid habitat	Potential effects to salmonid biology
timber harvest in the riparian zone	Increased incident solar radiation	Increased temperature, light levels, and primary production	increased susceptibility to disease; increased food productivity; changes in growth rate and age at smolting
timber harvest in the riparian zone	decreased supply of LWD	decreased cover, storage of gravel and organic debris, and protection from high flows; loss of pool habitat and hydraulic and overall habitat complexity	decreased carrying capacity, spawning gravel, food production, and winter survival; increased susceptibility to predation; loss of species diversity
timber harvest in the riparian zone	increased, short-term input of LWD	increase in number of pools and habitat complexity; creation of debris jams	increased carrying capacity for juveniles and winter survival; barrier to migration and spawning and rearing habitat
timber harvest in the riparian zone	increased influx of slash	increased oxygen demand, organic matter, food, and cover	decreased spawning success; short-term increase in growth
timber harvest in the riparian zone	stream bank erosion	reduced cover and stream depth	increased carrying capacity for fry; decreased carrying capacity for older juveniles; increased predation
timber harvest in the riparian zone	stream bank erosion	increased in-stream fine sediment; reduced food supply	reduced spawning success; slower growth rates for juveniles

Forest Practice	Potential effects to stream environment	Potential effects to salmonid habitat	Potential effects to salmonid biology
timber harvest on upslope areas	altered stream flow	temporary increase in summer stream flow	temporary increase in survival of juveniles
timber harvest on upslope areas	altered stream flow	increased severity of peak flows during storm season; bedload shifting	increased egg mortality
timber harvest on upslope areas and road construction and use	increased erosion and mass wasting	increased in-stream fine sediment; reduced food supply	reduced spawning success, growth and carrying capacity; increased mortality of eggs and alevins; decreased winter hiding space and side-stream habitat
timber harvest on upslope areas and road construction and use	increased erosion and mass wasting	increased in-stream coarse sediment	increased *or* decreased carrying capacity
timber harvest on upslope areas and road construction and use	increased erosion and mass wasting	increased debris torrents; decreased cover in torrent tracks; increased debris jams	blockage to migration of juveniles and spawning adults; decreased survival in torrent tracks
timber harvest on upslope areas and road construction and use	increased nutrient runoff	increased primary and secondary production	increased growth rate and summer carrying capacity
timber harvest on upslope areas and road construction and use	stream crossings	barrier in stream channel; increased sediment input	blockage or restriction to migration; reduced spawning success, carrying capacity and growth; increased winter mortality

Forest Practice	Potential effects to stream environment	Potential effects to salmonid habitat	Potential effects to salmonid biology
Scarification and slash burning	increased nutrient runoff	increased primary and secondary production	temporary increased growth rate and summer carrying capacity
Scarification and slash burning	increased input of fine organic and inorganic sediment	increased sedimentation in spawning gravels and production areas; temporary increase in oxygen demand	decreased spawning success; increased mortality of eggs and alevins

5.8.2 Roads

Road networks are components of various land-use activities and are widespread throughout California's watersheds. The process of road building can damage stream habitat and riparian corridors as a result of channelization from roads built on top of stream banks. Continued use of roads that meander through riparian habitats increase runoff and deposit contaminants (e.g., engine oil, gasoline) into adjacent streams. Increased watershed imperviousness as a result of roads with impermeable surfaces such as compacted dirt, gravel, or pavement, has been found to alter natural flow regimes, stream morphology, natural erosion processes, and water chemistry of lowland streams used for salmonid spawning and rearing in the Puget Sound area of Washington (May et al. 1997). This assuredly occurs elsewhere where logging roads parallel the stream channel.

Roads, especially poorly maintained rural or forest roads, can contribute large volumes of inorganic sediment into streams and rivers from mass wasting or landslides. Culverts can change channel morphology and alter natural sediment transport. Culverts and road crossings can cause habitat fragmentation, acting as barriers to the movement of aquatic organisms including fish (Warren and Pardew 1998; O'Hanley and Tomberlin 2005; Bouska and Paukert 2009). Negative effects from road construction and use are exacerbated by the unstable geology of many NC summer steelhead watersheds. In the Van Duzen River, Shaun Thompson (CDFW) has observed pools up to 14 feet completely filling with sediment over the course of one winter season. Given there are only about six pools used by summer steelhead for holding in any given year, poor land use could quickly eliminate this adult holding habitat (S. Thompson, CDFW, pers. comm., July 14, 2020).

5.8.3 Dams

Large dams were primarily built for flood protection and sources of water supply, and power generation in California. Dams have the potential for major ecological and hydrographic impacts on streams and interrupt the natural cycle of a river over time. Dams impede the transport of gravels from upstream as well as alter the natural hydrograph, often reducing the volume of flow necessary for sediment transport and scour. By disrupting natural sediment transport, water downstream of a dam can possess higher kinetic energy due to lack of sediment load. This clear, sediment-deprived water may erode and incise the channel as well as coarsen streambed material (Kondolf 1997). Dams can also reduce natural peak flows downstream. Without adequate flows to dislodge sediment, fines may accumulate and have the potential to embed spawning gravels downstream of the dam (Kondolf 1997). Reduced peak flows may have negative effects on juvenile steelhead, which rely on these flows to emigrate to the ocean. Adult summer steelhead migrating to their summer holding grounds also depend on these peak flows, especially to reach areas above flow-dependent barriers. Base flows can be altered, as well, with consequences stemming from both increases or decreases such as redd scouring or dewatering and juvenile stranding. However, in some cases like

Matthews Dam on the Mad River, dam releases of water during summer months ensure water is present and can be considered beneficial to salmonids, especially summer steelhead (HBMWD 2004). Historically, summer flows became subsurface and the river channel would go dry during the late summer and fall, but releases from Ruth Lake now augment downstream flows year-round even in drought conditions (HBMWD 2004). Additionally, Matthews Dam has relatively little impact once the reservoir is full and spilling during the winter months (M. Sparkman, CDFW, pers. comm., July 10, 2020). Changes to normal flow regimes can also increase or decrease water temperatures downstream by releasing reservoir water that is epilimnetic or hypolimnetic, respectively (Sylvester 1963 as cited in Kubicek 1977; Moore 1967). Reductions of normal streamflow below an impoundment can cause increased water temperatures in downstream areas (Sylvester 1963 as cited in Kubicek 1977). There are three major dams within the NC summer steelhead range, two of which are complete barriers to anadromy.

Mad River: Robert W. Matthews Dam (previously known as Ruth Dam) was constructed about 80 miles upstream from the mouth of the Mad River in southern Trinity County in 1961 (CDFG 2002). The intended purpose of the dam was to supply water for industrial and domestic use as well as provide hydroelectricity. R.W. Matthews Dam is a complete barrier to upstream migration and influences streamflow downstream of the dam. However, as mentioned above, Matthews Dam maintains summer flows at 40-60 cfs as measured by the Highway 299 USGS flow gage (#11481000) on the Mad River. According to the NMFS Multispecies Recovery Plan (2016b), Matthews Dam is only considered to block about two miles of historical spawning and rearing habitat, and these two miles likely are made up of low-quality habitat, a portion of which was intermittently dry during the summer (NMFS 2016b).

Eel River: Scott Dam was built in 1922 and is one of two dams that make up the PVP, along with Cape Horn Dam. The PVP is owned by PG&E and operates to store seasonal runoff from the upper Eel River drainage and divert about 90,000 acre-feet of water to the Russian River on an annual basis (PVP 2019). Scott Dam forms Lake Pillsbury and Cape Horn Dam forms Van Arsdale Reservoir. A 9,258-foot diversion tunnel moves water from the Eel River to the East Fork Russian River for municipal and agricultural uses (CDFG 1997a).

The 12-mile stretch between Scott and Cape Horn dams has high quality spawning and rearing habitat for salmonids with cool water temperatures, ideal spawning substrate, and decent riparian cover (CDFG 1997a). Water temperatures are optimal due to releases from Lake Pillsbury. However, high temperatures downstream of the dam are often lethal to salmonids and low flows as well as sparsity of riparian cover contribute to the low quality of habitat (CDFG 1997a).

Construction of Scott Dam and formation of Lake Pillsbury was likely the main cause for the loss of summer steelhead in the upper Eel River (NMFS 2016b). According to a 1982 fisheries study on the PVP by VTN Oregon, Inc., Scott Dam blocks over 35 miles of major channels, including 8 miles of the mainstem Eel River, and 25 miles of minor channels that contained habitat accessible to steelhead (VTN Oregon, Inc. 1982). More recent estimates are much higher with a range of 198 – 288 miles of potential steelhead habitat existing above Scott Dam (Cooper 2017). Lake Pillsbury, formed by Scott Dam, and Van Arsdale Reservoir, formed by Cape Horn Dam, were both found to have significant effects on downstream water temperature (Kubicek 1977). Lake Pillsbury has substantial thermal stratification from late spring through early fall (PG&E 2017). With reservoir water being drawn down throughout the summer, hypolimnetic water can be depleted, leaving only warm water (24°C or higher) from the epilimnion by the late summer or early fall. Temperature of water releases, thus, is dependent on how much water remains stored in the lake. In years when the cool hypolimnion has been depleted, water release temperatures during the late summer and early fall have exceeded 22°C and in some years water temperatures have exceeded 20°C for over a month. DO in the hypolimnion decreases through the early summer and is ultimately depleted by late July. However, as water is released through the discharge structure of the dam it is aerated and DO concentration increases (PG&E 2017).

Minimum flow requirements are in place year-round for the Eel River below Scott Dam and below Cape Horn Dam in attempt to mimic the natural hydrograph of the system (PG&E 2017). As a result of PVP operations, flows have regularly been lower and less annually variable than unimpaired flow conditions (NMFS 2002). Middle and upper reaches of the Eel River have experienced accelerated flow attenuation and increased temperatures, which have occurred during the spring and early summer, and may decrease the window when conditions are suitable for juvenile steelhead emigration (VTN 1982; NMFS 2002). Low summer flows also have the potential to inhibit upstream migration of summer steelhead through the mainstem Eel River (NMFS 2002).

New evidence shows that the GREB1L allele found to be associated with the premature migration phenotype is still present in resident Rainbow Trout residing above the dam (Kannry et al. 2020). This discovery is evidence of the potential for a summer steelhead population to re-establish if Scott Dam were to be removed. The Federal Energy Regulatory Commission (FERC) regulates the PVP and the 50-year license expires in 2022 (PVP 2019). After providing FERC with a Notice of Intent (NOI) to renew their license, PG&E later withdrew their NOI in 2019 along with their pre-application document (PVP 2019). A Planning Agreement was initiated by the Mendocino County Inland Water and Power Commission, Sonoma County Water Agency, California Trout, Inc., and Humboldt County in May 2019 to look into obtaining a new license for the PVP (Feasibility Study Report 2020). These parties filed a Notice of Intent (NOI) with FERC to obtain a new license for the PVP, and the Round Valley Indian Tribes were included in an amended version of the Planning Agreement shortly thereafter. The NOI parties submitted a Feasibility Study Report of their potential licensing proposal in May 2020 to FERC. Included in their potential licensing proposal is a plan for the complete removal of Scott Dam with the intent to restore access of anadromous fish to the watershed areas upstream of Scott Dam and Lake Pillsbury (Feasibility Study Report 2020).

5.8.4 Water Diversions

Diversion of water for domestic, industrial, and municipal uses is typically continuous by nature while agricultural diversions tend to have seasonal fluctuations, drawing more water during the summer and fall. More recently, illegal marijuana cultivation has been shown to have significant effects on streamflow, as well. Diversion structures can have similar, albeit smaller scale, impacts to salmonids as those of large dams. Water temperatures can increase in impoundments or slow-moving backwaters that result from diversion structures (Axness and Clarkin 2013), which may pose a risk to young salmonids. Decreases in flows from water diversions can increase downstream temperatures, reduce the amount and quality of available habitat for salmonids (Axness and Clarkin 2013), and potentially increase exposure to predation. These conditions may be exacerbated in low water conditions during dry seasons. Reductions of flow from water diversions may dewater the channel downstream (Axness and Clarkin 2013), which could reduce or eliminate critical habitat for rearing and outmigration, and in some cases stream drying may result in mortality of juvenile salmonids. Aquatic organisms, such as juvenile steelhead, may also become trapped in diversion ditches and be unable to escape due to lack of habitat connectivity (Axness and Clarkin 2013). Unscreened diversions may transport juvenile steelhead into unsuitable areas (Moyle et al. 1995), increasing mortality and opportunity for predation. Diversions containing fish screens present alternative problems with fish becoming impinged on some types of screens when water velocity is too high. Another potential impact of these diversions is impeded upstream passage of adults (Axness and Clarkin 2013). See *Sections 5.8.9* and 5*.8.10* for more on water diversions specific to agriculture and cannabis cultivation.

According to the California Fish Passage Assessment Database (PAD), there are upwards of thirty unscreened water diversions in the upper extent of the Mad River. There are even more unscreened diversions on the Mattole River, most of which are around or upstream of Whitehorn, CA, indicating that many could be municipal or domestic. There are likely diversions in other NC summer steelhead watersheds that may not be reported and thus are currently not included in the PAD. The number of diversions for illegal marijuana grows is unknown.

5.8.5 Artificial Barriers

Artificial barriers or structures that have the potential to prevent or reduce fish passage include dams, road crossings, concrete channels for flood control, energy dissipators for erosion control, gravel mining pits, pipeline crossings, and potentially others. These structures can have surface or subsurface effects and eliminate or fragment aquatic ecosystems by blocking or impeding migration and altering nutrient cycling patterns, flow regimes, sediment movement, channel morphology, and species composition (Poff et al. 1997; Ward 1998; Bunn and

Arthington 2002). Alterations of this nature reduce habitat availability and quality as well as native biodiversity.

There were two known artificial gravel dams constructed annually on Redwood Creek to create swimming holes during the summer. These structures likely impeded summer steelhead access upstream during the latter part of their migration, but neither have been built for many years.

5.8.6 Gravel Mining and Extraction

Gravel mining has been present in Northern California's coastal watersheds for close to a century and is particularly problematic in streams south of Humboldt Bay, which include sectors of the Eel River system (McEwan and Jackson 1996). Other NC summer steelhead streams affected by gravel extraction include the Mad River and, to a minor degree, Redwood Creek and the Mattole River. There are three methods used to mine gravel: dry-pit mining, wet-pit mining, and bar skimming (Kondolf 1993). Wet-pit mining, or in-channel extraction of gravel within a perennial stream (Kondolf 1993), was the most common method used in the Mad River (Stillwater Sciences 2010), though currently bar skimming is used most frequently (M. Sparkman, CDFW, pers. comm., July 13, 2020). Dry-pit mining is the excavation of gravel from dry streambeds and bar skimming is removing the top layer of a gravel bar above water level at low flows (Kondolf 1993).

While all forms of mining can greatly impact the aquatic ecosystem, geomorphology, and flow hydraulics of the stream, instream wet-pit mining is likely the most detrimental to salmonid productivity and survival. Removing coarse sediment from an active channel results in channel incision by trapping bedload sediment (coarse sediment, sand, and gravel) in the excavated pit and starving the water moving downstream of sediment (Kondolf 1993). The sediment-deprived water has higher kinetic energy and erodes the stream bed to replenish its bedload. The resulting stream bed is dominated by larger substrate as smaller particles get washed downstream, leaving poor spawning habitat for salmonids (Kondolf 1993, 1997). Highly charged flows also have the potential for downstream redd scouring (Kondolf 1997).

Bar skimming has been the main method used throughout the Eel River and Mad River drainages, though in recent years improved methods, such as trenching, have begun to be used (NMFS 2015). Bar skimming often greatly reduces habitat complexity by scraping off the multifaceted topography, leaving the channel flat and lacking structure (Kondolf 1993). This method of gravel extraction also reduces stability of the stream in areas surrounding the removal site (Kondolf 1993). Wet trenching removes sediment from dry areas of a channel adjacent to the wetted perimeter and is typically used only when there is a need to protect salmonid habitat or mitigate impacts on the depth and width of the channel (NMFS 2015). Dry trenching takes a similar approach removing material from dry gravel bars. Berms that are created during the gravel extraction process are ultimately breached or constructed to allow breaching to prevent stranding of fish once the main channel connects with the trench (NMFS 2015).

Impacts of gravel mining can also include altered geomorphology, changes in turbidity, and decreased biomass and biodiversity of aquatic invertebrates and fish (Brown et al. 1998). Impacts on salmonid habitat, specifically, have also been studied and documented. The major effects of gravel mining on salmonid habitat include reduced channel complexity and habitat diversity, floodplain disconnection, removal of spawning gravel, damaging of existing spawning habitat, disturbances to redds that lead to egg mortality, disturbances to fish in the stream, increased turbidity, changes in composition of aquatic organisms that can accumulate through multiple trophic levels (Oregon Water Resources Research Institute 1995).

5.8.7 Estuarine Habitat

Estuaries of Redwood Creek and the Mad and Eel rivers have been substantially altered by levees, hard structure armoring, tide gates, and modifications for agricultural and rural development (Moyle et al. 2017). Habitat that does remain for juvenile salmonid rearing is affected by turbidity, and water quality issues from runoff and sediment (Moyle et al. 2017).

The Eel River estuary is the largest estuary within the NC summer steelhead range. Use of the Eel River estuary by multiple steelhead life stages is extensive year-round, but quality of the estuary habitat is very poor (NMFS 2016b). In the past, the estuary and tidal wetlands of the Eel River were diked and filled for agricultural uses and flood protection (NMFS 2016b). Only about 40% of the estuarine habitat remains due to diking and construction of levees. Use of the Eel River estuary by juvenile salmonids is particularly critical due to predation and competition pressures from the Sacramento Pikeminnow in the mainstem river (NMFS 2016b). Restoration efforts through the Salt River Ecosystem Restoration Project, The Wildland's Conservancy Eel River Estuary Preserve, and the Department's Ocean Ranch Unit of the Eel River Wildlife Area are expected to increase habitat quality and availability in the Eel River estuary (NMFS 2016b).

The Redwood Creek estuary has also been rated as poor quality for juvenile steelhead (NMFS 2016b). Flood control levees have blocked off sloughs, wetlands, tributaries, and have eliminated about 50% of the available space. The estuary has been subject to simplification including removal of cover and velocity refugia of off-channel habitat and loss of LWD. High water temperatures and low DO contribute to poor water quality that is further exacerbated by diversion culvert operations (NMFS 2016b).

The Mad River estuary alterations have led to reduced habitat complexity and loss of off-channel rearing habitat (NMFS 2016b). Natural sloughs were blocked off and the mainstem channel was straightened and channelized to reduce flooding. Construction of gravel berms, riprap, and planting of riparian foliage caused a 32% reduction of active channel area since the mid-1900s (NMFS 2016b). Unfortunately, estuaries of the Mad River and Redwood Creek are not currently being restored.

5.8.8 Livestock Grazing

Grazing of livestock on riparian land has occurred for hundreds of years in California, damaging thousands of miles of stream habitat. Livestock in the riparian zone have many direct effects on fish, including salmonids, and on the aquatic ecosystem as a whole. Ungulates remove foliage that provides cover for fish, acts as a stabilizing mechanism for stream banks, and mitigates high water temperatures (Moyle 2002). Streamside plants also serve as food and habitat for insects, which are a large component of fish diet (Platts 1981). Livestock also consume aquatic plants that provide cover for fish and aquatic insects while simultaneously stirring up sediment, which increases turbidity. Trampling of stream banks by livestock can cause bank collapse, removing undercuts that act as cover for fish (Moyle 2002). Bank collapse is thought to be the most widespread effect of livestock grazing on fish abundance since livestock typically congregate next to streams for shade, forage, and drinking water (Platts 1981). Bank collapse also contributes to erosion and filling of essential pool habitat, which can ultimately lead to more simplified channels with less usable habitat (Moyle 2002). Livestock compress soil of streamside meadows, which reduces its capacity to hold water and exacerbates runoff. This excess volume of water causes downcutting of the channel and formation of a gully where a meandering stream used to exist (Moyle 2002). Lastly, livestock pollute the aquatic environment with excrement, which primarily contributes to bacterial contamination of the water (Platts 1981) and may cause nutrient loading. Bacteria in streams does not directly affect fish or suitability of their habitat but is an indicator of water quality (Platts 1981).

Cattle grazing has been noted as a problem in select areas of the Middle Fork Eel River drainage, impacting water quality through removal of riparian vegetation, siltation, and deposition of waste (Jones and Ekman 1980). Cattle have been observed grazing streamside in the North Fork of the Middle Fork Eel River and other tributaries more so than in the mainstem Middle Fork Eel due to less availability of forage (Roelofs 1983). In the past five years cattle grazing has been observed streamside in the upper Middle Fork Eel River and in the North Fork Eel River (S. Thompson, CDFW, pers. comm., July 14, 2020). Private lands in the Van Duzen and Mad river basins have also been noted to support livestock grazing (Roelofs 1983). The Mattole River basin has experienced low levels of livestock grazing on forestland areas. Ranching in this basin, like in the Middle Fork Eel River, was mostly focused on cattle (Downie et al. 2003). Redwood Creek has also sustained heavy grazing in the lower basin, though sedimentation from livestock activity is mild compared to the effects of road building and use (Redwood Creek Watershed Analysis 1997). Throughout Humboldt County, cattle ranching uses about 470,000 acres of land, accounting for just over 20% of the total land area (Humboldt County 2012).

5.8.9 Agricultural Impacts

Some areas within the NC summer steelhead range are subject to agriculture and vineyards (Moyle et al. 2017). Removal of natural landscape features like wetlands, forests, and grasslands, to plant annual crops requiring repetitious tilling, fertilization, and harvest can permanently decrease soil infiltration capacity and result in increased agricultural runoff (Spence et al. 1996). These seasonal patterns can alter the natural hydrograph of a stream, increasing peak flows, decreasing summer baseflows, and lowering the level of the water table. Channelization of the stream and removal of riparian vegetation and LWD decreases habitat complexity, increases water temperatures, and decreases bank stability. Agricultural runoff, in addition to being warm in temperature, has high nutrient levels and contains chemical insecticides and/or herbicides, which contribute to poorer water quality (Spence et al. 1996). Agriculture accounts for a small proportion of land use in NC summer steelhead watersheds (Humboldt County 2012).

5.8.10 Cannabis Cultivation

One of the more recent threats to salmonids in coastal watersheds of Northern California in the past few decades is the cultivation of marijuana. Coastal Northern California is a prime location for this activity due to its remote, forested landscape and lack of human development. Although cannabis cultivation falls within the parameters of agriculture, much of the industry is unregulated due to illegal operations. Illicit use of pesticides, barrier construction and diversion of water for irrigation, and destruction of habitat to create space for the marijuana grows have become grave threats to salmonids, particularly in remotely located first and second order streams (NMFS 2016b).

The largest impact of marijuana cultivation on wildlife comes from water use for irrigation. The primary growing season for cannabis is during the summer, at which time there is very little rainfall because of California's Mediterranean climate (Bauer et al. 2015). Alternative methods for obtaining water are required to sustain growing operations and growers turn to streams for their irrigation needs. Bauer et al. (2015) studied impacts of cannabis water diversions in four Northern California watersheds and showed that cannabis cultivation may remove up to 23% of the annual seven-day low flow in upper Redwood Creek, which was the least affected watershed in the study.

Dillis et al. (2019) looked at irrigation sources of 901 cannabis grow sites in Humboldt, Trinity, Mendocino, and Sonoma counties in 2017. They found that wells were the primary source of water extraction, especially during the summer months, but surface and spring water diversions were also heavily relied on year-round (Dillis et al. 2019). Humboldt County sites reported less well use than other counties, relying more on surface and spring water diversions, likely due to the availability of these water sources throughout the growing season as a result of greater amounts of annual precipitation. Dillis et al. (2019) also determined that there were differences in water use between compliant and noncompliant sites across the four counties studied. Noncompliant sites relied more heavily on surface and spring water diversions than compliant sites, for which well extraction was a primary water source (Dillis et al. 2019). Subsurface pumping has the potential to decrease instream flows, especially when wells are shallow and hydrologically connected to the stream. These effects are lessened with greater distance of wells from the stream.

Severely decreased flows, as a result of direct surface or spring water diversion, or from well extraction, can have detrimental consequences especially for streams that already suffer from low flow conditions. Without sufficient flow to transport and remove sediment, streams can become embedded, decreasing salmonid spawning and rearing habitat suitability. Floodplains can also become disconnected, cutting off essential fish nursery habitat, which decreases productivity (Poff et al. 1997). Loss of this essential habitat has the potential to increase predation and competition within and among species (CDFG 2004). Reducing instream flows can increase water temperatures to harmful or lethal levels for over-summering fish and result in stagnant water conditions that promote the growth of cyanobacteria, which can be lethal to salmonids and other aquatic species (Power et al. 2015). Maintaining enough flow to sustain cool water temperatures is crucial to the survival of over-summering adult summer steelhead (Moyle et al. 2017). Grantham et al. (2012) showed a positive correlation between juvenile steelhead survival and summer streamflow. Those authors speculate that this is likely due to the increase in area and suitability of habitat, water quality, and the availability of food resources (Grantham et al. 2012).

Most, if not all, streams in the NC summer steelhead range are likely affected by legal or illegal cannabis cultivation. The lower Van Duzen River, lower South Fork Eel River, middle South Fork Eel River, and the Mattole River have been identified as cannabis priority watersheds (CASWRCB 2018). This designation is for areas with a high concentration of cannabis cultivation and where non-compliant cannabis activities could be detrimental to the local environment. These watersheds also may contain or support essential habitat for aquatic or terrestrial species, have flow conditions that are lethal or close to lethal for aquatic organisms, suffer from high water usage and diversions, have restoration projects underway, are listed as an impaired body of water under the Clean Water Act Section 303(d), are designated as fully appropriated streams, or are a “Wild and Scenic River” as determined by the California Public Resources Code section 5093 (CASWRCB 2018). A significant portion of the cold water refugia in the Van Duzen River used by adult summer steelhead for over-summer holding is maintained by seeps, springs, and tributaries that could be altered by upslope land use (S. Thompson, CDFW, pers. comm., July 14, 2020). Extensive cannabis cultivation is occurring in an area called McClellan Mountain, which will probably have an increasing impact on cold water inputs that are critical to maintaining suitable instream temperatures. Rare peat fens exist on McClellan Mountain that are likely responsible for water temperatures staying cool enough for steelhead holding through the summer. Some of these fens have been previously damaged and dewatered at least in part by cannabis development (S. Thompson, CDFW, pers. comm., July 14, 2020). A 2016 study on cannabis agriculture within Humboldt County watersheds identified 4,428 grow sites within the study area (60 watersheds out of 112 total), of which 19% and 4% were located within 500 m and 100 m, respectively, of steelhead habitat (Butsic and Brenner 2016).

5.8.11 Water Quality and Temperature

Due to the impacts of watershed uses described previously, most NC summer steelhead streams are listed under Section 303(d) of the 1972 Clean Water Act (CWA). The CWA requires states, territories, and federally recognized tribes to keep a list of water bodies that do not meet prescribed water quality standards. Governing bodies must also establish total maximum daily load (TMDL) parameters for each affecting pollutant (CalEPA 2018). CWA Section 303(d) impaired NC summer steelhead watersheds include multiple hydrologic sub-areas of the Middle Fork Eel and North Fork Eel rivers, the South Fork Eel River, Upper Mainstem Eel River, Van Duzen River, Mad River, Mattole River, and Redwood Creek (CASWRCB 2020a). North Coast streams supporting NC summer steelhead that are listed under the CWA (listed above) are polluted by a variety of dissolved metals, including aluminum, among other substances, and they are all listed as temperature-impaired systems. Temperature impairment is primarily a result of industrialized logging on large scales, flooding in 1955 and 1964, and cumulative impacts. The Redwood Creek Hydrologic Unit listing is being updated to reflect that four of its tributaries are no longer temperature impaired, although the rest of the hydrologic area continues to fall short of temperature standards (CASWRCB 2020a). Many are also impacted by pesticide runoff (CASWRCB 2020b). Sedimentation and siltation have been determined to be issues in the Mad River, Mattole River, Redwood Creek, South Fork Eel River, Van Duzen River, and areas of the Middle Fork, North Fork, and Upper Mainstem Eel rivers (CASWRCB 2020b). Likely the entire mainstem Eel River is affected by sediment.

5.8.12 Conclusions on Habitat Condition

An important consideration in the case of NC summer steelhead is how future habitat and climate conditions will affect population trajectory (Spence et al. 2008). Although NC summer steelhead have access to much of their historical habitat in most streams they still occupy, these streams have been severely affected by a combination of natural conditions and anthropogenic activities in the region. Methods used in timber harvest and gravel mining have improved in recent decades; however, legacy effects of these activities remain. Livestock grazing and agriculture, to a smaller extent, persist within the lower areas of Northern California watersheds, as well. Cannabis cultivation is a relatively new, although potentially serious, threat to the region. Major impacts from this industry come from unregulated, illicit grow operations that illegally remove and sequester water from streams. Additionally, with R.W. Matthews and Scott dams still in place, historical steelhead habitat in the Mad and Eel rivers remains inaccessible to steelhead, though with respect to the Mad River, the percentage is small compared to the total habitat in the river. Given the factors contributing to habitat loss and degradation within NC summer steelhead watersheds, it is likely that survival and productivity will continue to be significantly impacted as these threats persist into the future, particularly in light of climate change impacts.

5.9 Fishing and Illegal Harvest

There are no freshwater fisheries that allow harvest of natural-origin NC summer or winter steelhead. The only harvestable steelhead in Northern California streams are those of hatchery-origin. Even though harvest of wild steelhead is not legal, catch and release impacts are still pertinent. There are restrictions on types of gear anglers can use during certain times of year. Barbless hooks are required and only artificial lures may be used during certain times (California Freshwater Sport Fishing Regulations 2020 – 2021). As of 2015, California Code of Regulations, Title 14, Section 8.00 enforces seasonal closures on the Eel River, Mad River, Mattole River, Redwood Creek, Smith River, and Van Duzen River to protect wild steelhead during times of reduced habitat accessibility due to low flow conditions.

NC summer steelhead suffer primarily from poaching rather than legal recreational fishing. Summer steelhead are especially vulnerable to poaching due to the nature of their summer holding conditions and in many cases these habitats are on remote private property. They are highly susceptible to netting, spearing, and trapping by hand as they are highly visible and easily approachable in the holding pools (Roelofs 1983). In a study by Taylor and Barnhart (1996) on summer steelhead hooking mortality on the Mad and North Fork Trinity rivers, avoidance behavior was observed when summer steelhead were approached by divers, though the fish were still visible and approachable up to one meter (Taylor and Barnhart 1996). It was presumed that avoidance behavior and the resulting exposed positioning would render summer steelhead vulnerable to predation via spearfishing. Additionally, summer steelhead were observed to rarely leave their holding pools even when disturbed by diver observation (Taylor and Barnhart 1996). Taylor and Barnhart also found that temperature and hooking mortality had a positive relationship with temperatures over 20°C associated with most of the mortalities observed in the study. This correlation of temperature with hooking mortality rate is significant to summer steelhead adults that hold in-river during the hottest months of the year and thus, are even more vulnerable to fishing pressure.

Due to the remoteness of adult summer steelhead holding habitat, law enforcement of illegal fishing activity is difficult (Moyle et al. 2008). Evidence of poaching, mostly in the form of fishing tackle, has been apparent in areas of summer steelhead refugia on the Middle Fork Eel River that are closed to fishing (CDFG 1966 – 2018). Gillnets and weirs made of fencing have also been used to poach adult summer steelhead migrating upstream as recently as within the past decade (S. Thompson, CDFW, pers. comm., July 14, 2020). The Van Duzen River, Mad River, and Mattole River also experience unknown rates of poaching (Moyle et al. 2008). On the Van Duzen River, gear used for poaching has been found in areas closed to fishing while conducting annual adult summer steelhead surveys in three of the past nine years (S. Thompson, CDFW, pers. comm., July 14, 2020). Roads built to support timber harvest in more remote areas of these watersheds may contribute to poaching pressures due to increased accessibility (Roelofs 1983). Much of the NC summer steelhead watersheds are comprised of private land ownership, where poaching may be more likely due to greater privacy and less accessibility for law enforcement.

There are no ocean fisheries for steelhead in California and few are ever caught by ocean salmon anglers.

6. INFLUENCE OF EXISTING MANAGEMENT MEASURES

Several state and federal environmental laws apply to activities undertaken in California that may provide some level of protection for NC summer steelhead and their habitat. There are also restoration, recovery, and management plans along with management measures specific to hatchery operations, disease mitigation, gravel extraction practices, habitat restoration, recreational fishing, and research and monitoring that may benefit NC summer steelhead. The following list of existing management measures is not exhaustive.

6.1 Statewide Laws

6.1.1 National Environmental Policy Act and California Environmental Quality Act

The National Environmental Policy Act was enacted in 1970 to evaluate environmental impacts of proposed federal actions. Major federal actions such as the adoption of official policy, guidance documents for use of federal resources, or new federal programs or projects (40 C.F.R. §1508.18) will trigger the NEPA process. This process involves three levels of analysis: 1) Categorical Exclusion determination (CATEX), 2) Environmental Assessment (EA) / 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 behind the proposed action will prepare an EA, which concludes whether the action will result in significant environmental impacts. If no significant impact is expected the agency will issue a FONSI document. Alternatively, if the action is determined to have a potentially significant effect on the environment, an EIS will need to be prepared that includes 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 (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 NC steelhead DPS is listed as threatened under the federal ESA, NC summer steelhead may receive some protections from NEPA.

The California Environmental Quality Act (CEQA) is similar to NEPA; it requires environmental review of discretionary projects proposed to be carried out or approved by state and local public agencies unless an exemption applies (Public Resources Code § 21080). Under CEQA, the agency with the primary responsibility for carrying out or approving a project within its substantive authority (Lead Agency) is charged with determining whether an Environmental Impact Report (EIR), Negative Declaration, or Mitigated Negative Declaration is required for the project and preparing the appropriate document (California Code of Regulations, Title 14, § 15051). When there is substantial evidence the 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 effects and alternatives (Public Resources Code § 21082.2(a) and (d)). CEQA differs substantially from NEPA in requiring mitigation for significant adverse effects to a less than significant level unless overriding considerations are documented. CEQA requires an agency find that projects may have a significant effect on the environment if they have the potential to substantially reduce the habitat, decrease the number, or restrict the range of any rare, threatened, or endangered species (California Code of Regulations, Title 14, §§ 15065(a)(1), 15380). CEQA establishes a duty for public agencies to avoid or minimize such significant effects where feasible (California Code of Regulations, Title 14, § 15021). Impacts to NC summer steelhead, as part of the NC steelhead DPS, which is listed as threatened under the ESA, must be identified, evaluated, disclosed, and mitigated, as appropriate, under the Biological Resources section of an environmental document prepared pursuant to CEQA.

6.1.2 Clean Water Act and Porter-Cologne Water Quality Act

The Federal Water Pollution Control Act was established in 1948 and was the original version of the Clean Water Act. After being heavily amended in 1972, the Federal Water Pollution Control Act became known as the Clean Water Act (CWA). The CWA was enacted to establish regulations for the discharge of pollutants into waters of the United States and create surface water quality standards. Section 401 of the CWA requires any party applying for a permit or license for a project that may result in discharge of pollutants into 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 appropriate requirements of state law. Section 404 of the CWA forbids the discharge of dredged or fill material into waters of the United States, including wetlands, without a permit from the United States Army Corps of Engineers (USACOE). The USACOE is directed to issue their permit for the least environmentally damaging practicable alternative.

The Porter-Cologne Water Quality Act was established by the State of California in 1969 and has goals that align with those of the CWA such as establishing water quality standards and regulating the discharge of pollutants into state waters. The California State Water Resources Control Board (CASWRCB) and nine Regional Water Boards share responsibility for implementation and enforcement of the Porter-Cologne Act. These entities are required to formulate and adopt water quality control plans that describe how water quality standards will be attained.

In accordance with Section 303(d) of the CWA the USEPA, CASWRCB, and the Regional Water Boards are responsible for identifying “impaired” or polluted waters within the state. The listing leads to the development a TMDL program, which determines the “allowable” amount of sediment and temperature for the watershed. The primary purpose of the program is to assure that beneficial uses of water, such as steelhead habitat, are protected from detrimental increases in sediment and temperature and other pollutants defined in Section 502. The development and implementation of the TMDL is the responsibility of the State of California. The USEPA is required to review and approve the list of impaired waters and each TMDL. If USEPA cannot approve the list or a TMDL it is required to develop its own.

Waters within the boundaries of the NC steelhead DPS fall under the jurisdiction of the North Coast Regional Water Quality Control Board (NCRWQCB). In 2004, the NCRWQCB adopted the Total Maximum Daily Load Implementation Policy Statement for Sediment Impaired Receiving Waters in the North Coast Region in an effort to control sediment waste discharges and restore sediment impaired water bodies in the region (Resolution R1-2004-0087). The Sediment TMDL Implementation Policy states that Regional Water Board staff shall control sediment pollution by using existing permitting and enforcement tools. Redwood Creek, Mad River, Mattole River, Van Duzen River, Eel River (Upper, Middle, and Lower), as well as the North, Middle, and South Forks of the Eel are all listed as impaired by sediment and water temperature, and TMDLs have been developed.

6.1.3 Federal and California Wild and Scenic Rivers Act

The United States Congress enacted the federal Wild and Scenic Rivers Act (WSRA) in 1968 thereby creating the National Wild and Scenic River System (16 U.S.C. § 1271, et seq.). The WSRA is intended to preserve select rivers with outstanding natural, cultural, and recreational values as free flowing for the enjoyment of present and future generations. The WSRA prevents the federal government from licensing, funding, or otherwise assisting in dam construction or other projects on designated rivers or river segments. The WSRA does not give jurisdiction to the federal government over development projects on private property with designated rivers. 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.

Within the range of NC steelhead, only sections of the Eel River mainstem and some of its tributaries have been designated by the State (1972) and the federal government (1981) as wild and scenic. The following sections have been designated: from the river mouth of the Eel River mainstem to 100 yards below Cape Horn Dam; the Middle Fork from its confluence with the mainstem to the southern boundary of the Yolla Bolly Wilderness Area; the South Fork from its confluence with the mainstem to the Section Four Creek confluence; the North Fork from its confluence with the mainstem to Old Gilman Ranch; and the Van Duzen River from the confluence with the Eel River to Dinsmore Bridge. These sections encompass a total of 398 miles (National Wild and Scenic Rivers System 2020).

6.1.4 Lake and Streambed Alteration Agreements

Fish and Game Code Section 1602 requires entities to notify the Department of activities that "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." If the activity will substantially adversely affect an existing fish and wildlife resource, the Department shall issue a Lake or Streambed Alteration Agreement to the entity that includes reasonable measures necessary to protect the fish and wildlife resources (Fish and Game Code, §1602, subd. (a)(4)(B)). Authorization as provided under CESA or another appropriate mechanism in the Fish and Game Code is required to authorize take of species listed as candidates, threatened, or endangered under CESA.

6.1.5 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 (CDFA) 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 (California Code of Regulations., Title 3, § 8102, subdivision (w)). Waste discharge and water diversions associated with cannabis cultivation are regulated by the CASWRCB (California Code of Regulations., Title 3, § 8102, subdivision (p)).

6.1.6 Forest Practice Act

The Forest Practice Act was originally enacted in 1973 to ensure that logging operations in California are conducted in ways that aim to preserve and protect the State's fish, wildlife, forests, and streams. This law and the associated regulations adopted by the California Board of Forestry and Fire Protection are collectively known as the Forest Practice Rules (California Code of Regulations., Title 14, Chapter 4). The Forest Practice Rules implement the provisions of the Forest Practice Act in conjunction with other pertinent laws, including CEQA, Porter-Cologne, CESA, and the Timberland Productivity Act of 1982. The California Department of Forestry and Fire Protection (CalFire) enforces these laws and regulations as they pertain to logging on private land. Current forest practices are much improved as compared to before this act, particularly with regards to limiting the acreage of clear-cuts.

To further protect listed anadromous salmonids and their habitats, the Anadromous Salmonid Protection (ASP) rules were approved by the State Board of Forestry and Fire Protection (BOF) at their September 2009 meeting held in Sacramento, California. In October 2013, the BOF approved a revision to the rules under the “Class II-L Identification and Protection Amendments, 2013” rule package. The Final Statement of Reasons of the ASP rules adopted by the BOF explained that the rules are intended to protect, maintain, and improve riparian habitats for state and federally listed anadromous salmonid species. These rules are permanent regulations that replace the temporary Threatened or Impaired Watershed Rules which were originally adopted in July 2000 and readopted six times thereafter (CalFire and CDFW 2014).

The BOF’s primary objectives in adopting the ASP rules were: (1) to ensure the rules adequately protect listed anadromous salmonid species and their habitat, (2) to expand opportunities for restoring the species’ habitat, (3) to ensure the rules are based on sound science, and (4) to meet Public Resources Code Section 4553 for review and periodic revisions to Forest Practice Rules. The main goals of the BOF for the rule revisions were to update the rules based on new science; provide a high level of protection for listed species; contribute to anadromous salmonid habitat restoration; be consistent with partner agency mandates; and promote landowner equity, flexibility, and relief opportunities (CalFire and CDFW 2014). However, logging of large trees within the riparian zone can still occur.

6.1.7 Federal Power Act

The Federal Power Act and its major amendments are implemented and enforced by FERC, requiring licenses for specified dams that generate hydropower. One of the major amendments of the Federal Power Act mandates that FERC shall give consideration to the protection, mitigation, and enhancement of fish and wildlife and related spawning grounds and habitat (16 U.S.C. § 797, 803). The term for a hydropower license granted by FERC is typically 30-50 years.

As a federal agency, FERC must comply with federal environmental laws prior to issuing a new license or relicensing an existing hydropower project, which includes NEPA and ESA. When FERC considers whether to re-license a hydropower project, it must review the project to ensure it is consistent with federal and state comprehensive plans for improvement, development, or conservation of a waterway. 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 (including associated habitat) affected by the 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.

In addition to following the Federal Power Act and other applicable federal laws, the Porter-Cologne Water Quality Act requires a Water Quality Certification (WQC) from the CASWRCB for all non-federal dam operators. The CASWRCB must consult with the Department regarding the needs of fish and wildlife prior to issuing the WQC. Consequently, CASWRCB includes conditions in the WQC that seek to minimize adverse effects to native species, such as NC steelhead.

R.W. Matthews Dam

Humboldt Bay Municipal Water District (HBMWD) owns and operates Robert Matthews Dam in the upper watershed of the Mad River. Ruth Reservoir holds approximately 48,000 acre-feet of water primarily for industrial and municipal supply. HBMWD also diverts directly from the Mad River near Essex. In 1981, FERC granted an exemption for a hydroelectric plant at Matthews Dam. Power production is incidental to water released for the HBMWD's water supply function. There are also Ranney wells in the Mad River associated with HBMWD water usage.

Section 404 of the CWA requires that the HBMWD must secure a USACOE permit every five years. Since ESA species, including NC steelhead, are present, USACOE must consult with NMFS and USFWS prior to issuing any permit. NMFS must issue a Biological Opinion to the USACOE, determining if the HBMWD actions pose jeopardy to the continued existence of these species, or if the actions pose a significant adverse effect on critical habitat.

The HBMWD prepared a Habitat Conservation Plan (HCP) in support of an Incidental Take Permit under Section 10 of ESA. The HCP describes conservation measures to minimize and mitigate adverse impact to the listed species. Discharge below Matthews Dam is also regulated by the State of California Water Rights Board and the Department for the protection and preservation of fish.

Potter Valley Project

The PVP is operated by PG&E and consists of Scott Dam and Lake Pillsbury, Cape Horn Dam and Van Arsdale Reservoir, and a diversion tunnel and powerhouse located on the East Branch Russian River. Within the current FERC license set to expire in April of 2022, a flow schedule was prescribed based on recommendations from a Fisheries Review Group (consisting of scientists from PG&E, USFWS, NMFS, and the Department) convened to determine a flow schedule that minimizes the impact of the project on salmon and steelhead.

In January 2019, PG&E withdrew from the license application process. FERC subsequently issued a solicitation for new applicants and a consortium including the Mendocino County Inland Water and Power Commission, Sonoma County Water Agency, CalTrout, Humboldt County, and The Round Valley Indian Tribes notified FERC of their intent to prepare a Feasibility Study Report. The report was filed in May 2020 and describes a licensing proposal for the PVP that includes shifting the timing and magnitude of diversions to improve and protect downstream fishery resources, and the removal of Scott Dam to restore volitional anadromous fish access to the upper Eel River.

6.2 Species Recovery Plans and Regional Management Plans

6.2.1 Forest Plans

Northwest Forest Plan

In 1994, BLM and the Forest Service adopted the Northwest Forest Plan (NWFP) to guide the management of over 37,500 square miles of federal lands in portions of northwestern California, Oregon, and Washington. The NWFP includes an Aquatic Conservation Strategy that aims to maintain and restore distribution, diversity, and complexity of features and processes of watersheds that support aquatic and riparian species (Reeves et al. 2018). The NWFP's Aquatic Conservation Strategy is comprised of riparian reserves, key watersheds crucial to at-risk fish species, watershed analysis, and watershed restoration. Guidelines for riparian reserves prohibit timber harvest and manage roads, grazing, mining, and recreation to achieve the objectives of the Aquatic Conservation Strategy. Timber harvest is not allowed in key watersheds prior to completing a watershed analysis, although harvest can still occur in the riparian zone on private property. Tier 1 watersheds were selected based on presence of at-risk anadromous salmonids, bull trout, and resident fish species as well as having a high potential for being restored as part of a watershed restoration project. Tier 2 watersheds are important sources of high-quality water though they may not support at-risk fish species (USDA and USDI 1994).

National Forest Land and Resource Management Plans

Forest plans have been developed for Mendocino National Forest and Six Rivers National Forest to provide a framework for guiding ongoing land and resource management operations. The goal of these documents is to provide a management program that balances use and protection of forest resources. Forest plans provide guidance by way of general management direction and goals but do not include project-level recommendations. The plans also establish monitoring and evaluation requirements to ensure management actions are being implemented and verify estimates of outputs and effects of these actions.

Mendocino National Forest Land and Resource Management Plan
https://www.fs.usda.gov/detailfull/mendocino/landmanagement/planning/?cid=fsbdev3_004518&width=full

Six Rivers National Forest Land and Resource Management Plan
https://www.fs.usda.gov/detailfull/srnf/landmanagement/planning/?cid=stelprdb5084033&width=full

Six Rivers National Forest Watershed and Fisheries Restoration Program

The Six Rivers National Forest's Watershed and Fisheries Restoration Program goals are to implement the Aquatic Conservation Strategy which was developed to restore and maintain the ecological health of watersheds and aquatic ecosystems on federal lands (FEMAT 1993). In meeting the objectives of the Aquatic Conservation Strategy, the program implements restoration activities identified in the Department's Recovery Strategy for California Coho Salmon (CDFG 2004), and NMFS' Final Recovery Plan for Southern Oregon/Northern California Coast Coho Salmon (NMFS 2014).

6.2.2 Habitat Conservation Plans

Several timber companies in California own lands with important aquatic habitat for listed salmonids. In Humboldt County alone there are over one million acres of timber production land, accounting for about 44% of county land (Humboldt County 2012). Developing a Habitat Conservation Plan, including aquatic habitat, is a regulatory condition of harvesting timber on these lands. A primary goal of an aquatic conservation plan is to conserve aquatic habitat for and mitigate impacts on listed species such as Chinook Salmon, Coho Salmon, and steelhead.

Humboldt Redwood (formerly Pacific Lumber) Company

The Humboldt Redwood Company HCP covers 211,700 acres of private coast redwood and Douglas fir forest in Humboldt County (HRC 2015). It has a 50-year HCP/incidental take permit that was executed in 1999, revised in 2015 as part of its adaptive management strategy, and expires on March 1, 2049. The HCP includes an Aquatics Conservation Plan with measures designed to maintain or achieve, over time, a properly functioning aquatic habitat condition for salmonids. These conservation measures include assessing the potential risk of soil erosion from roads and from exposed forest soils; stream flows and potential changes to flows; shade cover; stream channel conditions including large wood in streams, distribution of deep pools and spawning gravels; and habitat and distribution of fish. The plan also includes periodically analyzing the data to adaptively manage prescriptions based on the watershed's progress toward properly functioning conditions.

Green Diamond Aquatic Habitat Conservation Plan

GDRCO owns timber lands in three watersheds inhabited by NC summer steelhead—Redwood Creek, Mad River, and Eel River. They prepared an Aquatic Habitat Conservation Plan (AHCP) and Candidate Conservation Agreement with Assurances and subsequently received a Biological Opinion from NMFS in 2007 (GDRCO 2006). The plan identifies conservation measures that are expected to minimize and mitigate the impacts from take of anadromous species and includes a variety of protection measures designed to restore and maintain riparian and upslope processes that create, restore, and maintain aquatic habitat. Conservation measures focus on LWD recruitment, shade retention, and control of the delivery of coarse and fine sediments to aquatic habitat from forest management activities.

6.2.3 Other Restoration and Management Plans

Coastal Multispecies Recovery Plan for California Coastal Chinook Salmon, Northern California Steelhead, and Central California Coast Steelhead (NMFS 2016) – A recovery plan developed by NMFS in 2016 providing a framework for conservation and survival of three ESA-threatened species of salmon and steelhead. The Department funds recovery actions identified in this plan primarily through the Fisheries Restoration Grant Program. https://www.fisheries.noaa.gov/resource/document/final-coastal-multispecies-recovery-plan-california-coastal-chinook-salmon

Eel River Action Plan – A compilation of information and recommendations prepared by the Eel River Forum in 2016. https://caltrout.org/wp-content/uploads/2016/06/2016.03.FINAL_.EelRiverActionPlan.ERF_.pdf

Eel River Salmon and Steelhead Restoration Action Plan (CDFG 1997a) – A 1997 CDFG plan with restoration actions to address known problems in the Eel River watershed that have caused declines in Salmon and steelhead populations.

Mad River Total Maximum Daily Loads for Sediment and Turbidity – Document by the U.S. Environmental Protection Agency establishing TMDLs for sediment and turbidity in the Mad River in accordance with Section 303(d) of the CWA.

https://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/mad_river/pdf/Mad-TMDL-122107-signed.pdf

Management Plan for the Summer Steelhead Rainbow Trout (*Salmo gardneri*) in the Middle Fork of the Eel River (Jones and Ekman 1980) – A 5-year Middle Fork Eel River summer steelhead management plan from 1980 jointly prepared by CDFG and USFS and adopted as interim resource management direction prior to the Mendocino National Forest management plan.

Mattole Headwaters Groundwater Management Plan – A plan developed by Sanctuary Forest's Mattole Flow Program to manage, protect, and enhance groundwater and surface water in compliance with California Water Code section 10750 et. sec. http://sanctuaryforest.org/wp-content/uploads/2014/12/Mattole-Headwaters-Groundwater-Management-Plan.pdf

Mattole Integrated Coastal Watershed Management Plan – A 2009 report by the Mattole River and Range Partnership detailing a plan for watershed restoration and water quality management with a series of strategies for each specified management area. http://www.mattole.org/wp-content/uploads/2014/08/WatershedPlan_Final_w_Cover.pdf

Mattole River Total Maximum Daily Loads for Sediment and Temperature – Document by the U.S. Environmental Protection Agency establishing TMDLs for sediment and temperature in the Mattole River in accordance with Section 303(d) of the CWA.

https://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/mattole_river/110707/mattole.pdf

Middle Fork Eel River Total Maximum Daily Loads for Temperature and Sediment – Document by the U.S. Environmental Protection Agency establishing TMDLs for temperature and sediment in the Middle Fork Eel River in accordance with Section 303(d) of the CWA. https://19january2017snapshot.epa.gov/www3/region9/water/tmdl/middleeel/tmdl.pdf

North Fork Eel River Total Maximum Daily Loads for Sediment and Temperature – Document by the U.S. Environmental Protection Agency establishing TMDLs for sediment and temperature in the North Fork Eel River in accordance with Section 303(d) of the CWA. https://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/eel_river_north_fork/pdf/final.pdf

Redwood Creek Integrated Watershed Strategy – A 2006 report prepared by the Redwood Creek Watershed Group with the goal of improving and protecting water quality and supply and aquatic and riparian habitat in the Redwood Creek watershed. https://www.nps.gov/redw/learn/management/upload/RWC%20IWS%20Final.pdf

Redwood Creek Sediment Total Maximum Daily Load – Document by the U.S. Environmental Protection Agency establishing a TMDL for sediment in Redwood Creek in accordance with Section 303(d) of the CWA. https://archive.epa.gov/region09/water/archive/tmdl/redwood/rwctmdl.pdf

South Fork Eel River Total Maximum Daily Loads for Sediment and Temperature – Document by the U.S. Environmental Protection Agency establishing TMDLs for temperature and sediment in the South Fork Eel River in accordance with Section 303(d) of the CWA. https://19january2017snapshot.epa.gov/www3/region9/water/tmdl/eel/eel.pdf

Steelhead Restoration and Management Plan for California (McEwan and Jackson 1996) – A 1996 CDFG statewide steelhead management plan providing guidelines for steelhead restoration and management that can be incorporated in stream-specific project planning. https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=3490

Strategic Plan for Management of Northern California Steelhead Trout – A 1998 CDFG document listing management actions proposed to continue, expand, or be implemented in response to NMFS' proposal to list the Northern California steelhead ESU as threatened under ESA.

Upper Main Eel River and Tributaries Total Maximum Daily Loads for Temperature and Sediment – Document by the U.S. Environmental Protection Agency establishing TMDLs for temperature and sediment in the upper mainstem Eel River and tributaries in accordance with Section 303(d) of the CWA. The upper mainstem Eel River was added to California's 303(d) impaired water list in 1992.

https://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/eel_river_upper_main/pdf/uer-tmdl-final-12-28.pdf

Van Duzen River and Yager Creek Total Maximum Daily Load for Sediment – Document by the U.S. Environmental Protection Agency establishing TMDLs for sediment in the Van Duzen River and Yager Creek in accordance with Section 303(d) of the CWA. The Van Duzen River was added to California's 303(d) impaired water list in 1992. https://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/vanduzen_river/pdf/vanduzen.pdfhttps://www.waterboards.ca.gov/northcoast/water_issues/programs/tmdls/vanduzen_river/pdf/vanduzen.pdf

Van Duzen Watershed Project: Watershed Management Plan – A 2010 document compiled as part of the Van Duzen Watershed Project providing critical information on the state of the Van Duzen watershed and goals and strategies for improved management.

6.3 Hatchery Operations

In November 2014, the Department submitted a final HGMP for the MRH winter steelhead program in the Mad River. The HGMP provides a framework for operating the program while minimizing potential impacts on ESA listed salmonids within the project area. The Department has implemented measures to reduce the impacts of the MRH steelhead program on natural-origin steelhead. These include: (1) marking 100% of the hatchery steelhead with an adipose fin clip to allow the identification of hatchery fish in the fishery, at the hatchery, and on the spawning grounds; (2) reducing steelhead production from 250,000 smolts released to 150,000; (3) increasing the effective population size of the hatchery broodstock to reduce the risk of inbreeding depression, if not enough natural-origin fish available, by splitting the eggs from each female into two lots and fertilizing each lot with a different male; (4) increasing the time of broodstock selection and spawning based on a bell-shaped curve that distributes the spawning throughout the complete run, and (5) increasing the number of natural fish used for broodstock by spawning natural-origin fish at a 1:1 ratio with hatchery-origin fish. Natural broodstock are now mainly collected offsite by the Mad River Steelhead Stewards Volunteer Program. The Department critically examines each fish prior to use for broodstock.

6.4 Disease

Hatchery-origin steelhead may have an increased risk of contracting and spreading fish diseases than wild fish due to relatively high rearing densities and stress from rearing conditions (Saunders 1991). These diseases could potentially be transmitted from hatchery-raised fish to wild populations post-release (Hastein and Lindstad 1991). Wild fish have the potential to transmit disease to hatchery stocks as well, though disease infection in wild populations is not well documented (Olivier 2002). Several bacterial infections have been identified in MRH steelhead that are caused by *Flavobacterium psychrophilum* (coldwater disease bacteria), *Flavobacterium columnare* (columnaris), *Flavobacterium branchiophilum* (bacterial gill disease), and motile *Aeromonas/Pseudomonas* spp. bacteria. A number of external parasites have also been identified, including *Gyrodactylus* spp., *Ambiphrya* spp., *Ichthyobodo necator* (costia), *Tetrahymena* sp., and *Ichthyophthirius multifilis* (ich). The parasite, *Tetracapsuloides bryosalmonae*, which is the causative agent of proliferative kidney disease and endemic to the Mad River watershed, has been observed in MRH production fish, as well (CDFW 2016).

The MRH is operated in compliance with all applicable fish health guidelines from the American Fisheries Society's *Blue Book: Suggested Procedures for the Detection and Identification of Certain Finfish and Shellfish Pathogens,* 2012 edition). These fish health guidelines ensure that fish health is monitored, sanitation practices are followed, and hatchery-origin fish are reared and released in good health. Department fish pathologists monitor hatchery programs regularly and conduct fish examinations at each life stage including tests for viruses, bacteria, parasites, or pathological changes. MRH also uses ultraviolet treatment to remove pathogens from the water that is used for fish culture purposes (CDFW 2016).

6.5 Gravel Extraction

The State Surface Mining and Reclamation Act (SMARA) of 1975 requires that all gravel mining operations include a Reclamation Plan describing the operation in detail, the environmental setting, and measures that will minimize adverse environmental effects caused by the operation. In 1991, the California Department of Conservation's Division of Mine Reclamation was created to provide a measure of oversight for local governments required to administer SMARA throughout California. Reclamation plans are further subject to CEQA and local land use code; CWA Section 401 (State Certification of Water Quality) and Section 404 (USACOE) that regulate the discharge of dredged or fill material into the waters of the United States. Operators may be required to obtain permits from the USACOE for dredge and fill operations, or other gravel extraction activities under Sections 401 and 404 of the CWA and/or Section 10 of the 1899 Rivers and Harbors Act. Instream gravel mining is subject to, and projects could be authorized by, a Lake or Streambed Alteration Agreement (Fish & G. Code § 1600 *et seq*.). Consultation with the NMFS and/or the Department pursuant to CESA and ESA may also be warranted.

NMFS compiled a National Gravel Extraction Guidance document in 2005 to replace their 1996 National Gravel Extraction Policy. The objectives of the guidance document are to 1) assist NMFS staff in determining if proposed gravel extraction operations comply with federal law, while 2) avoiding, minimizing, and mitigating any negative impacts on anadromous fishes and their habitats. The document is primarily focused on instream gravel mining rather than gravel extraction in marine environments. NMFS recommends that gravel mining operations should not affect migration, spawning, or rearing of anadromous fish. They also should not adversely impact existing or historical anadromous habitat. The 2005 National Gravel Extraction Guidance document applies nationwide but is flexible in that recommendations are made on a project-specific basis (Packer et al. 2005).

In 1992, the Humboldt County Board of Supervisors created the County of Humboldt Extraction Review Team (CHERT) to provide scientific oversight for gravel extraction projects. CHERT provides site specific recommendations on extraction designs submitted by the operators and their consultants, in addition to the previously mentioned state and federal agencies. There are currently several active gravel mining sites administered by CHERT including sites on the Mad River, the Eel River (and tributaries) and the Mattole River. One additional project on the Mad River, operated by the Blue Lake Rancheria, was not required to obtain permits from the State of California or Humboldt County, and thus, operates outside of the CHERT program. Their mining plans are reviewed by the USACOE and NMFS (CHERT 2019) since they have sovereign rights.

6.6 Habitat Restoration and Watershed Management

Federal, state, and local agencies involved in substantial habitat restoration projects in the NC summer steelhead range include NMFS, USFWS, USFS, The Department, the California State Wildlife Conservation Board, and NCRWQCB. These agencies administer restoration grant programs (e.g., Fisheries Restoration Grant Program). Fisheries Restoration Grant Program has funded many projects statewide and throughout the NC summer steelhead range. Restoration work in the region falls into the following categories: instream bank stabilization, improving fish passage, screening, instream habitat restoration, riparian habitat restoration, upslope watershed restoration, and water conservation measures. Certain projects targeting steelhead habitat have been completed in the Redwood Creek, Mad River, Van Duzen River, and Mattole River watersheds over the past 20 years and may have improved summer steelhead habitat. There are also potentially a few projects conducted in Humboldt and Mendocino counties funded through Propositions 1 and 68 that may have directly or indirectly benefitted NC summer steelhead.

The California Steelhead Report and Restoration Card program has funded various types of restoration and, to a lesser degree, monitoring projects since 1993, including instream habitat improvement, species monitoring, outreach and education, and watershed assessment and planning. A number of these projects have been implemented in NC summer steelhead watersheds including Redwood Creek, Mad River, South Fork Eel River, and Mattole River. On Redwood Creek a NCFR project specifically focused on summer steelhead recovery was funded for fiscal year 1997 – 1998 to identify cold water habitat that could be improved by reintroduction of woody debris with the goal of increasing the abundance of summer steelhead and other salmonids.

6.7 Commercial and Recreational Fishing

There are no commercial fishing operations for NC steelhead, however many waters within their distribution are popular destinations for recreational fishing. Sport fishing regulations do not allow harvest of naturally produced NC steelhead — all natural-origin fish must be immediately released. The fishery targets hatchery winter steelhead, which may be harvested. Through seasonal closures and gear restrictions, the regulations limit the potential impact on NC summer steelhead.

Low-flow trout and salmon angling restrictions are implemented by way of fishery closures when flows in Northern California rivers dip below their specified minimums (California Code of Regulations., Title 14, § 8.00, subdivision. (a)). These conditional stream closures are in place from September 1 through January 31 on the Mad River and October 1 through January 31 on the Eel River, Mattole River, Redwood Creek, and Van Duzen River. Barbless hooks are required in all anadromous waters. Artificial lures are also required during certain times of the year to decrease hook and release mortality. Some sections of these rivers known to harbor summer steelhead during holding and spawning are closed to fishing year-round. For example, the area from Cowan Creek to Deer Creek on the Mad River, the reach below Eaton Falls on the Van Duzen River, and the reach from Bar Creek up to Uhl Creek on the Middle Fork Eel River are closed all year. Seasonal closures in other reaches are in place to decrease angler interactions with steelhead during times coinciding with upstream migration when they are more physiologically vulnerable due to higher temperatures. These closures are typically during April and May for summer steelhead and October or November through December for winter steelhead.

6.8 Research and Monitoring Programs

6.8.1 Summer Steelhead Monitoring

Summer steelhead monitoring efforts in the form of snorkel surveys occur on Redwood Creek, Mad River, Middle Fork Eel River, Van Duzen River, and Mattole River. These surveys are not part of a comprehensive monitoring plan, but most are conducted annually when adequate funding and personnel are available. The Department conducts summer steelhead monitoring on the Middle Fork Eel and Van Duzen rivers, which is funded by a grant through the Sport Fish Restoration Act which also includes a number of other projects including adult salmonid monitoring at Van Arsdale Fish Station and outmigrant trapping of juvenile salmon and steelhead in the upper Eel River. Middle Fork Eel and Van Duzen surveys currently cover the full extent of summer steelhead habitat in both mainstems. The Mad River Alliance currently conducts summer steelhead surveys on the Mad River in collaboration with the Department, Blue Lake Rancheria, Bureau of Land Management, GDRCO, Hollie Hall and Associates, Stillwater Sciences, Timberland Resources Company, and the Wiyot Tribe (Mad River Alliance 2020). Surveys on the Mad River also focus on mainstem summer steelhead habitat. Redwood Creek summer steelhead surveys are conducted by the National Park Service and currently cover about 40 miles of the mainstem. Mattole River summer steelhead are monitored by the Mattole Salmon Group with surveys that generally cover the entirety of the mainstem and some portions of Bear and Honeydew creeks (N. Queener, Mattole Salmon Group, pers. comm., February 4, 2020). In all five streams there is some amount of tributary habitat that is not surveyed but may support summer steelhead.

6.8.2 California Monitoring Program

The California Monitoring Program (CMP), formerly the California Coastal Monitoring Program, was developed to monitor California's coastal Salmon and steelhead populations using statistically rigorous modeling combined with a variety of sampling and survey methods. The Department and NMFS have begun implementing the CMP in coastal watersheds in California for about a decade. The CMP is currently being conducted in the following watersheds within the NC steelhead DPS:

- Redwood Creek
- Humboldt Bay
- Mattole River
- South Fork Eel River
- Mendocino Coast
- Gualala River

The sampling framework for species monitoring is established using Generalized Random Tessellation Stratification (GRTS) to select stream sample units. GRTS is a widely used method for selecting random, spatially balanced samples and compromises between even distribution of samples and randomization. Generally, adult monitoring of Chinook Salmon includes counts of live fish, carcasses, and redds. Coho Salmon and steelhead adults are enumerated through redd surveys (CDFG 2011). The CMP sample frame in Northern California is currently designed around Coho and Chinook Salmon. Adult winter steelhead population estimates are generated but they generally do not account for the entire run due to spatial and temporal limitations of the Salmon-based sample frame. Summer steelhead are not enumerated through CMP, though ancillary information on summer steelhead may be collected.

6.8.3 Adult Salmonid Sonar Monitoring

The Department's Anadromous Fisheries Resource Assessment and Monitoring Program initiated a sonar monitoring program on the Mad River in 2013 to enumerate California coastal Chinook Salmon, Southern California/Northern Oregon Coho Salmon, and NC steelhead. The ARIS sonar detection coupled with methods for species apportionment have allowed the Department to develop annual population estimates for these three species. Sonar deployment, however, only extends from late August through March at the latest, so it does not enumerate summer steelhead.

A dual frequency identification sonar (DIDSON) device was deployed seasonally by the United States Geological Survey on Redwood Creek starting in 2009 to enumerate Chinook Salmon, Coho Salmon, and steelhead. This study was funded by the Department's Fisheries Restoration

Grant Program through the 2016-2017 monitoring season. As is the case with the sonar device on the Mad River, the DIDSON was removed from Redwood Creek each season prior to upstream migration of the summer steelhead run (Metheny and Duffy 2014).

6.8.4 Habitat Conservation Plan Monitoring

Both the Humboldt Redwood Company HCP and the GDRCO AHCP have elements of monitoring included. These monitoring operations can benefit NC summer steelhead by evaluating effects of land use operations on essential habitat. The Green Diamond AHCP incorporates effectiveness monitoring across the plan area in four categories: 1) rapid response monitoring, 2) response monitoring, 3) long-term trend monitoring/research, and 4) an experimental watersheds program. Projects in these categories include water temperature monitoring, spawning substrate permeability monitoring, sediment delivery and mass wasting monitoring, channel response monitoring, LWD monitoring, summer juvenile salmonid population estimation, out-migrant trapping, streamside slope assessments, long-term habitat assessments, BACI studies of harvest vs. non-harvest areas, and more (GDRCO 2006).

The Humboldt Redwood Company HCP, as part of the Aquatics Conservation Plan, includes aquatic monitoring mainly consisting of compliance and effectiveness monitoring. Compliance monitoring includes third-party monitoring, a timber harvest plan (THP) checklist ensuring THPs incorporate all relevant elements from the Aquatics Conservation Plan, a best management practice evaluation program, and application of findings. Effectiveness monitoring encompasses monitoring instream and upslope conditions including but not limited to LWD, riparian buffers, water temperature, sediment, and hillslope to evaluate the effectiveness of the Aquatic Conservation Plan (HRC 2015).

7. SUMMARY OF LISTING FACTORS

CESA's implementing regulations identify key factors relevant to the Department's analyses and the Commission's decision on whether to list a species as threatened or endangered. 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 (California Code of Regulations., Title 14, § 670.1, subdivision (i)).

This section provides summaries of information from the foregoing sections of this status review, arranged under each of the factors to be considered by the Commission in determining whether listing is warranted.

1. **Present or threatened modification or destruction of its habitat**: A multitude of human activities in combination with natural events and unstable topography of the region have led to extensive habitat degradation and fragmentation (see *Section 5.8 Habitat*

Condition). Although management practices have been improved, legacy effects of these disturbances persist within NC summer steelhead watersheds. Timber harvest, gravel mining, livestock grazing, and agriculture are continued land uses in these watersheds today and pose habitat threats to summer steelhead including sediment loading, turbidity, decreased flow, and increased water temperatures. Timber harvest is the most widespread land use in Humboldt County (Humboldt County 2012). Agriculture, including cattle ranching, is also a primary land use in the county. Together, timberland and agriculture make up about 60% of Humboldt County's unincorporated rural land use (Humboldt County 2012). Gravel mining occurs in NC summer steelhead watersheds, but to a much smaller degree than timber production and ranching (Humboldt County 2012). Cannabis cultivation is also an increasing threat, especially illicit grow operations, which destroy habitat to make space for the grows and use unregulated water diversions and pesticides (NMFS 2016b).

Dams in the NC summer steelhead range have also curtailed habitat use and affected habitat suitability. R.W. Matthews Dam on the Mad River and Scott Dam on the upper mainstem Eel River are both complete barriers to migration and eliminate all upstream habitat. Construction of Scott Dam likely caused the loss of most summer steelhead in the upper Eel River (NMFS 2016b). Matthews Dam and the PVP have also altered natural flow regimes in their respective river systems. Flows seem to be positively affected by releases from Matthews Dam on the Mad River (HBMWD 2004), but releases from Lake Pillsbury have been observed to increase water temperatures in the Eel River during the late summer and early fall of some years (PG&E 2017).

The Department considers present or threatened modification or destruction of habitat to be a significant threat to the continued existence of NC summer steelhead.

2. **Overexploitation**: There are no fisheries that target NC summer steelhead for harvest. Quantitative accounting of the effects of freshwater steelhead fisheries in coastal streams is not available; however, hooking mortality from catch-and-release fishing on hatchery steelhead likely impacts NC summer steelhead (Taylor and Barnhart 1996). Anecdotal evidence of poaching also exists, especially in remote areas of NC summer steelhead watersheds (CDFG 1966 – 2018; S. Thompson, CDFW, pers. comm., July 14, 2020), but there are no directed studies that quantify illegal take or identify poaching to be at a level that would affect NC summer steelhead abundance. Overexploitation has not been identified as a primary threat to summer steelhead; however, we note that with populations at low abundances even small amounts of fishing pressure can have significant impacts on these populations. The Department considers overexploitation to be a low to moderate threat to the continued existence of NC summer steelhead, but

further directed study is warranted to confirm this threat level.

3. **Predation**: NC summer steelhead experience natural predation in both the freshwater and marine environments (see *Section 5.5 Predation*). Steelhead have coevolved with many natural native aquatic and terrestrial predators including birds, pinnipeds, otters, and other fish species, though natural predation rates can increase when conditions are poor or population abundances are low. A predation threat that has been of particular concern since its introduction into the Eel River system is the non-native Sacramento Pikeminnow, which has spread throughout the drainage. These fish may consume juvenile steelhead as they emigrate to the ocean or when steelhead abundance is high in areas where they co-occur with large Sacramento Pikeminnow. Sacramento Pikeminnow likely pose greater threats in lower water years when water temperatures are higher and more suitable for the species. A study conducted in the mainstem Eel and South Fork Eel rivers showed that Sacramento Pikeminnow can consume steelhead but are generally opportunistic predators (Nakamoto and Harvey 2003). Comprehensive evaluation of Sacramento Pikeminnow predation throughout the Eel River drainage could help determine the level of impact these fish are having on salmonids including NC summer steelhead. Otters and pinnipeds have also been observed to prey upon steelhead in the Mad River and otters likely have easy access to summer steelhead during their holding period (M. Sparkman, CDFW, pers. comm., July 10, 2020). The Department considers predation to be a low to moderate threat to the continued existence of NC summer steelhead based off the available data on Sacramento Pikeminnow foraging and anecdotal accounts of pinniped and otter predation. Further directed study is warranted to confirm the level of impact of these predation threats on NC summer steelhead.

4. **Competition**: Steelhead competition occurs mostly with other salmonids but can occur when Sacramento pikeminnow are present as well. Competition tends to be more pronounced between sympatric populations of species that are closely related as they occupy similar habitat and ecological niches. Although Coho Salmon and non-native Brown Trout have been observed as competitors, steelhead are more generalist in their habitat selection, which increases their resilience to the pressures of competition (see *Section 5.6 Competition*). Effects of natural inter- and intraspecies competition are not considered by the Department to be a major threat to the continued existence of NC summer steelhead.

5. **Disease**: There are no documented disease issues in NC summer steelhead streams. The Klamath-Trinity drainage has experienced extensive outbreaks of *C. shasta*, particularly in Chinook Salmon, but this parasite has not been found to infect steelhead. Hatcheries tend to perpetuate disease transmission and hatchery-origin steelhead are generally

more susceptible to pathogen infection than natural-origin steelhead; however, MRH, the only hatchery within the NC summer steelhead range, has not experienced any serious disease outbreaks. Furthermore, MRH has specific protocols including fish health monitoring and ultraviolet water treatment to help prevent disease issues (CDFW 2016). The Department does not consider disease to be a threat to the continued existence of NC summer steelhead.

6. **Other natural occurrences or human-related activities**: With the impending effects of climate change, the limited amount of NC summer steelhead habitat will likely continue to decline in quality and extent. California's north coast may experience pronounced climate change impacts including rising water temperatures, intensified flooding, more frequent and persistent drought conditions, lower summer baseflows, altered hydrography especially in watersheds impacted by snowmelt and large-scale historical logging, ocean acidification, and sea level rise (see *Section 5.2 Effects of Climate Change*). These climate change impacts may have detrimental effects on steelhead habitat quality and availability. Water temperatures could become lethal more frequently and reduce thermal stratification of essential pool habitat, catastrophic floods have the potential to cause mass wasting and fill remaining pool habitat, drought conditions may cause streams to dry out earlier and to a greater extent, low flow conditions will impede fish passage, and estuary environments might be inundated by saltwater intrusion due to sea level rise.

   The winter steelhead propagation program at MRH may have some genetic and ecological impacts on natural steelhead populations (see *Section 5.4.1*), though these impacts have not been directly studied in Mad River steelhead populations. Although there is likely a high degree of separation between summer steelhead adults and hatchery winter steelhead adults on the spawning grounds, a small amount of interbreeding may still occur. Genetic consequences of hybridization between hatchery and natural fish can have negative impacts on fitness of natural populations (Araki et al. 2009). Hatchery steelhead can also increase competition and predation in wild populations (Steward and Bjornn 1990) and increases the risk of spreading disease (Hastein and Lindstad 1991), though there is no documentation of these issues specific to MRH on the Mad River.

   Natural occurrences and human-related activities are considered by the Department to be significant threats to the continued existence of NC summer steelhead.

8. PROTECTIONS AFFORDED BY CESA LISTING

It is the policy of the State to conserve, protect, restore, and enhance any endangered or threatened species and its habitat (Fish and Game Code § 2052). The conservation, protection, and enhancement of listed species and their habitat is of statewide concern (Fish and Game

Code § 2051(c)). If listed, NC summer steelhead would receive the protections of CESA, including the prohibition against unauthorized take. Under CESA, "take" is defined as to hunt, pursue, catch, capture, or kill, or attempt to hunt, pursue, catch, capture, or kill (Fish and Game Code § 86). Any person violating the take prohibition would be punishable under state law. The California 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 and Game Code §§ 2081, 2081.1, 2086, & 2835).

When take is authorized pursuant to an incidental take permit under CESA, impacts of the taking caused by the activity must be minimized and fully mitigated according to state standards (Fish and Game Code § 2081, subdivision (b)). Additionally, the Department would be prohibited from approving incidental take permits which could jeopardize the continued existence of the species in the state (Fish and Game Code § 2081, subdivision (c)).

In addition to restrictions on take of CESA-listed species, public agencies should not approve any proposed projects that could jeopardize the persistence of an endangered or threatened species, or that would adversely modify or destroy essential habitat of the species, if there are reasonable and prudent alternatives to the project that would be consistent with conservation of the species or its habitat (Fish and Game Code § 2053). If economic, social, or other conditions make such alternatives infeasible, mitigation and enhancement measures must be included to offset negative effects from project implementation (Fish and Game Code § 2054).

Listing NC summer steelhead under CESA could prompt changes to regulations on coastal steelhead fisheries that may have indirect positive impacts on NC summer steelhead incidental catch. Listing NC summer steelhead under CESA could also increase the chances that federal and state resource management agencies will allot funding for protection and recovery efforts that benefit NC summer steelhead. However, implementation of any recovery actions or other management would be complicated by the inability to visually differentiate summer from winter steelhead in most life stages.

9. LISTING RECOMMENDATION

CESA directs the Department to prepare a status review report for NC summer steelhead using the best scientific information available (Fish and Game Code, § 2074.6). CESA also directs the Department to indicate in its status review report whether the petitioned action is warranted (Fish and Game Code, § 2074.6; California Code of Regulations, Title 14, § 670.1, subdivision (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 and Game 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 Game Code, § 2067). To be listed under CESA, a candidate species must be considered a separate species or subspecies (Fish and Game Code, § 2062).

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). 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 (*Central Coast Forest Assn. v. Fish & Game Com.* (2018) 18 Cal. App. 5th 1191, 1236, *citing Cal. Forestry Assn., supra,* 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 has recognized that similar populations of a species can be grouped for efficient protection of bio- and genetic diversity (*Cal. Forestry Assn. v. Cal. Fish and G. Com.*, supra, 156 Cal.App.4th 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*.).

The summer steelhead ecotype represents an important diversity element of the steelhead species and provides genetic variation within NC steelhead populations that will help them persist through short-term and long-term changes in the environment. For this status review, NC summer steelhead were evaluated as if they constituted a distinct subspecies as was requested by the Petition. However, given the best available scientific information on the biological and ecological relationship between NC summer steelhead and other population components of California steelhead, the Department concludes that the NC summer steelhead ecotype should not be listed as a distinct species or subspecies under CESA.

The Department's status review recommendation is submitted to the Commission in an advisory capacity based on the best available science. In consideration of the best available scientific information contained herein, the Department recommendation is that NC summer steelhead should not be listed as a threatened or endangered species under CESA. This recommendation was based on the following considerations:

- **NC summer and winter steelhead are partially, but not fully, reproductively isolated.** Although summer steelhead are sometimes able to pass flow-dependent barriers that are not always passable by winter steelhead, spawning grounds likely overlap to some degree, especially in low-flow years. Spawning distribution is somewhat unclear in the NC steelhead DPS and varies between watersheds and spawning tributaries, but it is likely that the two ecotypes overlap both temporally and spatially during spawning. There is genetic evidence, in the form of heterozygosity at the GREB1L genomic region, that suggests interbreeding of summer and winter steelhead and an interconnected genetic legacy between the two ecotypes (Prince et al. 2017; Pearse et al. 2019; Kannry et al. 2020). This suggests that interbreeding occurs, at least occasionally, between the two ecotypes. Interbreeding of the two runs likely also occurred historically, though quantifying natural levels of interbreeding is difficult. Additionally, salmonid habitat is dynamic, so levels of interbreeding have probably fluctuated over multiple time scales and spatial scales (Ford et al. 2020).

- **Summer and winter steelhead ecotypes within the NC steelhead DPS are not genetically distinct.** Genetic exchange exists between summer and winter steelhead on contemporary and evolutionary time scales. There is substantial scientific literature and consensus within the scientific community that summer steelhead are more closely related to sympatric winter steelhead than to their life history equivalents in other watersheds (Chilcote et al. 1980; Thorgaard 1983; Nielsen and Fountain 1999; Arciniega et al. 2016; Prince et al. 2017). The genomic region consisting of GREB1L, ROCK1, and the intergenic region, although strongly associated with migration timing (Prince et al. 2017; Micheletti et al. 2018; Narum et al. 2018; Pearse et al. 2019), only accounts for a small amount of genetic variation between the two ecotypes. This concept is supported by recent studies that have looked across much larger portions of the genome (Micheletti et al. 2018; Ford et al. 2020). Summer and winter phenotypes, when studied across the migratory period, showed a continuum of allele frequencies with overlap among fish carrying the summer and winter migration variants (Pearse et al. 2019), which showed that individual fish can carry the alleles for both summer and winter migration timing. This was not observed by Prince et al. (2017) as they selected study populations that exhibited only the extremes of each life history type (Prince et al. 2019). Offspring from parents with alternate migration timing genotypes can be a mix of early, late, and intermediate migratory genotypes (Pearse et al. 2019) and heterozygotes may act as a reservoir for early migration alleles (Ford et al. 2020), though this is likely dependent on habitat conditions and patterns of dominance at the GREB1L/ROCK1 genomic region, which are not fully understood.

Pacific salmonid ESUs and DPSs are generally defined by geography as guided by patterns of genetic variation driven by geographic proximity rather than phenotype. Management units based solely on a phenotypic characteristic such as migration timing would not accurately reflect evolutionary lineages (Waples and Lindley 2018). There are also still areas of uncertainty surrounding the genetics behind run timing expression. Although there is a strong correlation of the GREB1L/ROCK1 genomic region with adult run timing, the mechanisms underlying expression of migration phenotypes are not yet fully understood (Ford et al. 2020). As a point of general agreement, Ford et al. (2020) advised that conservation units should continue to be defined by patterns of genetic diversity across the genome rather than by variation in small genomic regions associated with specific life history traits.

Summer steelhead are not present in all NC steelhead DPS streams; however, given the genetic connectivity within the DPS, summer steelhead should not be considered alone when assessing extinction risk of this segment of the steelhead species. Migration timing is a major difference between these ecotypes, but winter steelhead share many characteristics of summer steelhead and are similar in more ways than they are different. Given the significant overlap in the biological and ecological needs of summer and winter steelhead, factors threatening the survival of summer steelhead most likely impact winter steelhead to a similar degree.

This reasoning supports the ongoing management of summer and winter steelhead under the NC steelhead DPS. Because the Petition specifically requested the listing of NC summer steelhead under CESA, the Department did not evaluate the status of NC winter steelhead populations in this review (Fish and Game Code, § 2074.6).

10. ALTERNATIVES TO LISTING

If the Commission determines that listing under CESA is not warranted, NC summer steelhead will revert to the unlisted status under state law that it was under prior to candidacy. State incidental take permits will not be required for projects with the potential to take NC summer steelhead unless it is listed under CESA. Because NC steelhead are listed as threatened under ESA, projects with the potential to take NC summer and winter steelhead will be required to obtain federal and state permits currently in place pursuant to Section 4(d), 10(a), and 7(a)(2) of ESA and Sections 1002, 1002.5, and 1003 of the Fish and Game Code. The state will continue to negotiate Lake and Streambed Alteration Agreements, and comment on THPs, federal incidental take permits and recovery planning, and petitions and applications to the CASWRCB. Also, the Department will continue to act as the trustee agency for the state's fish, wildlife, and plant resources and responsible agency when it is in the role of issuing discretionary approvals within its regulatory authority for projects as defined under CEQA. In this role, the Department will review and comment on impacts to NC summer steelhead and recommend mitigation measures for these impacts as part of the CEQA review process.

If the Commission decides not to list NC summer steelhead, the Department will continue to participate in and support current or future programs designed to benefit NC summer steelhead and other anadromous fish. These programs include:

- Coordination with other agencies on the removal of Scott Dam if FERC issues a new license for the PVP that includes a plan for dam removal, coordination with state agencies to decrease impacts from timber related projects;
- Identifying and removing or retrofitting existing barriers to fish passage;
- Working with gravel extractors and other mining interests to avoid, minimize, or mitigate for impacts to fisheries resources;
- Continuing to restore and enhance salmon and steelhead habitat throughout the state through the Fisheries Restoration Grant Program and other granting programs;
- Participation in federal and state conservation and restoration programs operating in the petitioned area;
- Regulation of steelhead sport fishing in Northern California streams;
- Conducting research and monitoring programs; and
- Coordinating with other agency research and monitoring efforts.

11. RECOVERY CONSIDERATIONS

The Department's recovery objective for NC summer steelhead is to protect and expand existing natural-origin spawning populations and reestablish enough additional native populations in restored and protected streams to ensure persistence over a 100-year minimum time frame. Increased numbers, expanded distribution, and maintenance of genetic diversity will improve their probability of long-term survival within their native range. Recovery actions would focus on restoring, rehabilitating, and protecting habitat in natural holding, spawning, and juvenile rearing areas. Cold pool habitat with sufficient depth is particularly essential to supporting the summer steelhead life history in the adult holding phase.

Recovery would also require effective long-term status and trend monitoring of NC summer steelhead abundance and distribution throughout the petitioned area. Recovery goals must ensure that individual populations are sufficiently abundant to avoid genetic risks of small population size; therefore, these goals need to address abundance levels (adult spawning escapements), population stability criteria, distribution, and length of time for determining sustainability.

If NC summer steelhead is listed under CESA, the Department will develop appropriate downlisting or delisting criteria for NC summer steelhead, based on the best scientific information available. The Department will periodically reexamine the status of NC summer steelhead. When, in the Department's judgment, recovery goals and downlisting or delisting criteria have been met, the Department will make recommendations to the Commission regarding changing the status of this species.

Recovery of viable NC summer steelhead populations will require vigorous efforts by the Department, other government agencies, and the private sector to improve and expand habitat and support population expansion. Habitat conditions must be improved to provide the necessary spawning and rearing habitat to allow the NC summer steelhead populations to survive, diversify, and increase to levels sufficient to withstand droughts, unfavorable climatic and oceanic conditions, and other uncontrollable natural phenomena.

12. MANAGEMENT RECOMMENDATIONS

Although the Department recommends that NC summer steelhead not be listed under CESA, the summer steelhead ecotype represents an important diversity element that should be maintained within the NC steelhead DPS. The Department recommends the following measures be taken to effectively manage and preserve NC summer steelhead:

1. Implement comprehensive monitoring in all streams with extant NC summer steelhead populations and produce statistically robust population estimates.
2. Support and participate in the development of watershed specific efforts to effectively maintain and restore NC summer steelhead habitat by focusing on the combination of factors currently limiting the distribution and abundance of NC summer steelhead such as timber harvest, livestock grazing, gravel mining, and water extraction.
3. Improve and expand suitable and preferred habitat used by NC summer steelhead for summer holding, spawning, and juvenile rearing. Prioritize habitat restoration, protection, and enhancement in summer steelhead holding, spawning, and rearing areas. Habitat projects should focus on improving habitat complexity, riparian cover, instream wood, fish passage, and sediment transport, as well as enhancing essential deep, cold pool habitat for holding adults. Restoration should also be implemented in potential summer steelhead habitat not currently occupied by summer steelhead.
4. Continue research on GREB1L, ROCK1, and other relevant genomic regions to better understand the mechanism for run timing expression.
5. Investigate the role of resident *O. mykiss* as a potential reservoir for early migration alleles.
6. Investigate the effects of MRH winter steelhead on natural-origin steelhead populations.
7. Continue to operate MRH under its recently developed Hatchery and Genetic Management Plan.
8. Strengthen law enforcement in remote areas occupied by NC summer steelhead to reduce threats of poaching and illegal water diversions used for cannabis cultivation.
9. Evaluate current fishing regulations to determine any potential changes that could be implemented for further protection of NC summer steelhead.
10. Evaluate effects of the Eel River Potter Valley Hydroelectric Project flow operations and dam, water diversion, and fish ladder facilities on summer steelhead population viability. Support efforts to evaluate effects of Scott Dam removal on the upper Eel River. Eel River summer steelhead populations are important source populations for the early migration ecotype. Removal of Scott Dam provides potential for repopulation of summer steelhead in the upper Eel River basin.
11. Explore the potential for reintroduction of summer steelhead upstream of the location of Scott Dam.

12. Develop individual watershed management plans for watersheds with summer steelhead populations or the potential for reintroduction.
13. Investigate effects of hooking mortality and potential predation activity on NC summer steelhead.
14. Investigate potential impacts of water diversions on streamflow in NC summer steelhead streams and develop management criteria to reduce any adverse effects on summer steelhead.

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APPENDICES

Appendix A. Summary of Ford et al. (2020): Reviewing and Synthesizing the State of the Science Regarding Associations between Adult Run Timing and Specific Genotypes in Chinook Salmon and Steelhead: Report of a workshop held in Seattle, Washington, 27–28 February 2020

NOAA Fisheries' Northwest Fisheries Science Center convened a panel of fisheries geneticists in February 2020 to address the multitude of recently completed and active genetic studies on specific genomic associations with migration timing in salmonids. These studies have important conservation implications, especially in relation to CESA and ESA listings. The intent of the workshop was to discuss the current state of the science and identify areas of agreement, areas of uncertainty, conservation implications, and future research needs. The workshop was attended by federal, state, and academic geneticists and conservation planners. The proceedings became publicly available in June 2020 (Ford et al. 2020).

This appendix summarizes the main points presented in Ford et al. (2020) for reference in this California Endangered Species Act status review. Many of these points, which are summarized and reproduced below, refer to highly technical genetic and genomic research results and conclusions. Readers who require more information should refer to the original report referenced below.

Current State of Research

Summarizing the findings and recommendations presented in Ford et al. (2020), it is apparent that deconvoluting the genetic and genomic basis of run-timing is complex. It is generally accepted that run-timing phenotypic variation is strongly correlated with genetic sequence variation in a relatively small (~200 Kb) region of the GREB1L/ROCK1 region of chromosome 28 in Chinook Salmon and steelhead. Run-timing variation is also affected to a lesser degree by other genes and environmental factors.

There are two alleles in this region: an "early migrating" allele (E) and a "late migrating" allele (L). Fish with homozygous genotypes, EE and LL, exhibit early and late return timing, respectively. Heterozygotes (EL) generally exhibit an intermediate return timing, though, depending on the population, return can be skewed either early or late. The extent and importance of heterozygotes that possess both early and late arriving alleles is an active topic of debate. Results have been confounded by inconsistencies in sampling strategies between studies and effects due to habitat alterations over several decades.

It is unknown how genetic variation in the GREB1L/ROCK1 region actually causes variations in life history strategy – all of the studies to date have successfully established correlations, but not the actual biochemical pathways by which such variation functions in individual fish. Applying the current state of knowledge to conservation decisions is also a subject of debate.

There were a few areas of agreement and many areas of disagreement – the issue is far from settled. A key conservation point where participants were in agreement is that conservation units should continue to be defined by patterns of genetic diversity across the genome (e.g., microsatellite and SNP loci), not by variation in small genomic regions correlated with specific traits of interest, such as run-timing.

<u>**Areas of Agreement and Uncertainty**</u>

The following are verbatim points of agreement and uncertainty listed in Ford et al. (2020, 31–37). The authors note that they did not attempt to come to consensus on these points. Rather, these were statements generally agreed upon by the meeting participants. Readers should refer to the original report for expanded discussions of each point below.

Is the GREB1L/ROCK1 region responsible for adult migration timing, and if so by what mechanism?

Areas of agreement:

- A single region in the genome has a strong statistical association with adult run timing.
- The migration phenotype measured across prior studies is not standardized, and efforts should be made to do so.
- Marker development, validation, and standardization is extremely important.

Areas of uncertainty:

- The causal variant(s) for adult run timing remain to be identified.

What is the distribution of genetic variation for adult migration timing in space and time? Do the genes associated with migration timing have the same effect in populations inhabiting different environments and with different genetic backgrounds?

Areas of agreement:

- The GREB1L/ROCK1 association with run timing is best characterized in US West coastal populations for both Chinook salmon and steelhead, and to some degree in the Columbia River basin.

Areas of uncertainty:

- Our current understanding of both the contemporary and historical distribution of genetic variation in GREB1L/ROCK1, in association with run timing, is confounded by issues with phenotyping, influence of hatchery populations, and anthropogenic activities influencing access to habitat across space and time.

What is the pattern of dominance among haplotypes in the GREB1L/ROCK1 genomic region? What phenotype do heterozygotes express, and what is their fitness?

compared to homozygotes?

Areas of agreement:

- Heterozygotes are likely an important mechanism for the spread and maintenance of the early migration alleles over long time scales.

Areas of uncertainty:

- It may be too simplistic to focus on dominance of migration timing alone since genetic variation at the GREB1L/ROCK1 region also could influence other traits that are more difficult to study.

In what circumstances is it reasonable to conclude that the current distribution of GREB1L genes accurately reflects historical (pre-European contact) patterns? When/where is that not a good assumption?

Areas of agreement:

- Interaction between individuals with variable run timing has occurred historically, is expected, and likely varies depending on historical environmental conditions. However, anthropogenic impacts have also likely changed these interactions in many locations.

Areas of uncertainty:

- It is unclear how much demographic isolation from fall run is required for spring Chinook salmon to persist.

How common are large-effect genes? Is it likely that strong associations will be found between specific alleles and many other phenotypic/life-history traits in salmon?

Areas of agreement:

- Loci of large effect have been identified for other salmonid life-history traits.

Areas of uncertainty:

- More data are needed from whole genome sequencing to know the extent to which complex traits are controlled by single genes of large effect, or many loci of smaller effect and how this various among populations.

Prince et al. (2017) concluded that the haplotypes associated with early migration timing evolved only once within each species. Is that the case, or are the genetic variants more evolutionarily labile?

Areas of agreement:

- The evolutionary history of the GREB1L/ROCK1 region is complex and has not been well characterized throughout each species' entire range. But it is clear that the early and late haplotypes that have been well characterized evolved long ago in each species' evolutionary history. It is also clear, based on available data, that the allelic variants for early migration have not arisen independently via new mutations from the genomic background of late migration individuals in each watershed.

Needed Future Research

The participants outlined areas for future research, which are directly excerpted below (Ford et al. 2020, 37–38):

1. Better standardization and characterization of adult migration phenotypes in multiple populations and lineages, including when the 'decision' to migrate is made, how it relates to the timing of sexual maturity and the relationship(s) between the date of freshwater entry and subsequent upstream movements.
2. More thorough marker development and validation (see next section). Ideally, identification of the functional variant(s) in the GREB1L/ROCK1 region that cause alternative migration phenotypes.
3. Greater understanding of the physiological mechanisms leading to alternative migration phenotypes.
4. Tests for association of GREB1L/ROCK1 variation on phenotypes other than adult run timing, such as timing of sexual maturity or other life-history traits.
5. More thorough evaluations of the genetics of run timing variation, throughout the geographic range of Chinook salmon and steelhead, as well as studies in other salmon species in order to develop broad baseline data on the historical and current distribution of alleles at this locus. Current studies have been primarily focused on a limited number of West Coast and Columbia River populations. These investigations should include characterization of the full suite of genetic variants (and their effect sizes) contributing to run timing,
6. More thorough characterization of GRE1L/ROCK1 haplotype diversity and the phenotype and dominance pattern of each identified haplotype in multiple populations of both species, across their range.
7. Perform comparative analyses on systems with early-run and late-run populations that have been differentially impacted by human activities resulting in differing levels of interbreeding between life-history types, to determine how interbreeding might affect persistence of run type alleles.

Conservation Implications

Subsequent to the technical discussions, the participants discussed how the current state of knowledge should be applied to conservation decisions such as defining units for conservation, listing, and recovery. Their individual points are excerpted directly and presented here (Ford et al. 2020, 38–42):

Areas of agreement:

1. After discussion on whether conservation strategies might need to change based on the GREB1L/ROCK1 findings, the participants generally agreed that using patterns of genetic variation throughout the genome remains important for identifying conservation units, rather than identifying units based solely on small genomic regions associated with specific traits.
2. The workshop participants agreed that spring Chinook salmon and summer steelhead occupy a specialized ecological niche—upstream areas accessible primarily during spring flow events—that has made them particularly vulnerable to extirpation or decline due to habitat degradation.
3. The participants generally agreed that the evaluation of risk to early returning population groups (spring Chinook, summer steelhead) needs to consider what we now know about the genetic basis of adult return time.
4. The participants generally agreed that the finding that the early run trait has a simple genetic basis implies that it is at greater risk of loss than if it were highly polygenic because loss of the "early" allele(s) equates to the loss of the phenotype.

Areas of uncertainty:

1. One area of uncertainty and potential disagreement at the workshop was the degree to which run timing diversity in spring Chinook salmon is partitioned among populations versus among individuals within a population.
2. The extent to which observed contemporary levels of interbreeding between individuals with early and late run timing would be typical under historical environmental conditions is unknown
3. Understanding the conservation implications of dominance patterns at the GREB1L/ROCK region is also important and is complicated because of tradeoffs between the probability of persistence of the early-run allele and the feasibility of starting new early-run populations.
4. The dominance-recessive relationships might influence the success of colonization events.

5. Regardless to what extent current levels of interbreeding are a consequence of human mediated habitat alterations, such interbreeding and the common occurrence of heterozygotes at the GREB1L/ROCK1 region presents challenges for status monitoring, recovery planning, and other management actions.

6. Improved strategies are needed for monitoring run timing and associated genetic variation.

7. What conservation measures can be put into place now with existing knowledge? Conservation measures for spring run that were discussed included potentially shaping fisheries to focus disproportionately on fish with fall run timing, restoring access to spring-run habitat that has been blocked, considering restoring natural barriers that have been modified to increase fall-run access to historically spring-run habitats, and restoring more natural flow regimes (e.g., low summer flows that prevent mature migrating individuals from encroaching on premature habitat). Workshop participants agreed that the presence of heterozygotes does not in itself indicate a threat to the viability of spring run as these heterozygotes contain alleles that may be important to spring-run restoration. Some workshop participants also noted, however, that in some cases the presence of high proportions of heterozygotes might represent a departure from the historical conditions and a warning sign that the spring-run phenotype is at risk.

Issues specifically associated with steelhead

1. One major factor to consider regarding the conservation implications of the genetics of run timing diversity in steelhead is the existence of conspecific resident rainbow trout populations that may effectively act as reservoirs for the ‘early’ GREB1L/ROCK1 alleles.

2. Another factor to consider for steelhead compared to Chinook is the generally greater amount of life-history diversity found in O. mykiss.

Appendix B. Annual summer steelhead counts and data sources for five extant populations

Year	Redwood Creek[a]	Mad River[b]	Middle Fork Eel River[c]	Van Duzen River[d]	Mattole River[e]
1966	-	-	198	-	-
1967	-	-	241	-	-
1968	-	-	335	-	-
1969	-	-	-	-	-
1970	-	-	865	-	-
1971	-	-	997	-	-
1972	-	-	502	-	-
1973	-	-	1,422	-	-
1974	-	-	1,522	-	-
1975	-	-	1,149	-	-
1976	-	-	792	-	-
1977	-	-	654	-	-
1978	-	-	377	-	-
1979	-	-	1,298	31	-
1980	-	10	1,052	25	44
1981	16	6	1,601	6	-
1982	2	181	1,054	8	-
1983	7	37	666	13	-
1984	44 [f]	111	1,524	58	-
1985	44 [f]	52	1,490	-	-
1986	19 [f]	5	1,000	-	-
1987	15 [f]	18	1,550	52	-
1988	8	60	711	40	-
1989	0	20	727	4	-
1990	14 [f]	33	449	-	-
1991	15 [f]	66	691	31	-
1992	7	34	516	0 [g]	-
1993	8	48	622	-	-
1994	36	305	701	-	-
1995	18	569	1,148	2 [g]	-
1996	23	515	771	-	14
1997	37	297	513	4 [g]	16
1998	25	191	527	-	44
1999	10	85	451	4	16
2000	3	170	306	-	17
2001	1	194	422	-	17
2002	3	185	418	-	15
2003	6	483	657	80	9
2004	16	209	731	26 [g]	16

Year	Redwood Creek[a]	Mad River[b]	Middle Fork Eel River[c]	Van Duzen River[d]	Mattole River[e]
2005	22	211	626	40	20
2006	21	-	817	50	19
2007	13	-	534	100	16
2008	59	110	721	122	36
2009	9	-	396	65	33
2010	7	-	523	-	14
2011	8	-	837	110	18
2012	8	-	1,191	255	22
2013	15	282	872	162	56
2014	15	322	702	81	55
2015	5	336	445	54	19
2016	12	187	493	140	14
2017	2	151	323	86	14
2018	3	147	330	144	29
2019	3	117	434	67	7

"-" indicates no survey conducted.

[a] Source: National Park Service (2019 Redwood Creek Summer Steelhead Survey, pre-print report)

[b] Source: Data collated from USFS, GDRCO, CalTrout, and CDFW original survey reports.

[c] Source: CDFW internal data (S. Harris, CDFW, pers. comm., January 2, 2020)

[d] Source: CDFW internal data (S. Thompson, CDFW, pers. comm., March 23, 2020)

[e] Source: Mattole Salmon Group (N. Queener, Mattole Salmon Group, pers. comm., February 4, 2020)

[f] Some fish less than 16 inches included.

[g] Half pounders removed.

Appendix C. Numbers of summer steelhead observed, survey start and end points, and survey dates for Redwood Creek annual surveys conducted 1981 – 2019.

Original Index Reach: Lacks Creek to Tom McDonald Creek. (M. Sparkman, CDFW, pers. comm., January 30, 2020 [original file from D. Anderson, Redwood National Park])

Year	Index Reach: Number of summer steelhead	Total Reach: Number of summer steelhead	Total Reach Surveyed: Start to End Points and Distance (km)	Survey Dates
1981	16	16	Beaver Cr to Orick (51.5)	10 – 13 Aug
1982[a]	2	2	Stover Cr to Emerald Cr (22.5)	12, 14 Oct
1983	5	7	HWY 299 to Tom McDonald Cr (44.4)	22 – 25 Aug
1984	44+	44+	Index Reach (25.9)	8 – 10 Aug
1985	44+	44+	Index Reach (25.9)	20 – 22 Aug, 4 Sep
1986	19	19	Index Reach (25.9)	25 – 27 Aug
1987	14	15	1 mile downstream of Snow Camp Cr to Tom McDonald Cr (72.7)	14 – 16 Jul
1988	8	8 (6 spring Chinook Salmon)	Index Reach (25.9)	26 – 28 Jul
1989[b]	0	0	Lacks Cr to Bridge Cr (17.9)	31 Jul, 1 – 2 Aug
1990	14	14	Index Reach (25.9)	31 Jul, 1 – 3 Aug
1991	15	15	Index Reach (25.9)	5 – 8 Aug
1992	4 live, 1 dead	6 live, 1 dead	Lacks Cr to Hayes Cr (37.8)	3 – 6, 10 Aug
1993	2	3	Lacks Cr to Hayes Cr (37.8)	2 – 5, 9 Aug
		2[c]	Chezem Rd to Stover Cr (24.75)	18, 20, 27 Aug
		3[d]	Ayres Cabin to Chezem Rd (14.3)	28, 30 Aug
		Total: 8	76.9 km	

Year	Index Reach: Number of summer steelhead	Total Reach: Number of summer steelhead	Total Reach Surveyed: Start to End Points and Distance (km)	Survey Dates
1994	5	8	Lacks Cr to Hayes Cr (37.8)	1 – 4 Aug
		16[c]	Chezem Rd to Stover Cr (24.8)	4 – 6 Aug
		3[e]	HWY299 to Chezem Rd (2.0)	15 Aug
		9[d]	Bradford Cr to HWY 299 (20.1)	13 – 14 Aug
		Total: 36	84.7 km	
1995	5 live, 1 dead	7 live, 1 dead	Lacks Cr to Hayes Cr (37.8)	24 – 27 Jul
		8[c]	Chezem Rd to Bair Rd (7.6)	28 Aug
		2[d]	Bradford Cr to Ayres Cabin (7.9)	2 Sep
		Total: 18	53.25 km	
1996	1	1	Lacks Cr to Hayes Cr (37.8)	5 – 8 Aug
		21[c]	Chezem Rd to Stover Cr (24.8)	31 Jul, 1, 22 Aug
		1[d]	Bradford Cr to Chezem Rd (22.1)	10 – 12 Aug
		Total: 23	84.7 km	
1997	6	6	Lacks Cr to Hayes Cr (37.8)	4 – 7 Aug
		16[c]	Chezem Rd to Stover Cr (24.8)	6 – 8 Aug
		15[d]	Bradford Cr to Chezem Rd (22.1)	17, 24, 30 Aug
		Total: 37	84.7 km	
1998	4	4	Lacks Cr to Hayes Cr (37.8)	27 – 30 Aug
		21[d]	Bradford Cr to Stover Cr (46.9)	24 – 27, 31 Aug
		Total: 25	84.7 km	
1999	5	10	Lacks Cr to Hayes Cr (38)	2 – 5, 9 – 10 Aug
2000	3	3	Lacks Cr to Hayes Cr (38)	1 – 3, 7 – 9 Aug

Year	Index Reach: Number of summer steelhead	Total Reach: Number of summer steelhead	Total Reach Surveyed: Start to End Points and Distance (km)	Survey Dates
2001	0	1	Lacks Cr to Hayes Cr (38.1)	31 Jul, 1 – 2, 7 Aug
2002	3	3	Lacks Cr to Hayes Cr (38.1)	29 – 31 Jul, 1, 5, 7 Aug
2003	4	6	Lacks Cr to Downstream of Hayes Cr (38.8)	29 – 31 Jul, 4 – 6 Aug
2004	8 live, 1 dead	15 live, 1 dead	Lacks Cr to Downstream of Hayes Cr (39.0)	27 – 29 Jul, 3 – 5 Aug
2005	19	22 (1 Sockeye Salmon)	Lacks Cr to Downstream of Hayes Cr (38.9)	26 – 28 Jul, 3 – 5 Aug
2006	19 live, 1 dead	20 live, 1 dead	Lacks Cr to Downstream of Hayes Cr (38.9)	25 – 26, 31 Jul, 1 – 2 Aug
2007	11	13 (1 Sockeye Salmon)	Lacks Cr to Downstream of Hayes Cr (38.9)	24 – 26, 30 – 31 Jul, 1 Aug
2008	19	21	Lacks to Downstream of Hayes (38.9)	28 – 31 Jul, 4 – 6 Aug
		9[f]	Chezem Rd Bridge to Bair Rd Bridge (7.65)	28 Jul
		6[f]	Bair Rd Bridge to Beaver Cr (7.4)	29 Jul
		23[f]	Beaver Cr to Stover Cr (8.8)	30 Jul
		Total: 59	62.75 km	
2009	7	9	Lacks Cr to Downstream of Hayes Cr (38.9)	27 – 30 Jul, 3 – 5 Aug
2010	6	7	Lacks Cr to Downstream of Hayes Cr (38.9)	26 – 29 Jul, 2 – 4 Aug
2011	8	8	Lacks Cr to Downstream of Hayes Cr (38.9)	26, 27 Jul, 1 – 4, 8 Aug
2012	7 live, 1 dead	7 live, 1 dead	Lacks Cr to Downstream of Hayes Cr (38.9)	23 – 26, 30 – 31 Jul, 1 Aug
2013	5	15	Lacks Cr to Downstream of Hayes Cr (38.9)	29 – 31 Jul, 1, 5 – 7 Aug
2014	13	15	Lacks Cr to Downstream of Hayes Cr (38.9)	28 – 31 Jul, 4 – 6 Aug
2015	4	5	Lacks Cr to Downstream of Hayes Cr (38.9)	28 – 30 Jul, 3 – 5 Aug
2016	8	12	Lacks Cr to Downstream of Hayes Cr (41.6)[g]	25 – 28 Jul, 1 – 3 Aug
2017	1[g]	2	Lacks Cr to Downstream of Coyote Cr and Bridge Cr to Downstream of Hayes Cr (28.6)[h]	24, 31 Jul, 1 – 2, 16 Aug

Year	Index Reach: Number of summer steelhead	Total Reach: Number of summer steelhead	Total Reach Surveyed: Start to End Points and Distance (km)	Survey Dates
2018	2	3	Lacks Cr to Downstream of Hayes Cr (41.6)[g]	23 – 26, 30 – 31 Jul, 1 Aug
2019	3	3	Lacks Cr to Downstream of Hayes Cr (41.6)[g]	22 – 25, 29 – 31 Jul

[a] Survey from Stover Creek to Emerald Creek, 14 miles, covering most of index section and best pool habitat.

[b] Survey from Lacks to Bridge Creek, minus Garret to Panther Creek, a total of 11.1 miles. Covered best pool habitat.

[c] Tom Weseloh – California Trout

[d] Matt Smith – North Coast Fisheries Restoration

[e] Kirk Cohune

[f] Tom Weseloh – California Trout; Tom Shaw – USFWS-Arcata Office

[g] 2016 – 2019: total distance derived from LiDAR data to route creek and determine distances. Higher resolution LiDAR data changed distance values but the starting and ending endpoints remain the same.

[h] Incomplete Survey: 12.8 km reach between downstream of Coyote Creek and Bridge Creek not snorkeled. 69% of normal reach surveyed.

Appendix D. Mattole Salmon Group Summer Steelhead Data

(N. Queener, MSG, pers. comm., February 4, 2020)

Year	Adults (>16")	Half-pounders (12"-16")	Miles Surveyed	Mainstem Miles	Trib. Miles	Adults/ Mile	Half-pounders/ Mile
1996	14	36	25.3	23.6	1.7	0.55	1.42
1997	16	19	39.3	38	1.3	0.41	0.48
1998	44	85	44.9	44.6	0.3	0.98	1.89
1999	16	88	39.3	37.4	1.9	0.41	2.24
2000	17	96	32.55	32.4	0.15	0.52	2.95
2001	17	40	31.35	31.2	0.15	0.54	1.28
2002	15	22	29.45	29.3	0.15	0.51	0.75
2003	9	21	46.25	40	6.25	0.19	0.45
2004	16	44	46.75	40.5	6.25	0.34	0.94
2005	20	34	60.85	54.6	6.25	0.33	0.56
2006	19	38	64.85	58.6	6.25	0.29	0.59
2007	16	79	65.55	59.3	6.25	0.24	1.21
2008	36	73	65.55	59.3	6.25	0.55	1.11
2009	33	49	65.55	59.3	6.25	0.50	0.75
2010	14	43	67.25	60.4	6.85	0.21	0.64
2011	18	37	70.95	64.3	6.65	0.25	0.52
2012	22	38	66.65	60.4	6.25	0.33	0.57
2013	56	54	66.1	60	6.1	0.85	0.82
2014	55	33	58.4	55.9	2.5	0.94	0.57
2015	19	38	65.9	59.7	6.2	0.29	0.58
2016	14	36	65.9	59.7	6.2	0.21	0.55
2017	14	44	65.9	59.7	6.2	0.21	0.67
2018	29	45	60.5	55	5.5	0.48	0.74
2019	7	36	57.1	54.5	2.6	0.12	0.63