State of the Salmonids:
Status of California's Emblematic Fishes 2017

A report commissioned by California Trout

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We apologize to anyone whom we may have inadvertently failed to list here but had a hand in the generation of this report. Any errors or omissions are the responsibility of the authors.

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ABSTRACT

California has, or had, 32 distinct kinds of salmonid fishes. They are either endemic to California or at the southern end of their ranges. Most are in serious decline: 45% and 74% of all salmonids will likely become extirpated from California in the next 50 and 100 years, respectively, if present trends continue. Our results suggest that California will lose more than half (52%) of its native anadromous salmonids and nearly a third (30%) of its inland taxa in just 50 years under current conditions. Climate change is a major overarching threat driving population declines throughout California and strongly affects the status of 84% of all salmonids reviewed. In addition, dams, agricultural operations, estuary alteration, non-native species, production hatcheries, and myriad other human-induced threats have contributed to declines. 81% of salmonids in California are now worse off than they were in 2007, when the previous version of this report was prepared. The changes in species status are the result of the 2012-2016 historic drought, improved data collection and review, and an improved understanding of climate change impacts. Returning these iconic species to sustainable levels requires access to productive and diverse habitats which promote the full range of life history diversity necessary to weather change. We recommend (i) protecting and investing in fully functioning watersheds such as the Smith River and Blue Creek, (ii) protecting and restoring source waters such as Sierra meadow systems, groundwater, and springs so that the impacts of climate change are reduced, (iii) restoring function and access to once productive and diverse habitats such as Central Valley floodplains, coastal lagoons, and estuaries, (iv) adopting reconciliation ecology as a basis for management in human dominated landscapes, (v) improving habitat connectivity and passage to historical spawning and rearing habitat, and (vi) improving salmonid genetic management throughout California.

INTRODUCTION

Salmon, trout, and their relatives are the iconic fishes of the Northern Hemisphere. They are adapted for life in dynamic landscapes created by glaciers, volcanoes, earthquakes, and climatic extremes. Salmonids thrive through their mobility, moving freely between ocean and river systems; they show an extraordinary ability to adapt in isolation to extreme local conditions from deserts to rain forests. This has resulted in a handful of species producing hundreds of genetically distinct runs, races, and subspecies, all with life histories superbly tuned to local habitats (Behnke 2002, Moyle 2002).

Despite their adaptability, ease of culture, and economic importance, salmonid fishes are in severe decline in many of their native habitats; many populations have been extirpated (Montgomery 2003). The reasons for this are complex and multiple, but boil down to a combination of human competition for high quality water, alteration of the landscape through which rivers and streams flow, overfishing, use of production hatcheries to maintain fisheries, and introductions of non-native species as predators or competitors. Habitat restoration, especially restoration of flows to degraded rivers, has generally been a low priority.

Nowhere in the world is the diversity of salmonids and their problems more evident than in California. The state not only marks the southern end of the range of all anadromous species on the Pacific Coast, but its dynamic geology and climate have resulted in evolution of many distinctive inland forms, such as three kinds of golden trout in the southern Sierra Nevada. The diversity of salmonids (Figure 1) is also the result of California's large size (411,000 km^2), length (spanning 10° of latitude), and being adjacent to the California Current of the Pacific Ocean, one of the most productive ocean regions of the world (Moyle 2002).

This has resulted in the evolution of many genetically distinct populations, although there are just eight recognized native species. We recognize 32 distinct salmonids in California, 21 of them anadromous, 11 of them non-anadromous. Of these salmonids, 20 are endemic to California and only five are shared with neighboring states. These fishes can all be recognized as species for management purposes under definitions in the federal Endangered Species Act of 1973. However, they are classified as species, subspecies, Evolutionary Significant Units (ESUs), and Distinct Population Segments (DPSs).

Many (15, 47%) of California's salmonids are already recognized as threatened, endangered, or extinct by state and federal governments, but the only focused status reviews of all salmonids in the state have been those produced by the collaboration of the Center for Watershed Sciences at UC Davis and California Trout (Moyle et al. 2008, Katz et al. 2012). We undertook these status reviews because California salmonids are (i) culturally and economically important, (ii) characteristic of most of California's inland and coastal waters and serve as umbrella species for conservation of a much broader aquatic community, (iii) are exceptionally vulnerable to climate change, (iv) are not monitored as closely as they should be, especially those not listed under state and federal Endangered Species Acts (ESAs), and (v) are in a general state of decline.

Our goal of this report is to follow the 2008 previous report in establishing baseline evaluations to enable repeatable, systematic comparisons of species status over the years, in response to climate change and other factors. It has also been our perception (and still is) that lists of threatened and endangered species do not reflect the true condition of salmonids in California both because species that are not listed are in decline and because some species may actually recover under intensive management (Figure 1). For example, in 2008, the commercial salmon fishery was closed due to low adult returns. Each year since then, the fishery has been restricted or constrained in some way. At the writing of this report, the commercial salmon fishery is again closed across much of the state for the 2017 season, reflecting serious decline despite massive production of juveniles by hatcheries. Our report dives deeply into the status and trends facing all salmonids in California at the southern edge of their range, and does so at a time when the state is emerging from historic drought (2012-2016). While California’s climate, precipitation patterns, and trends in salmonids have changed, monitoring efforts, especially for species not listed under the Endangered Species Act, have remained insufficient.

In the first edition of *State of the Salmonids* (2008), the over-arching questions addressed were: What is the status of all California salmonids, both individually and collectively? What are major factors responsible for present status, especially of declining species? How can California’s salmonids be saved from extinction? A little less than 10 years later, we are still asking the same questions. This report, however, reflects improvements in our evaluation methods, including evaluating the effects of climate change, which provides a clearer picture of status and should be easier to repeat in future evaluations.

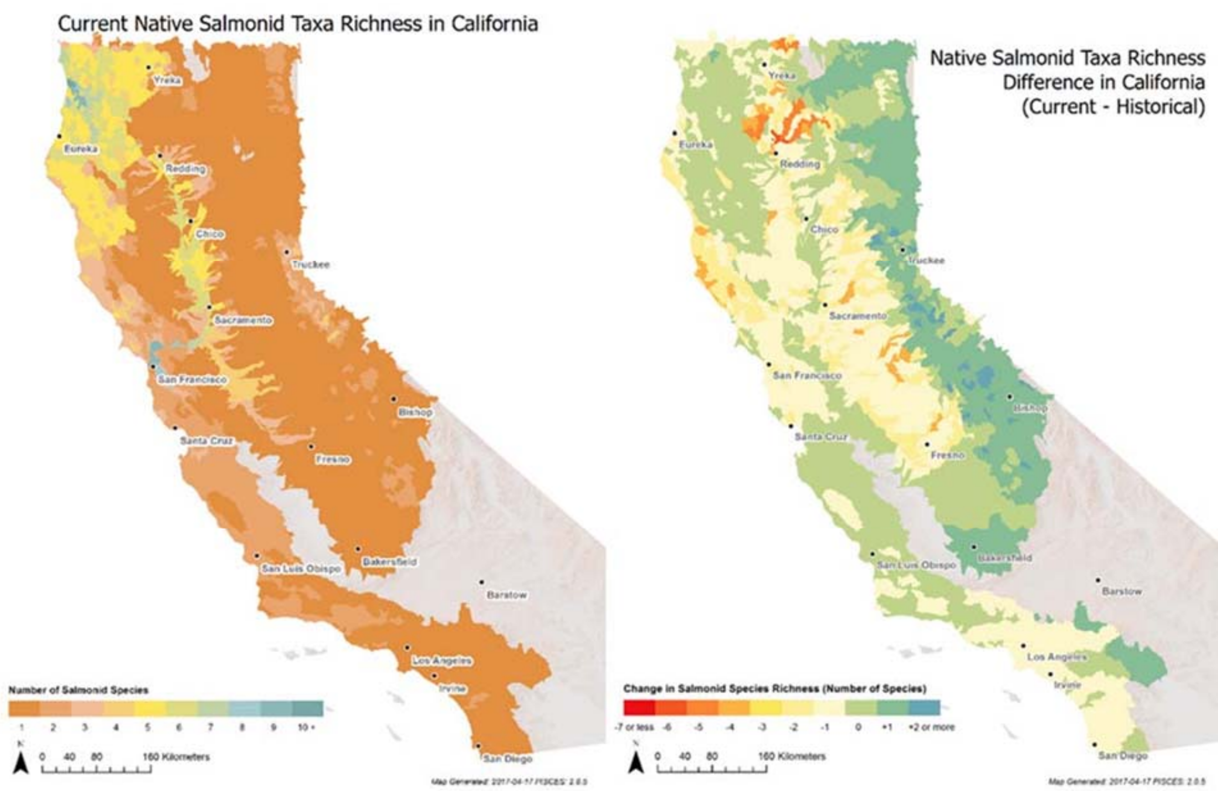


Figure 1. Current native salmonid richness in California (left) and native salmonid change in richness, pre-1971 to present (right).

METHODS

Sources of information

The subject of this report is salmonids native to California. It does not include introduced species such as brown trout (*Salmo trutta*), brook trout (*Salvelinus fontinalis*), lake trout (*Salvelinus namaycush*), kokanee salmon (*Oncorhynchus nerka*), or Colorado cutthroat trout (*Oncorhynchus clarkii pleuriticus*). To make the accounts as accurate and up-to-date as possible, we reviewed the recent literature, including published scientific reports and agency 'gray literature' documents as well as various other references. We also relied on our personal knowledge of each species and conducted interviews with dozens of species experts. Each account was externally reviewed by at least one expert, but often by multiple experts.

This report contains reviews of the biology and status of 32 kinds of salmonids in California, all written in a standard format (Table 1).

Table 1. Standard format of fish species accounts.

I. Status summary
Status category (Table 2) with a brief summary of status.

II. Description
Distinguishing features, beyond what is in Moyle (2002).

III. Taxonomic relationships
Brief history of taxonomy and systematics, with special emphasis on recent genetic studies.

IV. Life history
Synthesis of known information pertaining to life history.

V. Habitat requirements
Habitat description for major life history stages including basic physiological tolerances (temperature ranges etc.), if known.

VI. Distribution
Present and historic range of the species.

VII. Trends in abundance
An assessment of both long and short-term trends, using quantitative data where possible, but otherwise using expert opinion or independent trend assessments.

VIII. Factors Affecting Status
A catalog of threats, including a standardized table of anthropogenic factors limiting populations.

IX. Effects of climate change
An evaluation of effects of climate change on status in the next 100 years.

X. Status determination
An evaluation of status based on seven metrics along with a certainty evaluation and status ratings from other sources.

XI. Management recommendations
Ongoing management actions, as well as management recommendations.

Scoring protocol

The status for each salmonid was assessed with a standardized protocol, as developed in Moyle et al. (2015). Our goal was to make status determinations as transparent and repeatable as possible. The final status score was based on the average of seven metrics. Each metric was standardized on a 1-5 scale, where '1' was low (negative effect on status) and '5' was high (no or positive effect) and '2', '3' and '4' were intermediate values. The metrics were designed to cover all major factors affecting salmonid status in California, with minimal redundancy among metrics (Table 2). Metrics were not weighted, so each had the same impact on status ratings. However, a statistical analysis using similar scores for the entire native inland fish fauna of California indicated that no one metric dominated the final status scores (Moyle et al. 2011).

The assignment of criteria for the five possible scores within each of the seven metrics was based on literature reviews and personal knowledge, with some modifications since first published in Moyle et al. (2011). Two of the metrics, vulnerability to climate change and anthropogenic threats, required additional analysis before scoring took place. Climate change was initially scored, using eight metrics, in Moyle et al. (2013). Those climate change scores were used in the status review for each species presented here. The anthropogenic threats composite score was based on a detailed analysis of 15 factors outlined in the next section.

Table 2. Rubric used to assign scores to seven metrics developed to assess status of native salmonids in California. Final status score is the average of all seven metric scores. Each metric is scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. A certainty score of 1-4 was used to assign relative weights to the level of information available to make determinations, with 4 being the most reliable and 1 being the least reliable information (see Table 5).

Metric	Score	Justification
Area occupied	1-5	
Estimated adult abundance	1-5	
Intervention dependence	1-5	
Tolerance	1-5	
Genetic risk	1-5	
Anthropogenic threats	1-5	
Climate change	1-5	
Average	X.X	XX/7.
Certainty (1-4)	1-4	

Metrics

1A. Area occupied: resident salmonids

1. 1 watershed/stream system in California only, based on watershed designations in Moyle and Marchetti (2006)
2. 2-3 watersheds/stream systems without fluvial connections to each other
3. 3-5 watersheds/stream systems with or without fluvial connections
4. 6-10 watersheds/stream systems
5. More than 10 watersheds/stream systems

1B. Area occupied: anadromous salmonids

1. 0-1 apparent self-sustaining populations

2. 2-4 apparent self-sustaining populations
3. 5-7 apparent self-sustaining populations
4. 8-10 apparent self-sustaining populations
5. More than 10 apparent self-sustaining populations

2. Estimated adult abundance

1. ≤ 500
2. 501-5000
3. 5001-50,000
4. 50,001-500,000
5. 500,000 +

3. Dependence on human intervention for persistence

1. Captive broodstock program or similar extreme measures required to prevent extinction
2. Continuous active management of habitats (e.g., water addition to streams, establishment of refuge populations, hatchery propagation or similar measures) required
3. Frequent (usually annual) management actions (e.g., management of barriers, special flows, removal of non-native species)
4. Long-term habitat protection or improvements (e.g., habitat restoration) but no immediate threats need to be addressed
5. Self-sustaining populations that require minimal intervention

4. Environmental tolerance under natural conditions

1. Extremely narrow physiological tolerance in all habitats
2. Narrow physiological tolerance to conditions in all existing habitats or broad physiological limits but species may exist at extreme edge of tolerances
3. Moderate physiological tolerance in all existing habitats
4. Broad physiological tolerance under most conditions likely to be encountered
5. Physiological tolerance rarely an issue for persistence

5. Genetic risks

1. Fragmentation, genetic drift and isolation by distance, owing to very low levels of migration, and/or frequent hybridization with related fishes are the major forces reducing genetic viability
2. As above but limited gene flow among populations; hybridization can be a threat
3. Moderately diverse genetically, some gene flow among populations; hybridization risks low but present
4. Genetically diverse but limited gene flow to other populations, often due to recent reductions in habitat connectivity
5. Genetically diverse with gene flow to other populations (metapopulation structure)

6. **Vulnerability to climate change** (score from Moyle et al. 2013)

1. Vulnerable to extinction in all watersheds inhabited (< 17)
2. Vulnerable in most watersheds inhabited (17-22)
3. Vulnerable in portions of watersheds inhabited (23-27)
4. Low vulnerability due to location, cold water sources and/or active management (28-32)
5. Not vulnerable, most habitats will remain within tolerance ranges (> 32)

7. Anthropogenic threats (see Table 4)

1. 1 or more threats rated critical or 3 or more threats rated high - indicating species could be pushed to extinction by one or more threats in the immediate future (within 10 years or 3 generations)

2. 1 or 2 threats rated high - species could be pushed to extinction in the foreseeable future (within 50 years or 10 generations)
3. No high threats but 5 or more threats rated medium - no single threat likely to cause extinction but all threats, in aggregate, could push species to extinction in the foreseeable future (within the next century)
4. 2-4 threats rated medium - no immediate extinction risk but, taken in aggregate, threats reduce population viability
5. 1 medium all others low - known threats do not imperil the species

Averaging the seven metrics resulted in a final status score ranging from 1.0 to 5.0 (Table 3). Climate change was initially scored, using eight metrics, in Moyle et al. (2013). Those scores for individual species were used for each species status score presented here. Salmonids that scored between 1.0 and 1.9 were labeled Critical Concern and were regarded as being in serious danger of extinction in their native range. Species with scores between 2.0 and 2.9 were labeled High Concern and were considered to be under severe threat but extinction was less imminent than for species with lower scores. However, these species could easily drop into the first category if current trends continue. Species that scored 3.0 - 3.9 were considered to be under no immediate threat of extinction but were in long-term decline or had naturally small, isolated populations; they were labeled as Moderate Concern. Salmonids scoring 4.0-5.0 were regarded as Low Concern. The scores only apply to salmonids that spawn in California, so those with a wide distribution and abundance outside the state (e.g., coastal cutthroat trout) could receive low scores within the state, reflecting California's position at the edge of their range.

Table 3. Status categories, score ranges, and definitions for California salmonids.

Status	Scores	Definition
Extinct	0.0	Extirpated from inland waters of California
Critical Concern	1.0 - 1.9	High risk of extinction in the wild; abundance critically low or declining; current threats projected to push species to extinction in the wild in 10-15 generations.
High Concern	2.0 - 2.9	High risk of becoming a critical concern species; range and abundance significantly reduced; trajectory to extinction in 15-20 generations if no actions taken.
Moderate Concern	3.0 - 3.9	Declining, fragmented and/or small populations possibly subject to rapid status change; management actions needed to prevent increased conservation concern
Low Concern	4.0 - 5.0	Populations not in decline; abundant and widespread

Metric rationale

1. Area occupied

This metric is based on the known current distribution of each salmonid, as described in each account and based in part on information stored in the UC Davis PISCES data base (https://pisces.ucdavis.edu/). Anadromous and non-anadromous salmonids had different metrics because salmon and steelhead can move among watersheds more readily than resident trout and whitefish. This metric provides an estimate of how many distinct populations there were of each salmonid, based on geography.

2. Estimated adult abundance

Our goal here was to estimate the typical number of reproductive adults in the population. Given that solid data for this metric was generally lacking, the numbers are order of magnitude estimates, with each category 10X larger than the previous one.

3. Dependence on human intervention for persistence

All California salmonids rely to some extent on human actions for their existence. This metric is an attempt to estimate how quickly a salmonid would disappear from California altogether without intervention. The lowest score was given to fishes such as Sacramento River winter-run Chinook salmon, which would be quickly extirpated without a hatchery program and artificial maintenance of habitat below Keswick Dam. Higher scores go to salmonids that require continuous habitat improvements or barrier maintenance to keep them going in their native range, such as California golden trout. The actual score depended on our professional judgment.

4. Environmental tolerance under natural conditions

This metric is based on the known or likely physiological tolerances to extremes in environmental variables such as temperature, dissolved oxygen, and turbidity. The score was usually extrapolated from observations of salmonids in their natural environment because laboratory studies are lacking on many salmonids, especially for California salmonids, which are at the extreme southern end of their ranges. Our scoring takes into account that salmonids in general require high water quality for persistence but, some are more tolerant of poor water quality than others and/or are more likely to encounter it at some point (e.g., steelhead vs. coho salmon).

5. Genetic risks

Extensive genetic/genomic studies of most California salmonids are lacking but risks of inbreeding and outbreeding depression, bottlenecks, and hybridization are very real. We estimated these risks based on existing genetic studies, known population sizes, likely effects of fish of hatchery origin, and other factors. The more likely a salmonid was to have, or be on a trajectory towards, low genetic diversity, the lower the score. Low genetic diversity can increase probability of extinction in a changing environment.

6. Vulnerability to climate change

Climate change (global warming) is an increasing problem for salmonids because they are basically coldwater fishes living in a waterscape that is becoming warmer, with more variable flows in streams. Our scoring here largely reflects the 8-metric vulnerability scoring of Moyle et al. (2013) for the entire California inland fish fauna. However, information on the effects of climate change on watersheds and habitats used by each salmonids was also taken into account.

7. Anthropogenic threats

For each salmonid, 15 anthropogenic factors that limit, or potentially limit, viability were rated (listed below). Because of the subjective nature of these ratings, all authors had to agree with each rating. The ratings of these factors were then combined to create a single evaluation variable. Factors were rated on a five-level ordinal scale (Table 5), where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years, whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result; and a factor rated "n/a" has no known negative impact to the taxon under consideration. Descriptions of most of these factors, with access to literature on which they are based, can be found in Moyle (2002) and in the detailed accounts of each species in this document.

Anthropogenic threats

Major dams. Dams were recorded as having a high impact on a salmonid if they prevented access to a large amount of its range, caused major changes to habitats, significantly changed downstream water quality and or quantity, or isolated fish in streams above dams. The effects and impacts of reservoirs created by dams were also included.

Agriculture. Impacts from agriculture were regarded as high if agricultural return water or farm effluent heavily polluted streams, agricultural diversions severely reduced flow or affected migratory patterns, large amounts of silt flowed into streams from farmlands, pesticides had significant impacts or were suspected of having them, or other agriculture-related factors directly affected the streams in which a salmonid species lived. Agriculture was regarded as having a low impact if it was not pervasive in the watersheds in which the species occurred or was not causing significant degradation of aquatic habitats. The effects of marijuana growing, legal, illegal or quasi-legal, was considered in this metric.

Grazing. Livestock grazing was separated from other forms of agriculture because its effects are widespread on range and forest lands throughout California and because it can have disproportionate impacts on stream and riparian habitats. Impacts were considered high in areas where stream channel morphology has been altered (e.g., head cuts, stream bank sloughing, lie-back, loss of meanders) and riparian vegetation removed, resulting in streams becoming incised with accompanying drying of adjacent wetlands or meadow systems. Other impacts contributing to a high rating include removal of vegetation and unimpeded cattle movement through streams, resulting in large amounts of silt and nutrient input, increased summer temperatures, and decreased summer flows. Impacts were rated low where grazing occurs in watersheds occupied by a species, but changes described above are minimal.

Rural/residential development. As California's human population grows, rural development increasingly occurs in diffuse patterns along or near streams. Resulting impacts include water removal, streambed alteration (to protect houses from flooding, create swimming holes, construct road crossings, etc.), and pollution (especially from septic tanks and illegal waste dumping). Where such rural development is increasing rapidly and largely unregulated, it may cause major changes to stream habitat quality and quantity and was rated as a high impact. Where such housing is present but widely dispersed and/or not rapidly increasing, the effects were rated as low.

Urbanization. Development of towns and cities often negatively affects nearby streams, largely due to flood prevention, channelization, water diversion, and increased waste inputs. The timing and magnitude of flows are altered by the increase in impervious surfaces associated with heavily developed areas. Streams in urban settings may be channelized, sometimes confined to cement canals, and/or diverted into underground culverts, significantly reducing the quality of fish habitat. Pollution from surface runoff, sewage discharges and storm drains can substantially degrade water quality and aquatic habitats. The impacts from urbanization were rated high where a salmonid occupies or passes through urban-influenced habitats for one or more stages of its life cycle.

Instream mining. Widespread and often severe instream mining impacts occurred during the mid-19th and early 20th century in California, due largely to hydraulic mining. Many rivers were excavated, dredged and hydraulically mined for gold, causing dramatic stream degradation; these legacy effects are still evident in numerous watersheds (e.g., the 'Gold Fields' on the lower Yuba River and the expansive tailing piles along the lower American and Trinity rivers). Locally severe impacts also occurred as a result of instream gravel mining operations, for which large pits were dug into streambeds and stream banks and riparian vegetation were highly degraded. Such mining is now largely banned (in favor of mining off-channel areas) but lasting habitat impacts remain in many areas. Instream mining was usually rated moderate when present, although severe legacy effects at a localized level resulted in high ratings for impacts to some species.

Mining. This factor refers to hard rock mining, from which tailings may have been dumped into streams, largely due to proximity of mines to stream courses, along with toxic pollutants entering streams from mine effluents, mostly from abandoned mines. Effects of mercury mining, used for processing gold in placer and dredge mining, are also included. High ratings stemmed from large-scale mines, even if abandoned or remediated, that may constitute a major threat because their wastes are considerable and adjacent to rivers (e.g. Iron Mountain Mine, near Redding, and Leviathan Mine, in the upper reaches of the East Fork Carson River). Low ratings were applied to mines near water courses where effects were deemed to be minimal.

Transportation. Road and railroad construction historically followed river courses across many parts of California; thus, a large number of rivers and streams have roads and/or railroads running along one or both banks, often for long distances (e.g., Klamath, Trinity, and Salmon rivers). These transportation corridors generally confine stream channels and subject waterways to increased sediment input, pollution, and habitat simplification. Culverts and other passage or drainage modifications associated with roads often block salmonid migration or restrict fish movements, sometimes fragmenting populations. Unsurfaced roads can become hydrologically connected to streams, increasing siltation and changing local flow regimes, with corresponding impacts to aquatic habitats. Ratings were generated based on how pervasive and proximate paved or surfaced roads, unsurfaced roads, railroads, or other transportation corridors are to streams in the areas occupied by a given species.

Logging. Timber harvest has been a principal land use of forested watersheds in California since the massive influx of immigrants in the mid-19th century. Timber harvest that supported historic development of mining towns, mines, railroads, and suburban and urban development led to deforestation of most of California's timber lands, often several times over. Many heavily-logged watersheds are those that supported the highest diversity and abundance of fishes, including anadromous salmon and steelhead (particularly North Coast watersheds). Logging was generally unregulated until the mid-20th century, resulting in substantial stream degradation across the state. Impacts, past and present, include: increased sedimentation of streams, increased solar input and resultant increase in stream temperatures, degradation or elimination of riparian vegetative cover, and an extensive network of statewide unimproved roads to support timber extraction, many of which continue to contribute to stream habitat degradation. Logging continues across large portions of the state and is considerably better regulated than in the past, with fewer impacts. However, legacy effects of past unregulated timber harvest continue to impact streams across California. High ratings were applied where a salmonid occupies streams notably degraded by either legacy or contemporary impacts from logging. Low ratings were applied to salmonids that occupy forested watersheds where the impacts from logging have either been mitigated or are considered to be minimal.

Fire. Wildfires are a natural and fundamental component of California's landscape in most parts of the state; however, Euro-American activities (especially fire suppression for greater than 100+ years), coupled with climate change, have made modern fires more frequent, severe and catastrophic (Gresswell 1999, Noss et al. 2006, Sugihara et al. 2006). Transition from relatively frequent understory fires to less frequent, but catastrophic, crown fires has been implicated as a major driver in increasing extinction risk of Gila trout (*Oncorhynchus gilae*) in New Mexico (Brown et al. 2001). It is likely that similar changes in fire behavior in California will affect native fishes in the same fashion. Ratings were based upon the extent to which habitats occupied by a species exist in fire-prone watersheds. Larger, main-stem river systems (e.g., Sacramento River), not often directly influenced by fires, were given low ratings.

Estuarine alteration. Many California salmonids depend on estuaries for at least part of their life cycle. Most estuaries in the state are highly altered from human activities, especially diking and draining, as well as removal of sandbars between the estuary and ocean. Land use practices surrounding estuaries often involve extensive wetland reclamation, greatly reducing nutrient inputs, ecological function and habitat complexity of estuaries. Impacts to salmonid species that are highly dependent on estuary habitats for one or more portion of their life history and that occupy rivers or streams with altered or degraded estuarine habitats were rated high. Impacts to those salmonids not dependent on, but still using, estuary habitats or present in drainages with little-modified estuaries were rated low.

Recreation. Human use of streams, lakes and surrounding watersheds for recreational purposes has greatly increased with human population expansion in California. Recreational uses that may cause negative impacts to salmonid populations and their habitats include: boating (motorized and non-motorized) or use of other personal watercraft, swimming, angling, gold dredging, off-road vehicles, ski resort development, golf courses and other activities or land uses. Recreational impacts to salmonid populations are generally minor; however, concentration of multiple activities in one region or during certain portions of the year may cause localized impacts. Recreation was rated high in situations where one or more factors have been documented to substantially impact riparian or instream habitats (including water quality), salmonid abundance or habitat utilization (e.g., spawning disruption), or in instances where the salmonid has very limited distribution and recreational impacts may further restrict its range or abundance. Recreation was rated low in cases where one or more recreational factors exist within a salmonids range, but effects are either minimal or unknown.

Harvest. Harvest relates to legally regulated commercial, tribal, and recreational fisheries, as well as illegal harvest (poaching). If not carefully monitored and enforced, all harvest can have substantial impacts on fish populations, particularly those with already limited abundance or distribution, those which are isolated or reside for long periods in discrete habitats and are, therefore, easy to catch (e.g. summer steelhead), or those that are comprised of long-lived individuals or those that attain large adult size (e.g., Chinook salmon), making them especially susceptible to over-harvest. Harvest was rated high where a salmonid was affected by one or more stressors noted above and it is believed that harvest limits abundance. Harvest was rated low where legal take is allowed but harvest rates are low so do not appear to limit abundance (e.g. coastal cutthroat trout).

Hatcheries. Hatcheries and releases of hatchery-reared salmonids into the wild can negatively impact wild fish populations through competition, predation, potential introduction of disease, and loss of fitness and genetic diversity (Kostow 2008, Chilcote, Goodson, and Falcy 2011). Despite decades of widespread stocking of hatchery strains of inland and anadromous species, today, many California salmonids have no hatchery augmentation and/or occur in waters that are no longer stocked; hatchery influences are largely relegated to anadromous salmonids that occur in rivers blocked by major dams (e.g., the various races of salmon and steelhead) or those that occur in lake or reservoir habitats that are stocked for recreational purposes (e.g., Eagle Lake rainbow trout). The severity of hatchery impacts were rated based, in part, on hatchery dependence to support a salmonid and or the threat of interbreeding between wild and hatchery populations.

Non-native species. Non-native species (including fishes and other aquatic organisms, aquatic vegetation, etc.) are ubiquitous across many of California's watersheds; their impacts on native species through hybridization, predation, competition, disease, and habitat alteration can be severe (Moyle and Marchetti 2006). This factor was rated high if studies and publications exist that demonstrate major direct or indirect impacts from non-native invaders on a given salmonid. The presence of non-native species was rated low if there is potential for contact with salmonids, but no negative impacts are known.

Table 4. Criteria for ratings assigned to anthropogenic threat factors with correlated time-lines.

Factor Threat Rating	Criteria	Timeline
Critical	Could push species to extinction	3 generations or 10 years, whichever is less
High	Could push species to extinction	10 generations or 11-50 years, whichever is less
Medium	Unlikely to drive a species to extinction by itself but contributes to increased extinction risk	Next 100 years
Low	May reduce populations but extinction unlikely as a result	Next 100 year
Not applicable (n/a)	Metric is not applicable to species	

Certainty of information

Because the quality and quantity of information varied among salmonid species, each account was rated on a 1-4 scale, for certainty of the status determination that resulted (Table 5). A score of 1 represented a salmonid for which the rating largely depended on the authors' professional judgment, with little or no published information. Scores of 2 and 3 were assigned to salmonids with ratings based on moderate amounts of published or gray literature, or where gaps existed in some important areas. A score of 4 meant the account was based on highly reliable information, with good information in the peer reviewed and agency literature.

Related publications

Astute readers will find overlap in language with other reports produced over the years by Peter Moyle's laboratory group at UC Davis, which has been continuously evaluating the fish fauna of California since the first edition of *Inland Fishes of California* (1976) and the first edition of *Fish Species of Special Concern in California* (1989). The language has evolved and expanded as the information has expanded, through Moyle et al. (1995), Moyle (2002), the first edition of this report (Moyle et al. 2008) and Moyle et al. (2015). The writing and editing of all these documents has involved members of the Moyle laboratory, so Moyle has the ultimate responsibility for the 'self-plagiarism' that exists.

RESULTS

Of the 32 kinds of salmonids found in California, 20 (63%) are endemic to California and five more are also found only in Oregon, so 25 species (78%) of the salmonids have no refuges outside our region. One species (Bull trout, *Salvelinus confluentus*) is extirpated from California. Of the 31 remaining taxa, 14 (45%) are listed as threatened or endangered by federal or state governments, while another 14 (45%) are California Species of Special Concern. This means that nearly all (90%) of California's extant salmonids are formally recognized as being in danger of eventual extinction in the state, mostly within 50-100 years if present trends continue (Table 5). Two of the remaining species, chum salmon (*Oncorhynchus keta*) and pink salmon (*Oncorhynchus gorbuscha*), are widespread outside of California but southernmost populations have occurred in California, at least historically. However, these species are not recognized by fisheries agencies because of the lack of studies indicating that current individuals found in California are not 'strays' from more northern populations (mainly Washington) (Moyle et al. 2015).[1] Only the coastal rainbow trout (*Oncorhynchus mykiss*) is considered secure in its status; this 'species' is a taxonomic repository for all resident rainbow trout populations in the state, whether of natural or introduced origins. Rainbow trout from California have been introduced all over the world, often to the detriment of native fishes (Moyle 2002).

[1] Evidence compiled in the species accounts indicates reproducing populations of pink and chum salmon exist or have existed in the state, but because of their time and place of spawning, they are hard to detect. They are likely very rare or extirpated from California as self-sustaining populations.

Table 5. Listings by state and federal management agencies, status scores, and Levels of Concern for California's native salmonids, 2007 and 2017.

Species	Federal ESA/California ESA Listing	2017 Level of Concern	2008 Level of Concern[2]
Central California coast coho salmon	State and Federally endangered	Critical	Critical
Chum salmon[3]	None	Critical	Critical
Pink salmon	None	Critical	Critical
California coast Chinook salmon	Federally threatened	High	High
Central Valley fall-run Chinook salmon	State Species of Special Concern	High	Moderate
Central Valley late fall-run Chinook salmon	State and Federal Species of Special Concern	High	High
Central Valley spring-run Chinook salmon	State and Federally threatened	Critical	High
Sacramento River winter-run Chinook salmon	State and Federally endangered	Critical	Critical
Southern Oregon/Northern California coast coho salmon	State and Federally threatened	Critical	Critical
Upper Klamath-Trinity rivers spring-run Chinook salmon	Federal Sensitive Species, State Species of Special Concern	Critical	High
Upper Klamath-Trinity rivers fall-run Chinook salmon	Federal Sensitive Species, State Species of Special Concern	Moderate	Moderate
Southern Oregon/Northern California coast Chinook salmon	Federal Sensitive Species, State Species of Special Concern	Moderate	Low
Klamath Mountains Province summer steelhead	State Species of Special Concern	Critical	High
Northern California summer steelhead	Federally threatened	Critical	High
South-Central California coast steelhead	Federally threatened	Critical	High
Southern California coast steelhead	Federally endangered	Critical	High
Central California coast steelhead	Federally threatened	High	High
Central Valley steelhead[4]	Federally threatened	Moderate	High
Northern California winter steelhead	Federally threatened	Moderate	Moderate
Klamath Mountains Province winter steelhead	Federal Sensitive Species, State Species of Special Concern	Moderate	Low
Bull trout	Extinct	Extinct	Extinct
California golden trout	State Species of Special Concern	Critical	High
Eagle Lake rainbow trout	Federal Sensitive Species, State Species of Special Concern,	High	High
Kern River rainbow trout	State and Federal Species of Special Concern	Critical	High
Lahontan cutthroat trout	State and Federally threatened	High	High

[2] The 2008 status scores were converted to corresponding Levels of Concern to make findings from both analyses comparable.
[3] Chum salmon and Pink salmon were removed from the State Species of Special Concern list in 2015 because information on their status was regarded as inadequate by CDFW.
[4] Based on a greater understanding of the genetic relationships between steelhead and rainbow trout upstream of dams in the Central Valley, the scoring of these fishes changed from 2008 to 2017. Please see the full Central Valley steelhead account for details.

Little Kern golden trout	Federally threatened	High	High
McCloud River redband trout	State Species of Special Concern	Critical	High
Paiute cutthroat trout	Federally threatened	High	High
Coastal cutthroat trout	State Species of Special Concern	High	Moderate
Goose Lake redband trout	State Species of Special Concern	Moderate	Moderate
Coastal rainbow trout	None	Low	Low
Mountain whitefish	State Species of Special Concern	Moderate	Low

Our scoring of extant salmonid (N = 31) status indicates that 14 (45%) are of Critical Concern (score of 1.0-1.9), 8 (26%) are of High Concern, 7 (23%) are of Moderate Concern, and one (3%) is of Low Concern (Figure 2). These scores indicate that 23 (74%) salmonids in California are headed for extinction by the end of the century, if not sooner, if present trends continue. All salmonids in California are on a declining trajectory, except Coastal rainbow trout. In the first edition of the *State of the Salmonids* report (2008), 21 species were categorized as Critical or High Concern, while three species were considered of Critical Concern. In all, 25 (81%) of the salmonids were rated as worse off in the present analysis than they were in Moyle et al. (2008, Figure 2), which was presumably the result of three interacting factors: (a) continued decline from multiple factors, (b) improved scoring system, and (c) the 2012-16 drought.

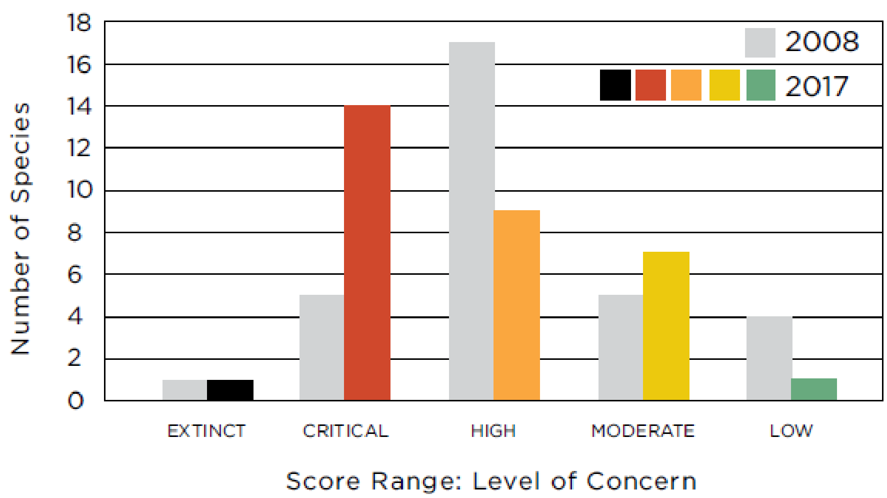


Figure 2. Change in Level of Concern in California's native salmonids, 2008 vs. 2017.

14 species were rated as Critical Concern - likely to go extinct in 50 years - compared to five species in 2008. Species most likely to disappear from California include coho salmon (*Oncorhynchus kisutch*) (two ESUs), chum salmon, pink salmon, Central Valley spring-run Chinook salmon (*Oncorhynchus tshawytscha*), Sacramento River winter-run Chinook salmon, Upper Klamath–Trinity River spring-run Chinook salmon, Klamath Mountains Province and Northern California summer steelhead (*Oncorhynchus mykiss irideus*), steelhead from three ESUs in south and central California, California golden trout (*Oncorhynchus mykiss aguabonita*), Kern River rainbow trout (*Oncorhynchus mykiss gilberti)*, and McCloud River redband trout (*Oncorhynchus mykiss stonei)*.

Of the 31 extant species, 21 are anadromous salmon and steelhead[5] while 11 are inland trout or whitefish species (Figure 3). 16 (76%) of the anadromous species were scored as Critical (11) or High (5) concern, indicating a high likelihood of extinction before the turn of the century. Seven (64%) of the inland trout species scored as Critical (3) or High (4) Concern.

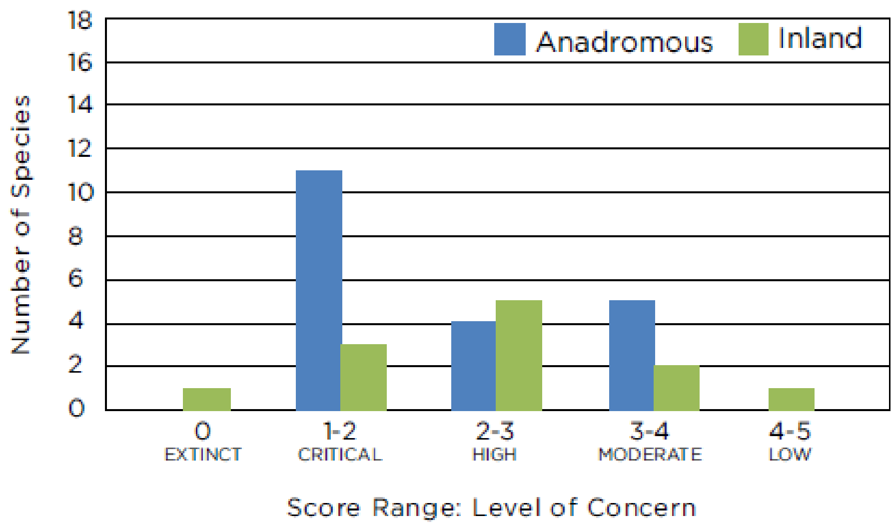


Figure 3. 2017 Levels of Concern for California's native anadromous and inland species.

Climate change

Climate change is the major, overarching anthropogenic threat threatening salmonids in California. Due to its importance to the continued persistence of all salmonids in California, it is highlighted here. While causes of decline of fishes in the Critical and High Concern categories were multiple and interactive, climate change was an important contributing factor for all species and was rated as a critical or high threat for 29 (87%) of the species (Figure 4). Anadromous species were most at risk from climate change, including Sacramento River winter-run Chinook, Central Valley late fall-run Chinook, Southern Oregon/Northern California Coast coho, Central California Coast coho, Upper Klamath-Trinity rivers spring-run Chinook, Central Valley spring-run Chinook, pink salmon, chum salmon, Central California Coast steelhead, South-Central California Coast steelhead, Southern steelhead, Klamath Mountains Province summer steelhead, and Northern California summer steelhead. Inland species most at risk included Eagle Lake rainbow trout, California golden trout, Little Kern golden trout, McCloud River redband trout, Kern River rainbow trout, and Lahontan cutthroat trout.

[5] Coastal cutthroat trout are included as anadromous species in this analysis, while mountain whitefish are an inland species.

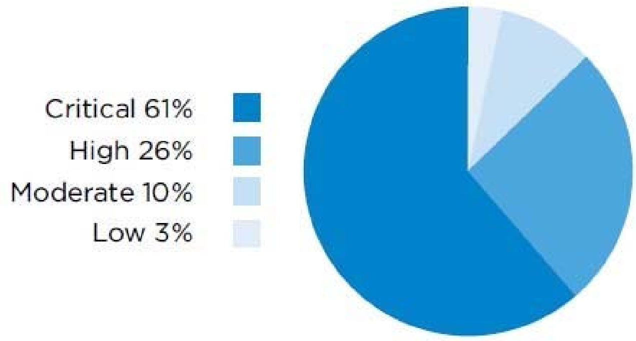


Figure 4. Climate change threat score for California's 31 extant native salmonids by percentage.

Other anthropogenic threats
Other leading causes of decline for anadromous species are estuary alteration (13 species), major dams and diversions (13 species), and agricultural operations (8 species). For inland species, non-native species such as non-native trout (7 species), fire (2 species), and hatcheries (2 species) were identified as the most common critical or high threats (Appendix A). Transportation (17 species) and hatcheries (16 species) also negatively impact a majority of California's salmonids.

DISCUSSION

California marks the southern range extent for Chinook, coho, chum, and pink salmon, but also steelhead, cutthroat, and redband trout. These populations are vitally important for species persistence due to their inherent genetic and life history diversity contributions to broader species dynamics (Hampe and Petit 2005, Schindler et al. 2010). Salmonids are also sentinel species; if they disappear so does the complex coldwater biota associated with them. For these reasons, the general decline of native salmonid taxa in California is of significant global concern. As stated by Peel et al. (2017, p.1389) "... the negative effects of climate change cannot be adequately anticipated or prepared for unless species responses are explicitly included in decision making and global strategic frameworks."

We project that 45% and 74% of salmonids in California will be extirpated in 50 and 100 years, respectively, if present trends continue. Since 2008, 25 of 31 taxa (81%) saw their level of concern increase, indicating that they are worse off today than they were less than a decade ago. Changes in species status are the result of the 2012-2016 historic drought, improved data collection and review, and an improved understanding of climate change impacts on fishes of California (see Moyle et al. 2013). In general, anadromous salmonids in California are more likely to face extirpation in the next century than their inland counterparts. The discrepancy is likely due to differences in habitat usage and the life histories associated with different forms. Salmon, steelhead, and coastal cutthroat trout rely on fresh, brackish, and saltwater habitats during different periods of their migratory life cycle. Conversely, resident salmonids rely on smaller geographic ranges sometimes confined to a single watershed, and may be exposed to relatively fewer threats throughout their lifecycle. It is also easier to take conservation measures for them, such as protecting small watersheds. Habitat quality, therefore, over a broad geographic area is of paramount importance for the continued persistence of anadromous salmonids (CDFW 2015c). Migration through California's severely altered landscape has exposed anadromous forms to simplified and degraded habitat throughout their range and increased their vulnerability to population declines. For instance, California's long history of dam and levee building has greatly reduced access to historical spawning and rearing habitat throughout the state (Yoshiyama et al. 2001, Hanak et al. 2011). Reduced access to diverse habitats may contribute to reductions in salmonid production and species resilience because fish are unable to express their full range of life histories under constrained habitats.

California's salmonids are highly vulnerable to climate change (Moyle et al. 2013), particularly because they encounter water temperatures that approach the upper limit of their thermal tolerances at the southern edge of their range. Climate change was found to be the major, overarching anthropogenic threat facing salmonids in California, with 84% of taxa rated as critically or highly vulnerable to climate change. This is not surprising considering that climate change is reducing availability of coldwater habitat for salmonids throughout the state and elsewhere. Wenger et al. (2011) projected nearly a 50% reduction in all trout habitat in the western United States by 2080, while Preston (2006) predicted a 60% decline in the United States by 2100. Overall, climate research in California and elsewhere indicates that earlier runoff, higher magnitude winter floods, more frequent and prolonged droughts, reductions in annual streamflow, and broad declines in thermal habitat for fishes are already occurring with greater changes anticipated (Stewart et al. 2005, Jefferson et al. 2007, Das et al. 2011). In addition, more precipitation in California is likely to fall as rain rather than snow in the future, which could negatively impact over-wintering rearing habitat for juvenile salmonids, and reduce the availability of cold water in rivers and streams in summer and fall months due to reduced snowpack (CDFW 2015c, Williams et al. 2016). Such abiotic shifts undoubtedly affect the physical habitat requirements of all salmonids and other coldwater fishes, but such changes are also anticipated to change species interactions, favoring introduced species (Moyle et al. 2013, Wenger et al. 2011, Muhlfeld et al. 2014).

Inland trout species with limited ranges, such as Eagle Lake rainbow trout and Little Kern golden trout may be particularly vulnerable to climate change. Range limited inland species are unable to naturally colonize adjacent watersheds or streams at higher elevations where cold water exists due to geologic or other barriers which inhibit dispersal. To complicate matters, degraded habitat, non-native species, hybrid trout, thermal barriers, and a host of anthropogenic threats are common downstream from many native inland trout headwater lakes and tributaries.

Migratory behavior makes anadromous forms vulnerable to climate change during both freshwater and marine life stages. Similar to inland species, climate change is also degrading migratory adult freshwater over-summering and juvenile rearing habitat through temperature increases and reductions in streamflow, and reduced access to coldwater tributaries for spawning (Williams et al. 2016). This may be a problem particularly for anadromous salmonids that over-summer in freshwater, such as spring-run Chinook salmon and summer steelhead. As stream temperatures increase, habitats become less suitable for native salmonids, causing changes in life history expression and timing such that juvenile salmonids may emigrate earlier and at smaller sizes to the Pacific Ocean, which may reduce survival (Mantua et al. 2015).

In the marine environment, climate change may reduce the powerful upwelling of the California

Current, which drives primary productivity and supports the food web for all marine life, including salmon and steelhead (Mantua et al. 2015). As a result, sea surface temperatures are also likely to increase over time. Elevated sea surface temperatures will likely cause changes in prey species composition, create thermal barriers to migrations, and, more generally, make marine habitats less suitable for growth and survival in the future (Wade et al. 2013).

Salmon, steelhead, and trout have adapted to a wide variety of climatic conditions in the past, and could likely survive anthropogenic shifts in climate in the absence of other anthropogenic stressors (Moyle et al. 2013). Yet, today, most salmonid species in California are less resilient than they once were. Improving resiliency requires an improvement in salmonid life history diversity. Salmonids have responded and adapted to environmental change for more than 50 million years due to variation in their life histories and behavior. Much of this variability is tied to differences in the timing of freshwater and ocean migrations. These timing differences contribute to life history diversity which, in turn, promotes species resilience to change. Over the last century, life history and behavioral diversity has been greatly diminished due to changes in habitat, discontinuity between habitats, genetic homogenization, and interactions with non-native species. The relatively recent reduction in salmonid life history and behavioral diversity means that salmonids are less able to adapt to a rapidly changing California. Access to diverse and productive habitats, and reductions in interactions between hatchery and wild salmonids, are fundamental to restoring salmonid resilience throughout California. Many of the historically productive and diverse habitats used by salmonids are either blocked behind dams and levees or are significantly altered and no longer function properly. Restoring such habitats and access to them is of paramount importance.

In short, if native salmon, trout, and other coldwater fishes are going to continue to be part of California's natural heritage, it is essential to invest in productive and diverse habitats to promote salmonid resilience. Below, we recommend a bold, forward-looking strategy that will benefit not only salmonids but coldwater dependent species in general, such as declining amphibians. The strategy we present is general and statewide. More specific application of this strategy can be seen, in part, in the *Sacramento Salmon Resiliency Strategy 2017* of the State Natural Resources Agency (CNRA 2017).

1) *Strongholds* - this entails protecting and improving through management, the best habitats left in California. These are the few fully functioning river ecosystems left in the state, that have relatively intact watersheds and high-quality habitat, such as the Smith River, Blue Creek, the Eel River, and Butte Creek among others. Protecting entire ecosystems is reason enough to manage systems like these in perpetuity the highest priority to protect salmonid diversity and production.

2) *Protect and Restore Source Waters* - protecting and restoring source waters, including meadows, springs, and groundwater, will allow them to continue to provide refuges for salmonids during stressful times and buffer the effects of climate change. Source headwaters are key to hydrologic connectivity and are vital during heat waves and drought.

3) *Restore Productive and Diverse Habitats* - restoring function to once-productive but now highly altered habitats can greatly improve rearing conditions for juvenile salmonids, especially

floodplains, coastal lagoons, estuaries, and spring-fed rivers. These types of habitats are relatively scarce, yet are vital nurseries for juvenile fishes and support robust growth rates when compared with typical in-river conditions. Improved growth prior to ocean migration and high life history diversity increases the likelihood of marine survival and adult returns to natal tributaries.

4) *Adopt Reconciliation Ecology as the Basis for Management* - Reconciliation Ecology recognizes that most ecosystems are altered by human actions, with people as a key part of the ecosystem. Therefore, highly managed ecosystems in working landscapes must play a major role in contributing to salmon diversity and abundance. If the mechanisms supporting growth and life history diversity can be recreated in human-dominated ecosystems, these landscapes can be put to work to support salmonids throughout California, while maintaining life history diversity. Current work on the Yolo Bypass in the Central Valley, for example, shows that rice fields can mimic natural floodplains with substantial growth benefits to juvenile salmon.

5) *Improve Habitat Connectivity and Passage to Historical Spawning and Rearing Habitat* - removing dams and fish passage barriers or providing volitional passage to historically important spawning and rearing habitats is key for persistence of many anadromous salmonids. Access to lost habitats will help boost population abundance, improve life history diversity, and population resilience to environmental changes. For populations downstream of dams, there is a need to institute scientifically-based environmental streamflow regimes throughout California that favor native species.

6) *Improve Genetic Management* - broad changes to the way salmonid hatcheries are operated throughout California need to be instituted. Changes should include reducing gene flow between hatchery and wild salmonids, minimizing hatchery straying into non-natal watersheds, marking hatchery fish with an adipose fin clip so that they can be readily distinguished from wild fish, and using strict mating protocols to discourage inbreeding and fitness reduction.

Ecosystem managers must be flexible and resilient in response to change wrought by the ever-increasing human demands on the planet. Failure to implement these strategies essentially means accepting the loss of most native salmonids in California, with a few maintained as low-diversity "boutique" populations, to remind us of our lost past and bleak future (Lackey, Lach, and Duncan 2006). Accepting the loss of salmonids means accepting the loss of numerous other coldwater species as well, and their special habitats. We don't have to accept these losses if we accelerate a bold conservation strategy now. We see salmonid conservation as a 'front line' effort that puts California in the lead for global biodiversity conservation in the Anthropocene (Kueffer, and Kaiser-Bunbury 2014, Chornesky et al. 2015).

SALMON

CALIFORNIA COASTAL CHINOOK SALMON
Oncorhynchus tshawytscha

High Concern. Status Score = 2.9 out of 5.0. Vulnerable to extinction in next 50-100 years if present trends continue and stream conditions deteriorate under climate change.

Description: Both male and female Chinook salmon have small black spots on the back, dorsal fin, and both lobes of the tail, though many adults can also have large, chevron-type spotting on the back and dorsal fin. Spotting on the caudal fin and the black coloration at the base of their teeth differentiate them from other sympatric salmonid species such as coho (*Oncorhynchus kisutch*), which have white at the base of the teeth. They have 10-14 major dorsal fin rays, 14-19 anal fin rays, 14-19 pectoral fins rays, and 10-11 pelvic fin rays. There are 130-165 scales along the lateral line. Branchiostegal rays number 13-19. They possess more than 100 pyloric caeca and have rough and widely spaced gill rakers, 6-10 on the lower half of the first gill arch.

Spawning Chinook adults are the largest Pacific salmonid, typically 75-80 cm SL, but lengths may exceed 140cm. California Chinook are usually smaller; Puckett (1972) found the average length of Eel River Chinook to be around 56 cm FL. The average weight is 9-10 kg, although the largest Chinook taken in California was 38.6 kg. Spawning adults may range in color from olive brown to dark maroon without streaking or blotches on the side. Males are often darker than females and develop a hooked jaw and slightly humped back during spawning. Juvenile Chinook have 6-12 parr marks, which often extend below the lateral line; the marks are typically equal to or wider than the spaces between them. Parr can also be distinguished from other salmon species by the adipose fin, which is pigmented on the upper edge, but clear at the base and center. Most fins are clear, though some parr begin to show spots on the dorsal fin as they grow. There are no morphological features to separate this Evolutionary Significant Unit (ESU) from other Chinook salmon ESUs, so separation is based on genetic data.

Taxonomic Relationships: The California Coastal Chinook salmon (CC Chinook) Evolutionary Significant Unit (ESU) includes Chinook salmon that spawn in coastal watersheds from Redwood Creek (Humboldt Co.) in the north to the Russian River (Sonoma Co.) in the south, inclusive. Chinook salmon found occasionally in coastal basins south of the Russian River (e.g., Lagunitas Creek, Marin Co.) are also under consideration to be in included in this ESU based on Williams et al. (2011), though no changes have been made to the ESU boundaries at this time. Recent genetic analyses demonstrate some differentiation among populations. Bjorkstedt et al. (2005) found that CC Chinook in the Eel River and northern watersheds differ from those on the Mendocino coast and in the Russian River. Differentiation among fish from different tributaries to the Eel River is low, suggesting high dispersal of Chinook in the basin. Generally, fish from the Russian River are genetically more similar to Chinook from the Eel River than to fish from the Central Valley fall Chinook ESU (Hedgecock et al. 2002, Bjorkstedt et al. 2005).

NMFS (2016b) divided the ESU into three groups through genetic analyses (Figure 1):

- 13 independent populations: Bear River, Big River, Garcia River, Humboldt Bay tributaries, Lower Eel River (Van Duzen River and Larabee Creek), Lower Eel River (South Fork and Lower mainstem Eel River) Little River, Mad River, Mattole River, Noyo River, Redwood Creek (Humboldt Co.), Russian River, and Upper Eel River;
- 3 supporting independent populations: Gualala River, Navarro River and Ten Mile River;
- 1 dependent population: Albion River

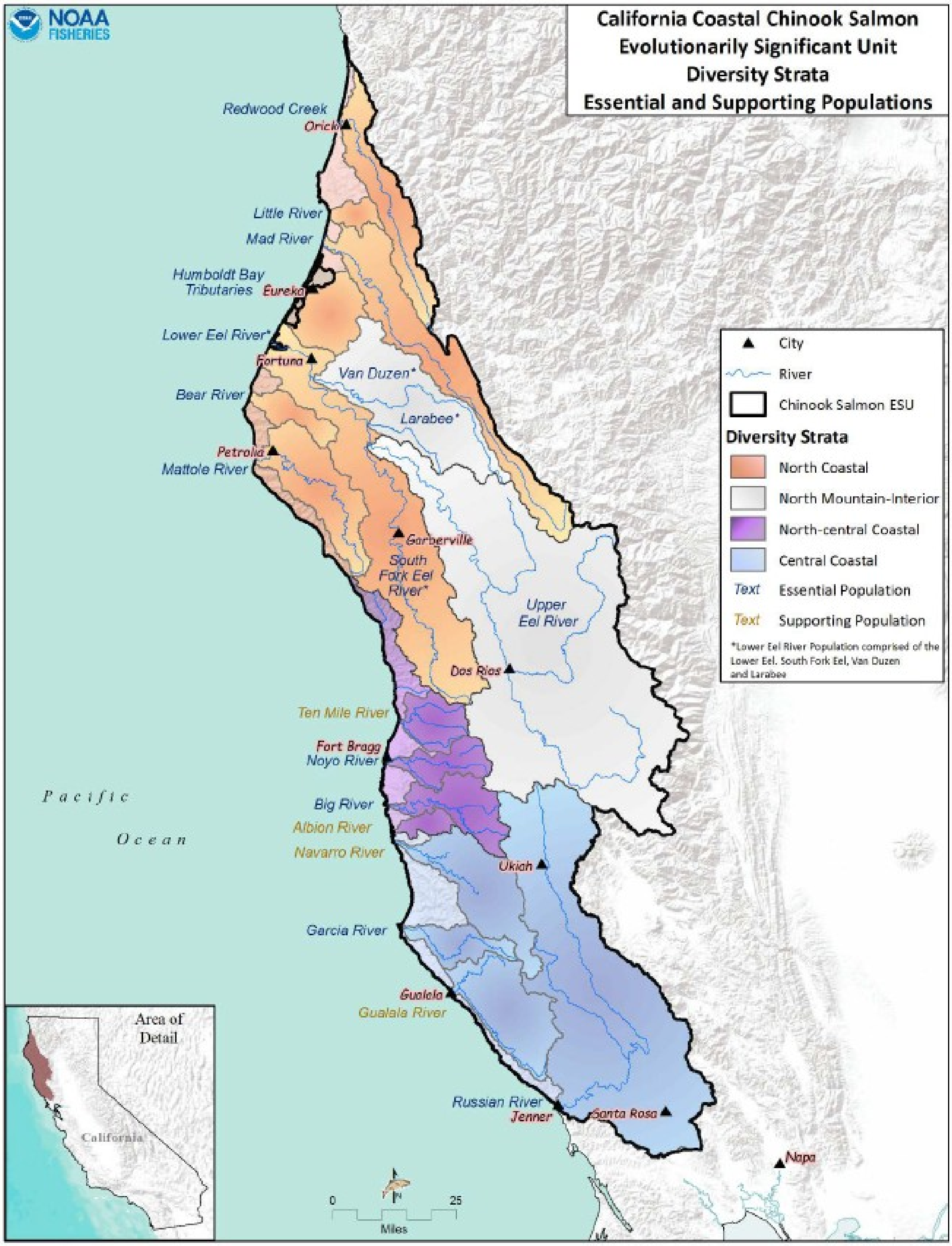


Figure 1. CC Chinook ESU boundaries and diversity strata. From NMFS 2016, Fig. 1. pg. 14.

Life History: Existing populations of CC Chinook in the ESU are characterized as a having a fall–run salmon based life history; the spring-run life history strategy has been lost throughout the ESU and represents a key source of genetic diversity loss in the ESU (NMFS 2016). There used to be significant natural variability in the timing of peak spawning runs of CC Chinook due to precipitation and its influence on stream flows and passage in relatively short coastal watersheds. For example, spring-run Chinook historically accessed streams in the Middle Fork and upper mainstem Eel River upstream of where Scott Dam now stands (J. Fuller, NMFS, pers. comm. 2017). CC Chinook typically return to their natal rivers between September and early November, often following large early winter storms. They are most abundant in open-estuary type systems throughout their range, and will stage in the lower reaches of rivers until cued upstream by hydrologic conditions (J. Fuller, NMFS, pers. comm. 2017). In smaller coastal watersheds or those with seasonal open estuaries, fall rains open the mouths by November (M. Sparkman, CDFW, pers. comm. 2016).

Spawning in the larger basins peaks between late October and December, but in smaller watersheds it follows the timing of entrance into the natal stream more closely due to the more flashy streamflows that allow passage into these systems (J. Fuller, NMFS pers. comm. 2016). CC Chinook salmon may spawn immediately or may rest in holding pools for considerable time when early storms permit entrance to rivers but not access to preferred spawning habitat upstream. Most Chinook salmon migration activity occurs during a few distinct movement events per year, usually dictated by fall rains, increases in river flows, and cooler water temperatures between September and December on the Russian River (SCWA 2008). In Redwood Creek in 2016, 95% of the adults swam upstream within two weeks of the mouth opening to the ocean. In drought type water years, the migration is more prolonged (M. Sparkman, CDFW, pers. comm.. 2017). Mature Chinook females produce 2,000-17,000 eggs (Moyle 2002). Adults die within a few days after spawning and their carcasses become a source of food for a wide array of animals, including juvenile steelhead and coho salmon. They also fertilize riparian and stream ecosystems with marine-derived nutrients and trace elements, presumably increasing carrying capacity of the streams for their own young.

The vast of majority of CC Chinook salmon demonstrate an "ocean-type" juvenile life stage. Under this life history strategy, fry emerge from the gravel in the late winter or spring and initiate outmigration within a week to months of emergence when they are relatively small, about 30-50 mm FL. Emigration of smaller fish is likely a function of low stream carrying capacity, with later emerging fry only finding saturated habitats, forcing them to seek unclaimed rearing habitat. As they grow, the parr move into deeper and faster water to seek greater foraging opportunities, dispersing downstream as they opportunistically forage on drifting terrestrial and aquatic insects. Slow water habitats are still important to juvenile Chinook, but are used primarily during daytime, when the fish hide in deep cover to reduce predation and for energy conservation. In streams such as Redwood Creek, the peak month in emigration is typically June through lower river sections. Small numbers of "stream-type" parr will oversummer in the northern coastal watersheds of this ESU; these large (ca. 10+ cm FL) juveniles migrate out to sea when stream flows rise following large fall rainstorms (Bjorkstedt et al. 2005) or as yearlings in the spring (M. Sparkman, CDFW, pers. comm. 2016).

CC Chinook may reside in estuaries, lagoons, and bays for a few months to take advantage of feeding opportunities and grow in size, and then exit these habitats gradually over the summer (Healey 1991). Historically, estuaries with year-round access to the ocean were favorable juvenile habitat and fish had greater flexibility to leave or to remain in the estuaries until fall/winter when habitat conditions change. The extended occupancy by smolts of these habitats suggests enhanced growth may benefit ocean survival (Williams 2006). In the Russian River, Cook (2005) observed Chinook to be habitat generalists found throughout the estuary. Juvenile Chinook were captured 38% of the time at tributary junctions within the estuary. At these locations they presumably fed on aquatic (drift) and terrestrial insects, supplied from the surrounding and upstream riparian corridors. Chinook presence in the Russian River estuary typically peaks in early June. Most juveniles have emigrated downstream past Sonoma County Water Agency's inflatable Mirabel Dam and diversion facility (Rkm 37), near Forestville, CA, by early July (CDFW 2007). Estuaries with summer-forming sandbars appear to have high juvenile mortality due to unfavorable summer estuarine water quality and habitat conditions. In 2007, large numbers of Chinook juveniles were observed in the Mattole River estuary in July, following a significant summer rain event. Although the estuary was closed to the ocean, by August very few Chinook were observed in the estuary or upstream habitats, suggesting mortality was very high due to the combination of lack of access to the ocean and inhospitable estuarine conditions (Mattole Salmon Group 2016).

Once they enter the ocean, CC Chinook salmon migrate along the California coast, often moving northward, where they mix with salmon from other river systems, including hatchery fish, and feed in the cool waters off of the Klamath-Trinidad region. Because salmon spend a majority of their lives at sea, shifts in ocean productivity play a large role in their survival, growth, and overall abundance. Chinook salmon are predators in the ocean, feeding on small fish and crustaceans such as copepods, anchovies, sardines, and krill. As their size increases, fish increasingly dominates their diet. This piscivorous diet provides for rapid growth on the order of 0.35-0.57mm/day (Healey 1991). CC Chinook salmon typically return after two to three years at sea; the most common ages-at-maturity for CC Chinook are three and four years. Five year and six year old fish contribute a small proportion to the spawning population, although their limited numbers may be due to effects of fisheries selectivity for larger individuals over the past century and in-river predation on the largest fish (Myers 1998).

Habitat Requirements: Habitat requirements for Chinook salmon are described in detail in Healey (1991) and Moyle (2002). Temperature is an important factor in Chinook salmon survival and growth, and tolerances vary with life history stage and by ESU (Table 1). Likewise, they are sensitive to dissolved oxygen levels, water clarity and other factors that indicate high water quality. For example, CC Chinook may not be as tolerant of warm temperatures as Central Valley Chinook salmon (J. Fuller, NMFS, pers. comm. 2017).

Table 1. Chinook salmon thermal tolerances in fresh water. All lethal temperature data is presented as incipient upper lethal temperatures (IULT), which is a better indicator of natural conditions because experimental designs use a slower rate of change (ca. 1°C/day). Fish living in the wild experience temperatures that fluctuate on a daily basis and rarely stay in warmer water for long. Information largely from McCullough (1999); this is a synthesis of data from throughout the range of Chinook salmon, so may not precisely reflect the tolerances of CC Chinook salmon.

	Sub-Optimal	Optimal	Sub-Optimal	Lethal	Notes
Adult Migration	< 10ºC	10-20°C	20-21°C	> 21-24°C	Migration usually stops when temperatures climb above 21ºC, with partial mortality occurring at 22-24ºC. Lethal temperature under most conditions is 24ºC. Fish observed moving at high temperatures are probably seeking cooler refugia.
Adult Holding	< 10ºC	10-16°C	16-21°C	> 21-24°C	Adults experience heavy mortality above 21ºC under crowded conditions, but will survive temperatures up to 24ºC for short periods of time. In some holding areas, maximum temps exceed 20ºC for over 50 days in summer.
Adult Spawning	< 13ºC	13-16°C	16-19°C	> 19°C	Egg viability reduced with exposure to higher temperatures.
Egg Incubation	< 9°C	9-13°C	13-17°C	> 17°C	This is the most temperature sensitive phase of life cycle. American River salmon have 100% mortality >16.7°C; Sac. River fall-run salmon mortality exceeded 82% at temperatures>13.9°C.
Juvenile Rearing	< 13°C	13-20°C	20-24°C	> 24°C	*Past exposure (acclimation temperatures) has a large effect on thermal tolerance. Fish with high acclimation temperatures may survive 28-29ºC for short periods of time. Optimal conditions occur under fluctuating temperatures, with cooler temperatures at night. When food is abundant, juveniles that live under conditions that fluctuate between 16 and 24ºC may grow very rapidly.
Smolt Migration	< 10ºC	10-19°C	19-24°C	> 24°C	Smolts may survive and grow at suboptimal temperatures but are susceptible to predation; in the lab, optimal temperatures are 13-17ºC (Marine and Cech 2004), but observations in the wild suggest a greater range.

Due to their large size, Chinook spawning use the largest substrate of any California salmonid for spawning, which consists of a mixture of small cobble and large gravel. Such coarse material has sufficient subsurface infiltration, which provides oxygen for developing embryos. As a result, the selection of redd sites is often a function of gravel permeability and subsurface water flow. For CC Chinook, a majority of suitable spawning habitat is in the upper main stems of rivers and lower reaches of coastal creeks. These habitats provide stable substrate and sufficient flows into late winter. Typically, redds are observed at depths from a few centimeters to several meters in water velocities of 15-190 cm/sec. Preferred spawning habitat seems to be at depths of 30-100 cm and at water velocities of 40-60 cm/sec. Redds are typically constructed over 2-15 m^2, where the loosened gravels permit steady access of oxygenated water (Healey 1991). Redd size is a function of female size as well as looseness of the substrate. For maximum embryo survival, water temperatures must be between 5º and 13º C and oxygen levels must be close to saturation. Under optimal conditions, embryos hatch after 40-60 days and remain in the gravel as alevins for another 4-6 weeks, usually until the yolk sac is fully absorbed before emerging as fry (M. Sparkman, CDFW, pers. comm. 2017).

Once alevins emerge with their yolk sac absorbed, they become fry, which tend to aggregate along stream edges, seeking cover in bushes, swirling water, and dark backgrounds. As they grow larger and become increasingly vulnerable to avian predators, especially herons and kingfishers, they move into deeper (> 50 cm) water. Larger juveniles may wind up in the tails of pools or other moderately fast-flowing habitats where food is abundant and there is some protection from predators. As they move downstream, they use more open waters at night, while seeking protected pools during the day. Pools that are cooler than the main river, from upwelling or tributary inflow, may be sought out by the migrating juveniles as daytime refuges.

Juveniles and smolts that reach the estuary use food-rich tidal habitats, especially areas with overhanging cover or undercut banks. When given the opportunity, they will move into areas that have flooded either tidally or from freshets, to forage. Estuaries that present complex and variable habitats (i.e., that are not channelized, diked, and drained) are optimal for juvenile salmonids just before they go out to sea (Wallace et al. 2015).

In the ocean, habitats for the first few months are poorly documented, but it is assumed that the fish stay in coastal waters offshore of the Klamath-Trinidad region, where the cold California Current creates rich food supplies, especially small shrimp, by upwelling. During the day, they avoid surface waters. Subadult Chinook salmon hunt anchovies, sardines, herring, and other small fish, typically at depths of 20-40 m, moving offshore and into deeper waters in response to temperature, food availability, and predators, such as orcas and sea lions.

Distribution: This ESU includes Chinook salmon that spawn in coastal watersheds from Redwood Creek (Humboldt Co.) in the north to the Russian River in the south, inclusive (Figure 1). Chinook salmon found occasionally in coastal watersheds south of the Russian River (e.g., tributaries to Tomales Bay, Marin Co.) are also under consideration to become part of this ESU, but no new information has warranted that change (NMFS 2016). In general, small coastal streams within this range can support Chinook salmon spawning and rearing as long as they have timely open-estuary conditions with sandbar formations that are not constraining ocean connectivity during migratory periods. CC Chinook salmon south of the Eel River are typically present in year-round open-estuary systems (e.g. Garcia River) and rarely observed in systems impeded by sandbar formations (e.g. Gualala River). In general, timing of precipitation events and sandbar breaching in systems south of the Eel River occur too late to allow successful Chinook migration (J. Fuller, NMFS, pers. comm. 2017). California Coast Chinook salmon are distributed at the southern end of the species' North American range; only Central Valley fall Chinook are found spawning further south. NMFS identified four regions of this portion of the California coast with similar basin-scale environmental and ecological characteristics (Bjorkstedt et al. 2005). Sixteen watersheds were identified in these four regions that have minimum amount of habitat available to support independently viable populations.

In the North Mountain-Interior Diversity Strata, the Upper Eel and Middle Fork Eel rivers contain independent populations, while the Lower Eel and Van Duzen rivers have the potential to support dependent populations. Historically, Chinook were present in the North Fork Eel River up to a location known as the Asbill Roughs and Split Rock (USFS-USBLM 1996). Chinook are annually observed in the Middle Fork Eel River, in Black Butte River, and near

Williams Creek, though Scott Dam on Lake Pillsbury limits their upstream range. They continue to be observed annually in the Outlet Creek drainage and in small tributaries feeding Little Lake Valley (S. Harris, pers. comm. 2007). In the North Coastal Region, Redwood Creek and the Mad, Lower Eel, South Fork Eel, Bear and Mattole rivers all contain sufficient habitat for independent populations. On the Mad River, boulder roughs near Bug Creek (Rkm 80) precludes Chinook salmon passage upstream (Mad River Watershed Assessment 2010). Little River and Humboldt Bay tributaries may potentially host independent populations. In the North-Central Coastal Diversity Strata, numerous watersheds in Mendocino County contain small runs that are dependent upon self-sustaining stocks in Ten Mile, Noyo, and Big rivers. Big River, in particular, boasts a large, open estuary that could potentially support a self-sustaining population of Chinook, but without sufficient data to determine run sizes and timing, any Chinook returns are unpredictable at best (T. Daugherty, NMFS, pers. comm. 2017).

Along the Central Coastal Diversity Strata, the Navarro, Garcia and Gualala rivers historically had independent populations, but numbers have declined substantially. While some Chinook are seen annually in the Garcia River, the Navarro and Gualala further south have very limited information and few confirmed sightings of Chinook recently (J. Fuller, NMFS, pers. comm. 2017). Additionally, the Russian River supports a self-sustaining population of its own, although the historical and current influence of hatcheries and straying of fish from nearby basins is uncertain (Chase et al. 2007, CDFW 2007).

Seventeen additional watersheds were identified by NMFS to contain CC Chinook, but due to limited habitat were believed not to support persisting populations of these salmon (Good, et al. 2005). While Chinook salmon are also encountered in the San Francisco Bay region, these fish most likely originated from Central Valley populations that have been trucked from hatcheries downstream to bypass predation and entrainment threats in the Delta. These fish are then acclimated in pens near Carquinez Strait and other locations in the San Francisco Bay and are not included in the ESU. In the ocean, CC Chinook salmon are most frequently encountered in commercial fisheries from the Oregon border south to San Francisco Bay during the months of July and August (Satterthwaite et al. 2014).

Trends in Abundance: CC Chinook salmon abundance has declined to levels that are well below recovery targets and high-risk depensation thresholds, or reductions in egg survival and productivity due to shrinking effective spawning populations, established by NMFS (NMFS 2016). The remaining small population sizes have rendered the ESU vulnerable to stochastic processes, such as earthquakes, landslides, droughts, or flooding, and may lead to reductions in genetic diversity, altered breeding structure, and shifts in population dynamics (NMFS 2016). Yoshiyama and Moyle (2010) provide a history of salmon in the Eel River basin, which presumably reflects what has gone on in all watersheds in the region; they estimate that abundance of CC Chinook has decreased by more than 90% of historical numbers, though reliable population estimates for the Eel are severely lacking (J. Fuller, NMFS, pers. comm. 2017). In general, lack of long-term population monitoring across the ESU range, especially for the Upper and Lower Eel watershed and the Mad River populations, makes comparisons of current and historical abundance difficult (NMFS 2016). Returning numbers of adult CC Chinook over the last five years have shown a mix in population trends among regions: extremely low numbers exist in North-Central Coast and Central Coast Diversity Strata (NMFS 2016).

North Coastal Diversity Strata. CC Chinook that inhabit northern watersheds of the ESU (between Redwood Creek and the Mattole River) and Humboldt Bay appear to have runs of a few thousand spawners annually. On Redwood Creek, sonar estimates ranged from 1,600 to 3,400 returning adults per year, while the Mad River had an estimated 2,200 returning adults in 2016 (M. Sparkman, CDFW, pers. comm. 2017). Within Humboldt Bay, the smaller coastal tributaries also likely supported combined runs of several hundred fish. Presumably, CC Chinook runs in steep coastal tributaries such as Freshwater Creek are likely less than fifty adults per year, and the Elk River have been limited by spawning habitat, but expansive spring-flooded baylands and estuarine habitats may have resulted in high parr-to-smolt survival (M. Wallace, CDFW, pers. comm. 2007, M. Sparkman, CDFW, pers. comm. 2017), resulting in higher-than-expected numbers of returning adults. Chinook salmon have been observed in declining numbers in Freshwater Creek over the past decade. Adults continue to be captured at the Humboldt Fish Action Council's permanent weir in the lowest reach of Freshwater Creek, but in 1997- 2001, 30-70% of returning Chinook were of hatchery origin. Recent returns have fluctuated considerably and a recent adult population estimate (2002-2003) was 133 ± 63 Chinook entering Freshwater Creek (Ricker 2005). The Little River has small (on the order of 800-1,000) annual returning Chinook as well (M. Sparkman, CDFW, pers. comm. 2017). The Mattole River contains a CC Chinook population that contains perhaps a thousand adults based on counts of 300-400 redds per year from 2012-2015 (Mattole Salmon Group 2015).

North Mountain Interior Diversity Strata. The North Mountain Interior Diversity Strata encompasses all but the South Fork and lower Eel rivers. Steiner Environmental Consulting (1998) estimated historical Chinook abundance in the Eel River system based on cannery records compiled by Humboldt County (Humboldt Public Works 1991). From 1857-1921, SEC (1998) estimated that the average catch was 93,000 Chinook and coho salmon combined per year, with a peak of 585,000 fish in 1877. Similarly, Berg Associates (2002:107) stated, "from 1853 to 1922, fish packing and cannery records documented from 15,000 to 600,000 salmonids caught annually in commercial fisheries" (citing NMFS 2000). A large majority of these salmon were presumably Chinook salmon, as they are the most abundant and accessible to fishermen. There are no records of how many fish survived to spawn, but a conservative estimate would be that the annual runs of Chinook in the Eel River (catch plus escapement) in this early period were on the order of 100,000-600,000 fish per year (NMFS 2016b).

The early unrestricted fishery presumably greatly depleted the runs, but there are only scattered records to indicate run sizes after the canneries closed down. In 1965, CDFG suggested that the Eel River Chinook escapement approximated 88,000 adults. This number is presumably much lower than historical escapement. The Potter Valley Diversion Project on the Upper Mainstem Eel River had already been diverting water to the agricultural valleys of the Russian River watershed for almost forty years at the time, and Chinook were facing challenges from flow alteration, habitat degradation, pollution, and unregulated fishing (Shapovalov 1941). Benbow Dam, which was seasonally constructed across the South Fork of the Eel River, averaged approximately 12,000 Chinook between 1938 and 1952 (http://www.hits.org/salmon98/history/damrecords2.html), and multiple egg collection and hatcheries operated throughout the Eel River until the 1960s. During the last decade of the Benbow Dam fishway between 1965-1975 (Taylor 1978), average Chinook salmon counts declined to less than 5,000 fish annually and have continued in their decline. Chinook spawning was reported to occur in the South Fork Eel River between Bull Creek and Laytonville, and in the mainstem between Holmes and Van Arsdale Reservoir (Puckett and Hinton 1974).

In the Upper Eel River in 1975-76, an estimated 367 Chinook salmon entered Tomki Creek, the most productive upper mainstem tributary below Van Arsdale Reservoir (Brown 1976). By the 1990s, basin-wide escapement often numbered fewer than 5,000 fish, with numbers in the upper reaches dwindling to fewer than 50 fish in many years. The Van Arsdale Fish Station provides annual estimates of the Chinook entering the Upper Eel River; NMFS (2016) documented a significant positive trend in returns of CC Chinook passing the counting facility from 2000 through 2013 (Figure 2).

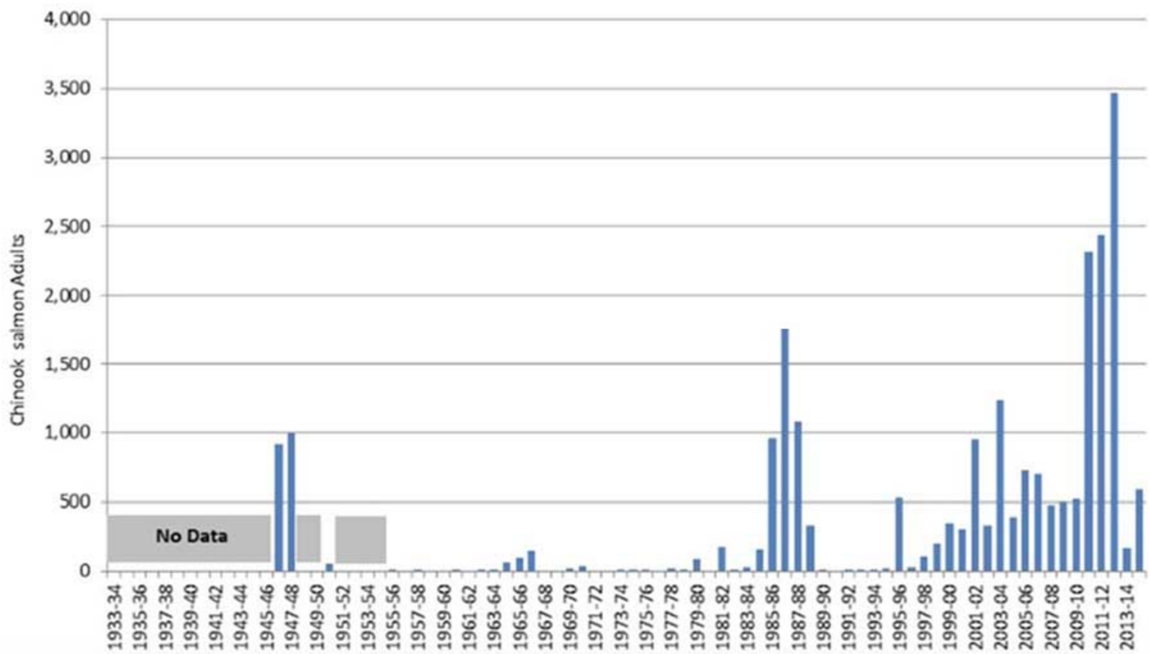


Figure 2. Adult Chinook salmon passing Van Arsdale Fish Station on the Upper Mainstem Eel River, 1933-2014. From NMFS 2016b, Fig. 1, pg. 91.

A number of the larger subbasins in the Eel River such as the Van Duzen, South Fork, and North Fork Eel rivers continue to support spawning runs, although monitoring data is limited and has only recently been improved through directed surveys in these watersheds. Redwood Creek, a small tributary to the lower South Fork Eel River once saw hundreds of Chinook returning annually, although numbers today fluctuate between 10 and 100 returning spawners (H. Vaughn, Eel River Salmon Restoration Project, pers. comm. 2016).

Eel River Basin wide salmonid population estimates are difficult to obtain due to the large watershed size, rugged and remote geography, and frequent low water clarity conditions that occur during adult salmon migration. The NMFS 2011 status review of North Coastal Chinook salmon (Williams et al. 2011) concluded "The lack of population-level estimates of abundance … continues to hinder assessment of its status." CDFW currently conducts spawning ground surveys of the South Fork Eel River and its tributaries via method designed to estimate coho spawning abundance. In some years CDFW has also surveyed salmonid spawning in Lawrence, Grizzly, Bull, Hollow Tree, Sproul, Outlet, and Tomki creeks. Since the 1940's CDFW has counted adult Chinook and steelhead passing a fish ladder at the Van Arsdale Fisheries Station (VAFS) located on the upper Mainstem Eel River Mile 158, 12 miles below Scott Dam at the end of anadromy (Figures 2 and 3).

Recent abundance trends have been upward (NMFS 2011; CDFG 2012), with several strong year-classes of Chinook salmon returning to the river. Chinook salmon adult returns at Van Arsdale Fish Station in 2011 and 2012 are recent examples of high Chinook abundance observed recently, and coho salmon spawning spawning estimates in the South Fork Eel River have remained steady. Higgins (2015) estimated total run-size for the 2014-2015 to be 12,500-20,000 spawners, 14,900-25,000 in 2013-2014 and 20,000-50,000 in 2012-2013. These estimates were made based on very limited direct observation surveys in the lower mainstem Eel combined with anecdotal observations from other locations across the basin. These numbers are hopeful because the river had very low flows during the recent drought (2013-2016). There are several possible explanations for the recent positive trend: (a) Improvements to land and forest management; (b) active restoration to improve salmonid habitat conditions and connectivity; (c) a period of recovery from the effects of the past century of timber harvesting and the devastating 1955 and 1964 floods; and (b) water years 2010 and 2011 had above-average spring runoff conditions, potentially contributing to higher juvenile survival and adult returns.

North-Central Coastal Diversity Strata. Monitoring data is sparse for coastal Mendocino watersheds such as Ten Mile, Noyo, Big, and Albion rivers. While CC Chinook are generally very low in abundance in these watersheds, generally numbering in the few hundreds of adults per year, they have been regularly reported over time and may currently support small but viable populations (NMFS 2016). Early logging practices likely severely depressed CC Chinook stocks in these rivers by eliminating passage along the main stems by frequent use of splash dams and loss of rearing habitat from heavy sedimentation of both rivers and estuaries. Observations of Chinook in these watersheds indicate that spatial gaps among populations are not as great as once believed (NMFS 2016).

Central Coastal Diversity Strata. CC Chinook in the Central Coastal Diversity Strata range from the Navarro River in the north to the Russian River in the south. Russian River Chinook are of uncertain genetic origin following close to fifty years of inter-basin stocking in the river between the early 1950s and 1999. Between 1980 and 1996, CDFG stocked approximately 2.25 million juvenile Chinook from various inter and intra-basin locations to establish a self-sustaining hatchery run. Unfortunately, returns were very low (< 300 adults/year) during this time. Although the Chinook hatchery program ended in 1999, biologists working for the Sonoma County Water Agency have observed more Chinook Salmon in the Russian River than any other anadromous salmonids present in the basin. In the 2005-2006 spawning season, more than 2,563 Chinook salmon were counted swimming through SCWA's fish ladder, and 1,383 to 6,081 Chinook were observed migrating past Mirabel Dam (Rkm 37) during the 2000 to 2004 spawning runs. These fish spawn primarily in the mainstem between Cloverdale (Rkm 101) and the confluence of the East and West Branches of the Russian River (Rkm 150), but spawning has also been observed in Austin, Santa Rosa, Green Valley, Dry, Feliz, and Forsythe creeks, but this contribution is smaller than that observed on Dry Creek (NMFS 2016b). Returns of adult Chinook observed passing the Sonoma County Water Agency's Mirabel diversion, a few river kilometers downstream of the Dry Creek confluence near Healdsburg, have been variable since the turn of the century (Figure 3).

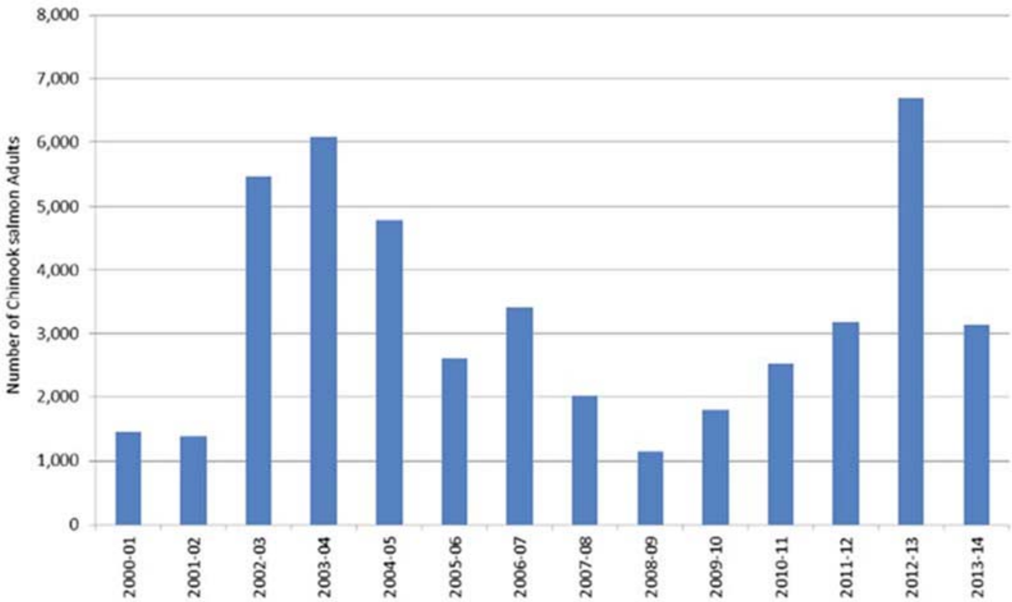


Figure 3. Minimum counts of adult Chinook salmon passing Mirabel diversion facility video station, Russian River (Sonoma Co.) 1980-2014. From NMFS 2016b Fig. 3, pg. 459.

Overall. CC Chinook salmon are now much less abundant across the ESU than they were historically, although monitoring has been always been sparse. It is reasonable to assume that in 'good' years, historic runs were on the order of 600,000 fish combined in the ESU, perhaps dropping to 30,000-50,000 in 'bad' years. Present numbers (even in good ocean years), based on insufficient data, seem to total about 5,000-30,000 fish annually. There are concerns about the extremely low returns of adult Chinook in the North-Central Coast and Central Coast strata, and the diminished connectivity among populations in the ESU that results from very small abundance in key watersheds (NMFS 2016).

Factors Affecting Status: The factors affecting CC Chinook salmon are multiple and interactive but somewhat different for each watershed (Moyle 2002). This multiple population structure provides some resilience to CC Chinook populations overall (portfolio effect). NMFS (2016b) found that broadly, the anthropogenic factors most likely to threaten the continued existence of CC Chinook are: channel modification, impacts from roads and railroads, and logging and wood harvesting. These and other threats are discussed more specifically below.

Dams. Dams and diversions affect water quality and quantity in both rural and urban watersheds. The main withdrawal of water in the ESU is the inter-basin transfer from the upper Eel River into the upper Russian River for Pacific Gas & Electric's Potter Valley Project. This project transfers less than 3% of the Eel River's total flows to the Russian River (Potter Valley Water 2016) via a 1.6 km tunnel to supply water during fall and spring for vineyard irrigation and municipal uses, which presumably indirectly helps to sustain all CC Chinook life stages in the mainstem Russian River. The transfer of water has presumably contributed to declines in Eel River CC Chinook runs by reducing flows available for out-migration by juveniles and for upstream spawning migration by adults. Water withdrawals from the Eel River likely affect water temperatures in the upper mainstem Eel, creating thermal barriers that restrict juvenile emigration to earlier in the spring. Dams on the Mad, Eel and Russian rivers have also influenced geomorphic regimes and decreased the quality of spawning habitat downstream through reduced flows and gravel recruitment. Ruth Dam on the Mad River is capable of drawing up to 75 million gallons of water daily, and can reduce flows during the low flow period between August and October, which overlaps with early migration of fall Chinook into the lower portion of the river. While the overall percentage of river water diverted is not very high, the timing of these diversions, and associated reductions in streamflow, have important consequences for salmonids. CDFW is currently working with Humboldt Bay Municipal Water District to ensure more water can be available for migrating and rearing fish during September and October if there is not much rain during this critical portion of the season (M. Sparkman, CDFW, pers. comm. 2017).

Despite the challenges posed by dams, reservoir operations that capture and store coldwater pool in critically dry years (2013-2015) can benefit salmonids downstream when released strategically (J. Fuller, NMFS, pers. comm. 2017). Scott Dam and the Potter Valley Project, which cut off between 291-463 km for steelhead and 89-127 km for Chinook salmon (Cooper et al., 2017, *in progress*), limit streamflows during certain times of the year, and reduces water in the Eel River through diversions to the Russian River.

Water quality and quantity is reduced by operation of Scott Dam (Eel River) and Coyote Valley Dam (Lake Mendocino, Russian River), which were near record-low levels from 2013-2015. Low storage levels led to reduced summer flows with high temperatures (> 20ºC) and turbidities downstream of the dams. In particular, a chronic challenge with managing coldwater pool and reducing the high turbidity flows exists at Coyote Valley Dam, to the detriment of salmonids in the Russian River (J. Fuller, NMFS, pers. comm. 2017). Low releases from these dams can reduce diversity of benthic invertebrates, an important food of juvenile Chinook and degrade spawning and rearing habitat for both salmon and steelhead, and delay juvenile Chinook out-migration (NMFS 2016). Low releases can have negative impacts on salmonids in the mainstem Russian and Eel rivers as well, especially during fall and early winter months (J. Fuller, NMFS, pers. comm. 2017). In addition, illegal diversions for marijuana cultivation have increased exponentially significantly threatening populations of CC Chinook, especially in Humboldt and Mendocino counties, through altering or dewatering creeks, use of pesticides, poisons, and fertilizers that degrade water quantity and quality (Bauer et al. 2015, NMFS 2016).

Urbanization. Urbanization presents multiple problems for CC Chinook in many parts of the ESU, especially in the lower portions of watersheds where they are most likely to spawn. Water quality is often degraded by urban pollution and runoff. The use of land around rivers and creeks for towns and farms has led to channelization, construction of levees, removal of instream habitat, and channel erosion. Increasing urbanization and other development through the southern portion of the ESU is straining the capacity for water agencies to meet municipal needs and this is likely to further increase water withdrawals and negatively impact CC Chinook.

While there are still issues to be resolved with parasites and toxic algae in the Eel River (Bouma-Gregson and Higgins, 2015), recent emphasis by the State Water Resources Control Board and the Environmental Protection Agency on water quality standards has generally improved water quality throughout the ESU since the last NMFS status update was completed in 2011. Development and implementation of total maximum daily load (TMDLs) limits on pathogens, heavy metals, salts, nutrients, turbidity, and temperature have protected clean water for wildlife (NMFS 2016), with presumably positive, if unmeasured, impacts on salmonids.

Agriculture. Likewise, many tributaries are facing increasingly frequent water withdrawals to expand and irrigate vineyards, marijuana, and other crops and to provide frost protection for grape vines, especially in Sonoma and Mendocino counties. Sonoma County's recently adopted Vineyard Erosion and Sediment Control Ordinance (VESCO) in 2012 will seek to control sediment discharge into streams and minimize potential erosion, but stops short of analyzing expanding vineyard operation impacts on future water diversions (NMFS 2016).

Ironically, flows diverted from the Eel River, via the Potter Valley Project, and stored in Mendocino Reservoir (Coyote Valley Dam) increase summer flows in the Russian River and may be responsible for improving consistent spawning flows for salmon in the river. Pacific Gas and Electric's Potter Valley Project implemented spring block water releases in 2012, 2014, and 2015 to encourage emigration of juvenile Chinook and benefit rearing steelhead with improved habitat accessibility (NMFS 2016). In addition, the Russian River watershed recently became NMFS's first national Habitat Focus Area to help improve water management by decreasing withdrawals for irrigation and by creating conditions that support juvenile salmonid use of the estuary (https://www.habitatblueprint.noaa.gov/habitat-focus-areas/russian-river-california/).

Logging. CC Chinook require intact and interacting riparian, freshwater and estuarine ecosystems to support critical growth during the freshwater and estuarine portions of their life cycle. Historical and current land use practices related to logging and its associated road construction continue to increase the vulnerability of CC Chinook to extirpation within all watersheds in this ESU, but especially in the smaller watersheds. In general, populations in such watersheds are imperiled due to reduction of spawning, incubation, and rearing habitats, mainly resulting from sedimentation. The biggest blows to their habitats occurred in 1955 and 1964, when record rainfall acting on hillsides denuded by years of logging, grazing, and road building caused large-scale erosion as huge, 1,000-year floods ripped through the basins (Yoshiyama and Moyle 2010). "The result was massive landslides, which filled streambeds and pools with loose gravels throughout the drainages. Enormous flows greatly widened stream channels and eliminated most riparian vegetation. Habitat for anadromous salmonids was greatly reduced when sections of stream subsequently became too warm and shallow for juveniles during the summer (Moyle 2002, p. 57)." See discussion of logging effects in the SONCC coho salmon account for a more complete historical perspective of the destructive 1964 flood that largely filled in the South Fork Trinity River (A. Hill, CDFW, pers. comm. 2017).

Continued erosion from abandoned logging area and increases in rural residential roads have created chronic sediment loads far above natural levels. This causes coarse substrate to become imbedded in fine sediment, which makes redd construction by spawning Chinook difficult and creates conditions unfavorable for embryo survival (Opperman et al. 2005). Large amounts of sediment reduce oxygen and metabolite exchange within redds and may entomb embryos. Sedimentation and loss of riparian tree cover (from floods, logging, and other factors) in combination reduce stream habitat complexity, simplifying aquatic food webs and reducing food for juvenile salmonids. Increased sediment has also been shown to reduce juvenile survival by altering feeding success through increased turbidity, reducing prey visibility, and irritation of gills. These factors can also create widened, shallow channels, in which existing high temperatures can be exacerbated and depths too shallow to support Chinook salmon juveniles. Finally, legacy impacts of logging have led to lack of large woody debris in streams that serves as important cover for all life stages of salmonids and high sediment loads from yarding practices that will take decades to recover (NMFS 2016).

Two major landholders, Humboldt Redwoods Company and Green Diamond Resource Company, are implementing habitat conservation plans to benefit salmonids and other species on their properties. Such activities include decommissioning priority road sites, riparian protection, replacing fish barriers at road crossings, replanting exposed soil, fisheries monitoring, adaptive management, and reporting turbidity and temperature data to improve conditions for salmonids

(NMFS 2016). The rapid implementation and expansion of such practices across all holdings of these two major companies has the potential to significantly benefit salmonids and other species.

Estuarine alteration. Estuaries, bays, and lagoons are increasingly being recognized as critical rearing habitats for various salmonids, especially those found in Humboldt Bay (Humboldt Co.) (Wallace et al. 2015), and Redwood Creek (Humboldt Co.). Numerous lagoons form at the mouth of rivers and creeks in this ESU when summer flows become too low to wash out mouth bars. While the timing of large flow events to open sandbars may be more important, this factor is exacerbated by upstream diversions. Lagoons become marginal habitat for juvenile Chinook salmon through the cumulative effects of sedimentation, habitat degradation, and poor water quality. CC Chinook juveniles presumably were once able to over-summer in these habitats. The Mattole River Estuary is the most obvious example of this and conditions in the estuary seem to increase mortality of CC Chinook and steelhead smolts that enter the estuary after its mouth has closed (Mattole Salmon Group 2016). In addition, once productive estuarine marsh habitats have been drained and diked for pasture, greatly reducing habitat available for rearing of juveniles, such as in the Eel River estuary. Redwood Creek, tributaries to Humboldt Bay, and the Eel River all have lost this estuarine complexity, contributing to the decline of salmon populations (Coastal Watershed Planning and Assessment Program 2017).

Mining. Gravel mining still continues today in the Mad, Eel, Van Duzen, and Russian rivers and in Redwood Creek. These operations have been increasingly regulated to minimize impacts in the mainstems of these rivers. The removal of coarse sediment may be beneficial to reduce impacts from increased bedload movement resulting from harmful upstream land practices, but if improperly undertaken, mining can create barriers to migration, increase spawning in channel areas that lack necessary flows for incubation, and decrease water quality from pollution and sedimentation. Gravel mining also creates seasonal barriers during critical migratory periods and can cause stranding of adult Chinook trying to enter tributaries.

Harvest. While CC Chinook are listed as threatened under the Endangered Species Act, they are the target of significant fishing pressure at sea as they become mixed with Chinook from other ESUs, including hatchery fish from the Klamath-Trinity system. NMFS (2016) admits that the effect of the fishery on CC Chinook populations is not known, but estimates that exploitation rates of CC Chinook are low following implementation of Pacific Fishery Management Council regulations targeted to protect large Klamath adults and undersized (< 56cm) sub-adults meant to reduce incidental harvest of CC Chinook and listed Central Valley winter- and spring-run Chinook. Despite these measures, in the absence of reliable sampling that can determine origins of Chinook caught at sea, ocean harvest rates of CC Chinook in some years may be underestimated (J. Fuller, NMFS, pers. comm. 2017). Some mortality undoubtedly occurs on CC Chinook during catch and release at sea; mortality estimates range from about 12-42% depending on methods and location. However, mortality is difficult to quantify because unmarked fish can belong to either the Klamath or CC Chinook ESUs (NMFS 2016).

In fresh water, a catch and release fishery is allowed for CC Chinook by CDFW in the Eel River. This fishery may cause some level of post-release mortality, especially if adults are targeted during periods of low flow (NMFS 2016). In order to combat this, NMFS and CDFW have worked together to impose low-flow fishing closures on many coastal rivers throughout Humboldt, Mendocino, and Sonoma counties under Title 14, Section 8, if stream flows are inadequate for salmonids to passively move upstream (NMFS 2016). The closure was due in part to protect fish that were exhibiting stressed behavioral responses due to low flow conditions and were forced to hold in sub-optimal reaches for extended periods of time (J. Fuller, NMFS, pers.

comm. 2016). Poaching, especially under such low flow conditions, has been documented to be a major concern in the Mad, Eel, Garcia, and Russian river watersheds and angler outreach campaigns have been enacted to raise awareness and increase compliance when water levels are low, especially during summer and fall months during drought (e.g., 2012-2016).

Alien species. Predatory alien fish species are significant problems mainly in the Eel and Russian river drainages. In the Eel River, Sacramento pikeminnow (*Ptychocheilus grandis*) were introduced illegally in 1979 and spread throughout much of the watershed (Brown and Moyle 1997). Their populations have fluctuated over time but it is highly likely that they are suppressing Chinook salmon populations through predation on emigrating juveniles (NMFS 2016); this impact is compounded by other factors discussed above. Sacramento pikeminnow are native to the Russian River, but are not as abundant as they are in the Eel River. Salmonids also face predation from abundant smallmouth bass (*Micropterus dolomieu*) in the Russian River. The effect of these predators is not known, but likely negative (J. Fuller, NMFS, pers. comm. 2017). Striped bass (*Morone saxatilis*) also occur in the lower Russian River at times, and may consume juvenile salmonids (NMFS 2016). However, over the past decade, the abundance of striped bass has declined to the point that recreational anglers that used to target them in the lower river no longer do (J. Fuller, NMFS, pers. comm. 2017).

Hatcheries. Local groups have operated small-scale hatcheries on Freshwater Creek (Humboldt Fish Action Council), Yager Creek (Pacific Lumber Company), Redwood Creek (S. Fork Eel River; Eel River Salmon Restoration Program), Hollow Tree (Salmon Restoration Association), and the Mattole River (Mattole Salmon Group), and the Mad River Hatchery (CDFW). These efforts have been shuttered since 2007 (NMFS 2016) in part because of concern over their negative impacts on the ESU and low returns. It appears that such hatcheries did not increase returns of adult CC Chinook; they may even have increased risks of extirpation in some watersheds when adult Chinook were being removed for hatchery use but returns of fish of hatchery origin were below replacement (Good et al. 2005; NMFS 2007). Williams and colleagues (2011) found that hatchery-origin Chinook Salmon straying into the Russian River were equally likely to be from the Central Valley fall-run as from other CC Chinook watersheds, indicating the CC Chinook are susceptible to competition from hatchery fish of diverse origins.

It is possible that a conservation hatchery could be used for a short time to bolster CC Chinook populations in small coastal rivers in Mendocino County, which could speed local recovery of the ESU, provided major habitat restoration programs were taking place at the same time. In addition, net pens have been proposed for use off of the Humboldt/ Mendocino coast to acclimate juveniles and increase survival to adulthood (J. Fuller, NMFS, pers. comm. 2017). However, there are major concerns with such an approach, because net pen usage tends to significantly increase straying rates to other watersheds, and may concentrate parasites.

Table 2. Major anthropogenic factors limiting, or potentially limiting, viability of populations of California Coastal Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	Medium	Dams on the Eel and Russian rivers reduce spawning, rearing, and migration habitat and flows.
Agriculture	Medium	Irrigation diversions in tributary streams reduce flows, especially from illegal diversions for marijuana cultivation.
Grazing	Medium	Chronic stream bank alteration in many areas.
Rural /residential development	Low	Rural and residential development is expanding in the region as large landholders continue to subdivide and sell parcels.
Urbanization	Medium	Mad and Russian rivers provide water for urban use. Urban areas of Sonoma and Mendocino counties have altered water quality in lower Russian River. Not a problem on Eel.
Instream mining	Low	Gravel mining in Russian and Eel rivers may reduce habitat for juvenile salmon.
Mining	Low	Hardrock mining limited.
Transportation	Medium	Roads create sediment and erosion (see logging).
Logging	Medium	Legacy effects and road-building create impacts.
Fire	Low	Causes siltation of streams, some loss of shade to cool water.
Estuary alteration	Medium	Estuaries highly altered with reduced rearing habitat and connectivity among habitats.
Recreation	Low	Boating, rafting, swimming, fishing likely have little impact.
Harvest	Medium	Subject to mixed stock commercial fishery at sea, which includes hatchery fish, though harvest levels closely managed.
Hatcheries	Low	Seven hatcheries once operated in the ESU range but have all been closed since 2007; possibility of a conservation hatchery for Mendocino Coast populations. Net pen proposals offshore may have major consequences for Chinook salmon if approved.
Alien species	Low	Sacramento pikeminnow in Eel are assumed to be a problem for juvenile Chinook; basses (*Micropterus and Morone spp*.) may pose predation threat to juveniles in Russian River.

Effects of Climate Change: Climate change in Northern California is predicted to result in warmer temperatures, diminished snowpack, more variable precipitation, increased ocean acidity, sea level rise, altered estuary dynamics, and altered marine and freshwater food webs; these factors together will cause reductions in salmonid distribution, growth, behavior, and survival (Williams et al. 2016).

CC Chinook were rated as "highly vulnerable" (score of 18/30) to climate change by Moyle et al. (2012), suggesting under the right circumstances at least some of the population can adapt to or find refuge from climate change. The biggest challenge facing this ESU may be adjusting to changes in flow timing and variability, which are functions of reservoir management in some streams that seek to provide water deliveries, reliable streamflow, and coldwater pool to support fall migrations of adult salmonids (J. Fuller, NMFS, pers. comm. 2017). In the majority of CC Chinook watersheds, natural flows unimpeded by dams are still the major requirement for embryo and juvenile survival. Without sufficient early fall storms, Chinook often will spawn in the accessible lower portion of a river's mainstem, rather than higher upstream. Thus, the relationship between timing of storms/flows and spawner migration timing is critical, and large storms following small ones can lead to significant loss of spawning productivity. This is believed to have occurred in Mattole River and Redwood Creek in recent years. In these locations, low counts of outmigrating juveniles despite high spawner abundance estimates have followed dry fall seasons, when flows needed to allow adults to reach spawning areas in middle and upper watersheds did not occur. Rapid declines in flows during spring may also strand juveniles in reaches where water becomes too warm for over-summering. Logging, urbanization, agriculture, and other factors also increase the amount and magnitude of run-off from storms, increasing their potential for negative effects on Chinook redds and juveniles.

Under the most likely future climate change scenarios for California, variability in timing and amount of precipitation is likely to increase, leading to more common and prolonged drought for much of the state. The historic "hot drought" in California (2012-2016) saw well below average precipitation every year, coupled with record high temperatures in three of the four years (NMFS 2016). This drought was ranked as the worst in perhaps 1,000 years in the state (Williams et al. 2016), and led to record low snowpack in 2015, anomalously high sea surface temperatures, and an over-reliance on groundwater pumping and illegal stream diversions to make up the deficit, robbing watersheds with important flows for salmonids during the warm summer and fall months. Future drought is an especially critical concern for CC Chinook salmon because the coastal watersheds they rely on are fed primarily by rain and not snowmelt, as the upper and Middle Forks of the Eel are, and the periods of lowest summer baseflows coincide with the time of greatest demand for irrigation water in the region (Williams et al. 2016). Further, both freshwater and saltwater survival are generally found to be lower across almost all salmon and steelhead populations on the West Coast during warmer years, suggesting populations may have declined during the ongoing drought (Williams et al. 2016). In fact, poor ocean survival has been identified as one of several drivers (along with reservoir operations and poor habitat quality during critically low water years) of decline in salmon abundance in California over the last decade (NMFS 2016). However, reductions in survival for juveniles during this time will only be confirmed in returning numbers of adults in the following 3-4 years (one generation).

Status Score = 2.9 out of 5.0. High Concern. CC Chinook are vulnerable to extinction in next 50 years, especially if climate change strongly affects both stream and ocean conditions. Katz et al. (2012) scored the status as 2.4/5.0 (vulnerable). The CC Chinook Salmon ESU was initially listed as threatened under the federal Endangered Species Act on September 16, 1999, but this was rescinded in 2002, due to the court case *Alsea Valley Alliance v. Evans*. In this action, the U.S. District Court in Eugene, Oregon, set aside the 1999 listing due to its exclusion of hatchery fish. A status review of the CC Chinook ESU and 15 additional ESUs was completed in 2005 (Good et al. 2005), and the CC Chinook ESU was again listed as threatened on June 28, 2005. The 2010 review concluded the status had not changed but that endangered status was likely in the future (NMFS 2012). Over the last five years, poor ocean conditions, drought (2012-2016, and exploding marijuana cultivation practices throughout the ESU range have had significant negative impacts on the CC Chinook ESU (NMFS 2016), and they remain listed as threatened.

Historic runs of Chinook salmon in the Eel River probably ranged between 100,000 and 800,000 fish per year, declining to roughly 1,000 fish per year in the 1990s and 2000s (Yoshiyama and Moyle 2010). Higgins (2015) noted an apparent increase in Chinook numbers spawning in the Eel in 2012-15 to as many as 25,000 fish, but this may not reflect a permanent upward trajectory. Overall, the CC Chinook have likely suffered declines in excess of 90% in the past 100 years. Whether or not the decline is continuing or abating is equivocal because of inadequate surveys in all rivers except Redwood Creek (NMFS 2016).

Table 3. Metrics for determining the status of California Coastal Chinook salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	3	ESU occupies 5 major watersheds.
Estimated adult abundance	3	All populations are under 1,000 spawners in most years but some mixing among populations; there have been recent increases in the Eel River population, but reliable data are lacking.
Intervention dependence	3	Long-term declines indicate intervention needed, especially in improved flows and habitat in the Russian and Eel rivers.
Tolerance	3	Fall-run life history allows for moderate tolerance of environmental conditions encountered.
Genetic risk	3	Major watersheds may have distinct populations, all threatened by small size and similar genetic issues. The loss of spring-run life history strategy was a major loss of diversity within the ESU.
Climate change	2	Likely to accelerate declines, especially in reservoir-dominated systems with reduced flows and altered channels.
Anthropogenic effects	3	8 Medium threats.
Average	2.9	20/7.
Certainty (1-4)	3	Fairly well studied.

Management Recommendations: CC Chinook are one of the few non-hatchery dependent ESUs of Chinook salmon remaining on the California coast, so management efforts to increase numbers need to focus on habitat/watershed restoration on a large scale and on marking all hatchery fish in other ESUs so they can be differentiated. The NMFS Coastal Multispecies Recovery Plan for the CC Chinook ESU assesses the biology, threats, and conservation considerations that will be part of such a recovery strategy for the ESU (NMFS 2016b). Included in this recovery plan are an estimated 2,630 km of stream habitat and 65 square km of estuarine habitats, which were designated as critical habitat on September 2, 2005. However, the designation has not improved returns of adult CC Chinook in most rivers. Considerable efforts to preserve and restore spawning and rearing habitat, fish passage, water conservation, monitoring, and outreach to local groups have been undertaken over the past two decades, to the tune of $250 million spanning 3,500 restoration projects through the California Fisheries Restoration Grant

Program (FRGP) (http://www.dfg.ca.gov/fish/Administration/Grants/FRGP/FundSummary.asp). Under this program, coordinated projects have tackled fish passage, water conservation, improving instream habitats, watershed monitoring, education and organizational support to watershed groups.

Pressing water quantity and quality issues need to be resolved in most of the ESU's basins to protect and restore habitat for salmonids. A balance of water sharing between the Russian and Eel rivers will influence the abundance of CC Chinook in these basins and is integral to recovery of both populations. While it appears that Chinook are able to exist within the historical and current hydrograph of the Russian River (Chase et al. 2007) and the lower Eel River, recovery of CC Chinook in the upper mainstem Eel River may benefit from restoration of the original hydrograph, which has been altered by operation of Scott and Van Arsdale dams. The Eel River likely supported multiple subpopulations of CC Chinook, but ecological changes in the Eel's mainstem now seem to favor species such as Sacramento pikeminnow.

The recently released NMFS Coastal Multispecies Recovery Plan (NMFS 2016b) lays out general goals to aid in the recovery of the CC Chinook ESU and de-listing from ESA: 1) Reduce the destruction, modification, or curtailment of habitat or range; 2) Ameliorate utilization for commercial, recreational, scientific, or educational purposes; 3) Abate disease and predation; 4) Adequately employ existing regulatory mechanisms for protecting CC Chinook salmon; 5) Ensure the status of CC Chinook salmon is at a low risk of extinction based on abundance, growth rate, spatial structure and diversity. A conservation strategy, drawing from the updated recovery plan (NMFS 2016b) for CC Chinook salmon, should:

- Develop a strategic land acquisition program to protect spawning habitats throughout the ESU. This should focus holistically on watersheds because sedimentation can only be ameliorated through watershed-wide reductions and groundwater pumping far from riparian habitats negatively impacts summer baseflows.
- Mark all hatchery salmon from outside the ESU that enter the mixed stock fishery off the California coast and/or set up a genetic/genomic recognition program to determine the origins of salmon in the fishery, such as Genetic Stock Identification (GSI) monitoring of all Pacific salmon to determine origins of and mortality on Chinook in ocean fisheries for management. As a starting point, the Eel and Russian rivers could stand in as indicator watersheds for the CC Chinook ESU as a whole and help evaluate exploitation rates based on returns to these systems, in combination with GSI monitoring.
- Develop and implement an extensive monitoring program for the entire Eel watershed to monitor recovery progress and adaptively manage restoration projects.
- Restore estuarine marshes and floodplains and improve lower river riparian corridors to increase juvenile-to-smolt survival. This is particularly important for the Eel River, Redwood Creek, and other rivers with historically extensive tidal and lagoon habitats.
- Establish a managed flow regime, similar to the historical hydrograph in volume and timing, for the Eel River below Scott and Van Arsdale dams and the Russian River below Coyote Valley dam to provide necessary migration of Chinook into upper portions of spawning habitat, and for juveniles to successfully migrate to sea. The entire operation of the water system that diverts Eel River water into the Russian River (Potter Valley Project), including the presence and operation of Van Arsdale and Scott dams, needs to be carefully evaluated to develop conservation strategies that prioritize reliable flows of high quality water at appropriate times to benefit CC Chinook Salmon in both rivers.
- Improve operation of Ruth Dam on the Mad River to benefit all anadromous fishes.

- Increase water allocated from Mendocino and Sonoma reservoirs for fish in the Russian River, in conjunction with reducing flows from the Potter Valley Project.
- Improve agricultural and forestry practices to reduce sedimentation, improve water quality, increase stream habitat complexity, and increase flows. Current logging harvest rates reduce viability of CC Chinook in multiple watersheds. Of particular importance is reducing amounts of water diverted for irrigation (or pumped from wells adjacent to streams) in small tributaries of regulated rivers and throughout the watersheds of undammed rivers (e.g., South Fork Eel).
- Conduct annual monitoring of spawner abundance and juvenile and smolt abundance for all major, remnant populations within the ESU.
- Promote municipal, industrial, agricultural outreach programs that conserve water, reduce pollution, and create awareness about CC Chinook as an indicator of healthy waters.
- Develop a Fishery Management and Evaluation Plan (FMEP) that will allow legal catch-and-release CC Chinook fishing in the Eel River (NMFS 2016), covers Tribal fishing rights to allow take of ESA-listed fish. As part of such a plan, consider closure of the mouths of CC Chinook rivers during certain portions of the commercial ocean fishery.
- Promote and expand application of the State Water Resources Control Board Policy for Maintaining Instream Flows that includes principles for water right administration and conservation from the Mattole River south to San Francisco Bay (NMFS 2016).
- Supply adequate funding for implementing and expanding use of the California Coastal Monitoring Program beyond the Russian and Eel watersheds to determine changes in population structure and dynamics.
- Address illegal diversions in all watersheds but especially the Eel and Russian rivers.
- Enforce special fishing regulations to protect Chinook, coho, and steelhead, especially in the Eel, Garcia, and Russian Rivers.
- Enforce Assembly Bill 2121 (1259.2 and 1259.4 in the California Water Code) to ensure protective flows remain instream for salmonids (NMFS 2016).
- Explore the feasibility of strategically using a temporary broodstock conservation hatchery along the Mendocino Coast to establish and bolster populations of CC Chinook in the short-term.

CENTRAL VALLEY FALL-RUN CHINOOK SALMON
Oncorhynchus tshawytscha

High Concern. Status Score = 2.7 out of 5.0. The number of spawners typically exceeds 100,000 fish each year but the run is largely supported by hatchery production.

Description: Members of the Central Valley (CV) fall-run Chinook salmon Evolutionary Significant Unit (ESU), like other Chinook salmon, have numerous small black spots on the back, dorsal fin, and both lobes of the tail in both sexes. This spotting on the caudal fin and the black coloration of their lower jaw make them distinguishable from other sympatric salmonid species. They have 10-14 major dorsal fin rays, 14-19 anal fin rays, 14-19 pectoral fins rays, and 10-11 pelvic fin rays. There are 130-165 scales along the lateral line. Branchiostegal rays number 13-19. They possess more than 100 pyloric caeca and have rough and widely spaced gill rakers, 6-10 on the lower half of the first gill arch.

Spawning adults are the largest Pacific salmonid, typically 75-80 cm SL, but lengths may exceed 140 cm. In California, Chinook are usually smaller, typically 45-60 cm SL. The average weight is 9-10 kg, although the largest Chinook salmon taken in California was 38.6 kg. Spawning adults are olive brown to dark maroon without streaking or blotches on the side. Males are often darker than females and develop a hooked jaw and slightly humped back during spawning. Juveniles have 6-12 parr marks, which often extend below the lateral line, and the marks are typically equal to or wider than the spaces between them. Parr can also be distinguished from other salmon species by the adipose fin, which is pigmented on the upper edge, but clear at the base and center. Some parr begin to show spots on the dorsal fin, but most fins are clear. There are no morphological features to separate this ESU from other Chinook salmon ESUs, so separation is based on genetics and life history characteristics, especially adult freshwater entry timing.

Taxonomic Relationships: Central Valley fall-run Chinook salmon are part of the CV complex consisting of four Chinook salmon runs differentiated by genetic differences, timing of spawning migrations, maturity of fish entering freshwater, spawning location, incubation duration, and out-migration timing of juveniles (Moyle 2002). The seasonal runs of CV Chinook salmon (winter, spring, fall and late fall) are more closely related to each other than they are to populations outside the CV (Williams 2006).

Winter- and spring-run Chinook are recognized as distinct ESUs, while the National Marine Fisheries Service groups the fall-run and late fall-run in a single ESU. This report differs from that taxonomy in that we regard the late-fall run to be a distinct taxon with a unique life-history strategy and specific management concerns. While the four runs of CV Chinook salmon were historically genetically distinct from each other, decades of interbreeding between fall and spring run in the Feather River below Oroville Dam and in Feather River Hatchery has produced Chinook salmon that return to the Feather River and tributaries in spring but that are nearly genetically identical to fall-run. CV fall run are the principal salmon raised in CV hatcheries to support fisheries and are released in large numbers. Adults from different hatcheries have a long history of straying to non-natal streams, resulting in a genetically near-uniform population found throughout the CV.

Life History: Chinook salmon life history strategies differ in the timing of their spawning migration, a fact implicit in the naming of the different "runs" according to the season in which they enter rivers from the ocean. However, the season of adult spawning migration is only one of the multiple life history attributes that differ between the runs of CV Chinook salmon (Table 1). Procession from one salmon life stage to the next is often characterized by movement between habitat types. This account focuses on life history and migratory characteristics specific to the CV fall-run which have a life history that minimizes time spent in fresh water. Fry and smolts out-migrate in spring before water temperatures become too warm, allowing fall-run to exploit the extensive lower elevation reaches of Central Valley rivers and streams, where temperatures exceed thermal tolerances during late spring, summer and early fall. In contrast, the other three CV Chinook salmon runs require cool, riverine habitats year-round.

Adult CV fall-run Chinook salmon begin entering fresh water in late summer and early fall as mature individuals and move relatively quickly to spawning grounds. Spawning usually occurs within several weeks to two months of freshwater entry. Peak spawning time is typically in October-November, but can continue through December and into January. Juveniles typically emerge from the gravel in December through March and rear in fresh water for 1-7 months, usually moving downstream into large rivers within a few weeks (Moyle 2002). Smolts tend to initiate migration during storm events and flow is positively correlated with migration rate (McCormick et al. 1998, Michel et al. 2013). In the clear upper reaches of the Sacramento River, out-migrating smolts employ a nocturnal migration strategy, a behavior likely influenced by decreased predation under cover of darkness. Likewise, reduced water clarity has a strong positive relationship with increased survival during out-migration. Presumably this is the result of the strong association between high turbidity and large flow events, which in combination reduce predation efficiency (Michel et al. 2013) and move fish rapidly downstream to floodplains and other rearing areas.

In the past, CV fall-run juveniles likely reared on formerly extensive valley floor floodplains. Juvenile fish foraging in the few remaining highly productive floodplain habitats grow much more quickly than those in major river channels (Sommer et al. 2001, Jeffres et al. 2008). Historically, this rapid growth before ocean entry likely increased survival of fall-run juveniles, which enter the ocean at relatively small size and young age compared to juveniles of other CV Chinook runs.

The slowest movement rates are observed in the estuary (Michel et al. 2013). From the estuary, juveniles move through the Golden Gate into the Gulf of the Farallones, which in late spring and early summer is typically an extremely food-rich region because of wind-driven upwelling associated with the California Current. Immature fish spend 2-5 years at sea, where they feed on fish and shrimp before returning as adults. Most of the fish remain off the California coast between Point Sur and Point Arena during this period, but many move into coastal waters of Oregon as well. Their movements in the ocean during this rearing period are poorly understood. Inshore, offshore and along-shore movements are likely in response to changing temperatures and upwelling strength. The vast majority of fall-run adults returning to the Central Valley and harvested in commercial and sport fisheries are 3 years old (Palmer-Zwahlen and Kormos 2015).

There are many exceptions to this general life cycle, including juveniles that spend as long as one year in fresh water. This yearling life history is likely supported by the novel habitat conditions found in the tailwater stream reaches below Central Valley rim dams where fall-run experience year-round cool temperatures at relatively low elevations. Essentially, dams have brought the perennially cool water conditions found in mountain canyons down to the valley floor. In general, the attributes of fall-run Chinook salmon that have allowed them to adapt to low-elevation regulated rivers have also made them the preferred run for hatchery production. Mature fish can be spawned as they arrive and juveniles only need to rear for a short time before being released into the rivers.

Table 1. Generalized life history timing of Central Valley Chinook salmon complex. Data from Yoshiyama et al. 1998.

	Migration period	*Peak migration*	*Spawning period*	*Peak spawning*	*Juvenile emergence period*	*Juvenile stream residency*
Sacramento River basin						
Late fall run	October–April	December	Early January–April	February–March	April–June	7–13 months
Winter run	December–July	March	Late April–early August	May–June	July–October	5–10 months
Spring run	March–September	May–June	Late August–October	Mid-September	November–March	3–15 months
Fall run	June–December	September–October	Late September–December	October–November	December–March	1–7 months

Habitat Requirements: The general habitat requirements of CV fall-run are similar to those of other Chinook salmon that minimize their time in fresh water (Healey 1991, Moyle 2002).

Adult migration: Fall run enter fresh water as mature fish with ripe gonads; they tend to migrate fairly directly to spawning grounds without holding for appreciable amounts of time, although low water conditions may delay upstream migration.

Spawning and egg incubation: A female salmon makes a redd (an area containing a series of individual nests) by repeatedly turning on her side and flexing her body so she pushes water down against the river bed. This action forces gravel up into the water column where it is carried a short ways downstream by the current. In this manner, the female "digs" an oval depression into the streambed and the current mounds the excavated material immediately downstream (Crisp and Carling 1989). Often, several males will court and fertilize the eggs of a single female, which she deposits in the "pot" of the nest. She will then move upstream and repeat the digging maneuver, removing fine sediment and burying the eggs in clean substrate. Chinook salmon use the largest substrate of any California salmonid for spawning, a mixture of large gravel and small cobble with a median particle diameter up to about 10% of their body length (Kondolf and Wolman 1993). Such coarse material allows sufficient water flow through the substrate to provide oxygen for developing embryos, while simultaneously removing metabolic waste. Redd size is a function of female size and substrate mobility. Redds are typically over 2-15 m^2 in size (Healey 1991) and observed at depths from a few centimeters to several meters and at water velocities of 15-190 cm/sec. The selection of redd sites is often a function of gravel permeability and subsurface water flow. Preferred spawning habitat seems to be at depths of 30-100 cm and at water velocities of 40-60 cm/sec.

By filling interstitial spaces between gravels, fine sediment can reduce water flow through the redd, effectively "smothering" embryos by denying them sufficient dissolved oxygen, causing toxic build-up of metabolic wastes, or physically hindering emergence from the gravel. For maximum embryo survival, water temperatures must be between 5º and 13º C and oxygen levels close to saturation. Incubation time is highly dependent on water temperature, dissolved oxygen (DO), and substrate permeability (Merz et al. 2004). Under optimal conditions, embryos hatch after 40-60 days and remain in the gravel as alevins for another 4-6 weeks, usually until the yolk sac is fully absorbed.

Chinook salmon are semelparous meaning that they spawn once and die, although individuals may survive for weeks after spawning. Especially where available spawning area is limited, late-arriving spawners may dig directly on top of previously constructed redds. Superimposition of redds can be a major mortality factor for incubating embryos and may result in a density-dependent relationship between the abundance of spawners and egg-to-fry survival.

Juvenile rearing and outmigration: Once alevins emerge and become fry, they tend to aggregate along stream edges, seeking cover in vegetation, swirling water, and dark backgrounds. As they grow larger and become increasingly vulnerable to avian predators, especially herons and kingfishers, they move into deeper (> 50 cm) water. Larger juveniles may often use the tails of pools or other moderately fast-flowing habitats, where food is abundant and there is some protection from predators. As juveniles move downstream, they use more open water during night while seeking protected pools during the day. Pools that are cooler than the main river, either from upwelling or tributary inflow, may be preferred by migrating juveniles as daytime refuges. The route by which Sacramento River smolts pass through the Sacramento-San Joaquin rivers Delta (Delta) has a significant effect on survival. Those that migrate through the interior Delta have higher mortality rates than fish remaining in the mainstem Sacramento River (Perry et al. 2010).

Fall-run tend to spend the least amount of time of any of the CV runs in fresh water largely because they must migrate to the ocean before their low elevation habitats get too hot during late spring and early summer. Juvenile use of off-channel habitat for rearing, including floodplains, improves growth prior to ocean entry (Sommer et al. 2001, Limm and Marchetti 2006, Jeffres et al. 2008). Off-channel habitat can also be important in the San Francisco Estuary (e.g., tidal marshes), but these habitats are now largely unavailable because they are cut off by levees. Today, less than 10% of historical CV wetland habitats remain accessible to CV salmon (Frayer et al. 1989, Hanak et al. 2011). The rich aquatic food webs found in off-channel wetland habitats along the valley reaches of rivers and in the Delta and estuary were a major driver of historical abundance.

Ocean habitats used for the first few months are poorly documented, but it is assumed that fish stay in coastal waters where the cold California Current, through upwelling, creates rich food supplies, particularly small shrimp. During the day, juveniles and subadults seem to avoid surface waters. Sub-adult Chinook salmon consume Pacific anchovies (*Engraulis mordax*), juvenile rockfish (*Sebastes spp.*), Pacific herring (*Clupea pallasii*), and other small fishes, typically at depths of 20-40 m and move offshore into deeper waters in response to temperature, food availability, and avoidance of predators.

Distribution: Central Valley fall-run Chinook salmon historically spawned in all major rivers of the CV, migrating as far as the Kings River to the south and the Upper Sacramento, McCloud, and Pit rivers to the north. Today, in the Sacramento and San Joaquin River watersheds, they spawn upstream as far as the first impassible dams. Passage into the mainstem San Joaquin River, above the confluence with the Merced River, is intentionally blocked at the CDFW-operated weir at Hills Ferry. Overall, it is estimated that approximately 60% of fall-run spawning habitat has been blocked by dams (Yoshiyama et al. 2001), although coldwater releases from dams now allow spawning in some places where it did not formerly exist (Yoshiyama et al. 1998). Fall-run Chinook salmon have been impacted less by dam construction than winter and spring-run Chinook salmon, because fall-run historically spawned in lower elevation stream reaches, up to 152-304 m (500–1,000 feet) above sea level (Yoshiyama et al. 2001). Levees block juveniles from accessing most historical floodplain and tidal marsh rearing habitats in the Central Valley and Bay-Delta.

Trends in Abundance: The historical abundance of fall-run Chinook salmon is difficult to estimate because populations declined before extensive monitoring and good record keeping. Yoshiyama et al. (1998) estimates that fall-run Chinook were historically one of the largest runs in the CV, with about a million spawners returning each year (Yoshiyama et al. 1998). Wild populations, however, were affected by a multitude of effects. Hydraulic mining operations during the Gold Rush Era buried spawning and rearing areas under mining debris before the first estimates of salmon population numbers. Likewise, Chinook salmon were extensively harvested in-river during the 19th century and, accurate, detailed records of run and river source were not documented. The exploitation by fisheries and alteration of California rivers during the Gold Rush likely reduced Chinook salmon abundance to about 10% of historical numbers by the early 1900s (Yoshiyama et al. 1998). Construction of large dams throughout the CV in the 1940s-60s further reduced wild Chinook populations. The extent of the impacts on CV Chinook populations is uncertain because artificial propagation also began during this era and no effort was made to differentiate wild Chinook from those produced by hatcheries. Until recent years, escapement estimates for CV fall-run salmon included both hatchery and natural-origin fish, with unknown relative proportions.

Table 2. Chinook salmon thermal tolerances in fresh water. All lethal temperature data are presented as incipient upper lethal temperatures (IULT), which is a better indicator of natural conditions because experimental designs use a slower rate of change (ca. 1°C/day). Information largely from McCullough (1999).

	Sub-Optimal	Optimal	Sub-Optimal	Lethal	Notes
Adult Migration	< 10ºC	10-20°C	20-21°C	>21-24°C	Migration usually stops when temperature climbs above 21ºC, with partial mortality occurring at 22-24ºC. Lethal temperature under most conditions is 24ºC. Fish observed moving at high temperatures are probably moving between cooler refuges.
Adult Holding	< 10ºC	10-16°C	16-21°C	> 21-24°C	Adults can experience heavy mortality above 21ºC under crowded conditions but will survive temperatures up to 24ºC for short periods of time. In some holding areas, maximum temperatures exceed 20ºC for over 50 days in summer.
Adult Spawning	< 13ºC	13-16°C	16-19°C	> 19°C	Egg viability is reduced with exposure to higher temperatures.
Embryo Incubation	< 9°C	9-13°C	13-17°C	> 17°C	This is the most temperature sensitive phase of the life cycle. American River salmon have 100% mortality > 16.7°C; Sac. River salmon mortality exceeded 82% > 13.9°C.
Juvenile Rearing	< 13°C	13-20°C	20-24°C	> 24°C	Past exposure (acclimation temperatures) has a large effect on thermal tolerance. Fish with high acclimation temperatures may survive 28-29ºC for short periods of time. Optimal conditions occur under fluctuating temperatures, with cooler temperatures usually at night. When food is abundant, juveniles may thrive at increased temperatures.
Smoltification	< 10ºC	10-19°C	19-24°C	> 24°C	Smolts may survive and grow at suboptimal temperatures but have a harder time avoiding predators; measured optimal temperatures are 13-17ºC (Marine and Cech 2004), but observations in the wild indicate a greater range.

From 1967 to 1991, an average of 250,000 adult fish returned to spawn with an additional 375,000 harvested each year in the commercial and sport fisheries (USFWS 2011). From 1992 to 2006, average escapement was nearly 400,000 with an annual average of 484,000 harvested by fisheries. In 2007, escapement plummeted to fewer than 100,000 fish with about 121,000 harvested in fisheries, prompting the first-ever closure of the California ocean salmon fishery. Returns dropped to 71,000 in 2008 and, in 2009, escapement reached a record low of 53,000 spawners, even as the ocean fisheries remained closed. Escapement in 2010 increased to 163,000 with a limited ocean fishing season and harvest of 20,400 fish. Central Valley escapement continued to rebound to approximately 228,000 fish in 2011 and 340,000 fish in 2012, and peaked at 447,000 in 2013 before dropping to 256,000 in 2014 and 152,000 in 2015, the last season for which data is available (CDFW GrandTab 2017).

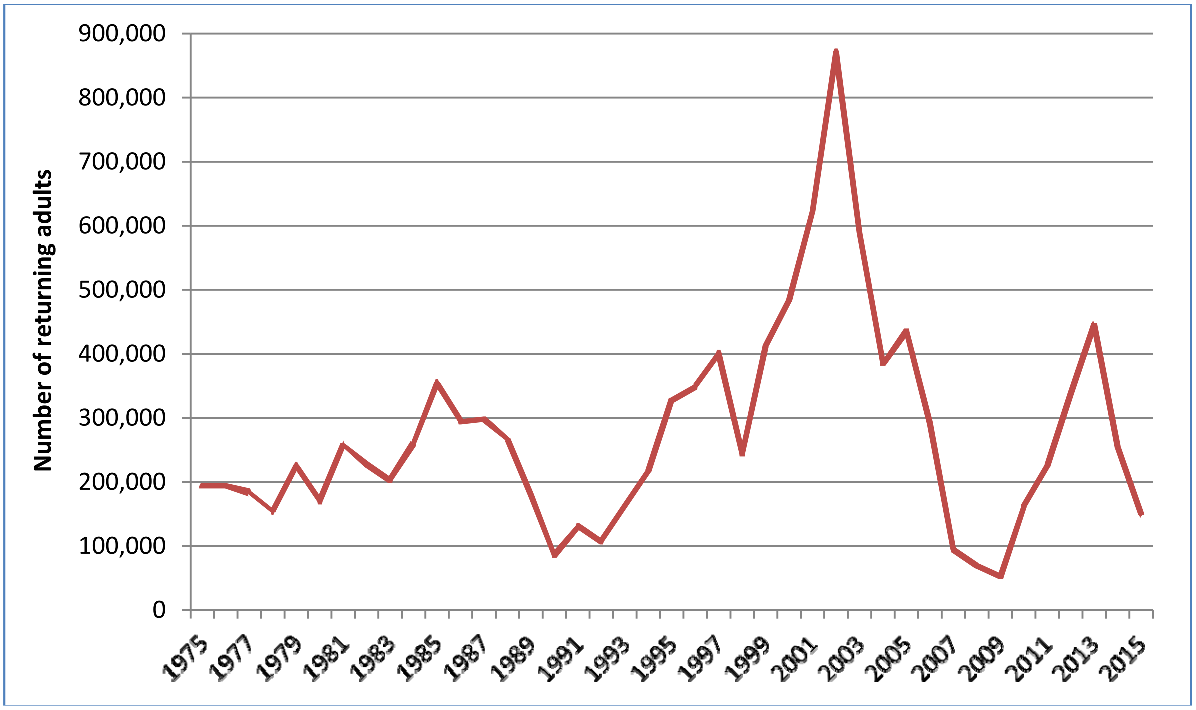


Figure 1. Estimated escapement (hatcheries plus in-river) of adult fall-run Chinook salmon in Central Valley rivers and streams. From: CDFW GrandTab, March 2017.

The effects of hatchery production on abundance and population dynamics of CV fall-run Chinook has been poorly documented, but improvements in tagging and monitoring programs since 2007 are enabling a better analysis of stock composition in the CV. Data from the CV Fractional Marking Program (Kormos et al. 2012, Palmer-Zwahlen and Kormos 2015) and otolith michrochemistry (Barnett-Johnson et al. 2007, Johnson et al. 2012) indicate that the vast majority of fall-run Chinook salmon are of hatchery origin (Figure 2). Stray rates between river basins are variable, but in most cases quite high (Kormos et al. 2012, Palmer-Zwahlen and Kormos 2015). Genetic evidence suggests that CV fall-run Chinook populations are now genetically homogenous and that natural origin and hatchery are fish indistinguishable from each other (Williamson and May 2003, Lindley et a l. 2009). Taken together, the evidence suggests that decades of hatchery production meant to augment natural stocks has instead replaced them.

Factors Affecting Status: Widespread and intensive development of the CV over the last 150 years has simplified river, floodplain, and estuarine habitats, altered ecological processes (i.e., hydrology, sediment transport, nutrient cycling) and fundamentally altered the CV Chinook salmon complex, from a diverse collection of numerous wild populations employing diverse life histories to one dominated by fall-run Chinook salmon produced in four large hatcheries (Lindley et al. 2009). Important factors continuing to threaten the viability of CV fall-run Chinook salmon include:

Dams. Large dams on the Sacramento River and its tributaries have blocked fall-run Chinook salmon access to much of their historical spawning grounds. Habitat downstream of the dams has been greatly altered. Regulated flows and resulting water temperatures are sometimes unsuitable for spawning and rearing. Gravels once washed down from upstream areas are now trapped in reservoirs, reducing spawning substrate below dams. Large quantities of gravel are now trucked to spawning areas below many dams to improve spawning habitat. Many large dams also have flow requirements for salmon spawning, egg incubation, rearing, and juvenile emigration. All restoration actions downstream of dams require regular, human intervention and their effectiveness at the population level is not well documented (Mesick 2001, Wheaton et al. 2004).

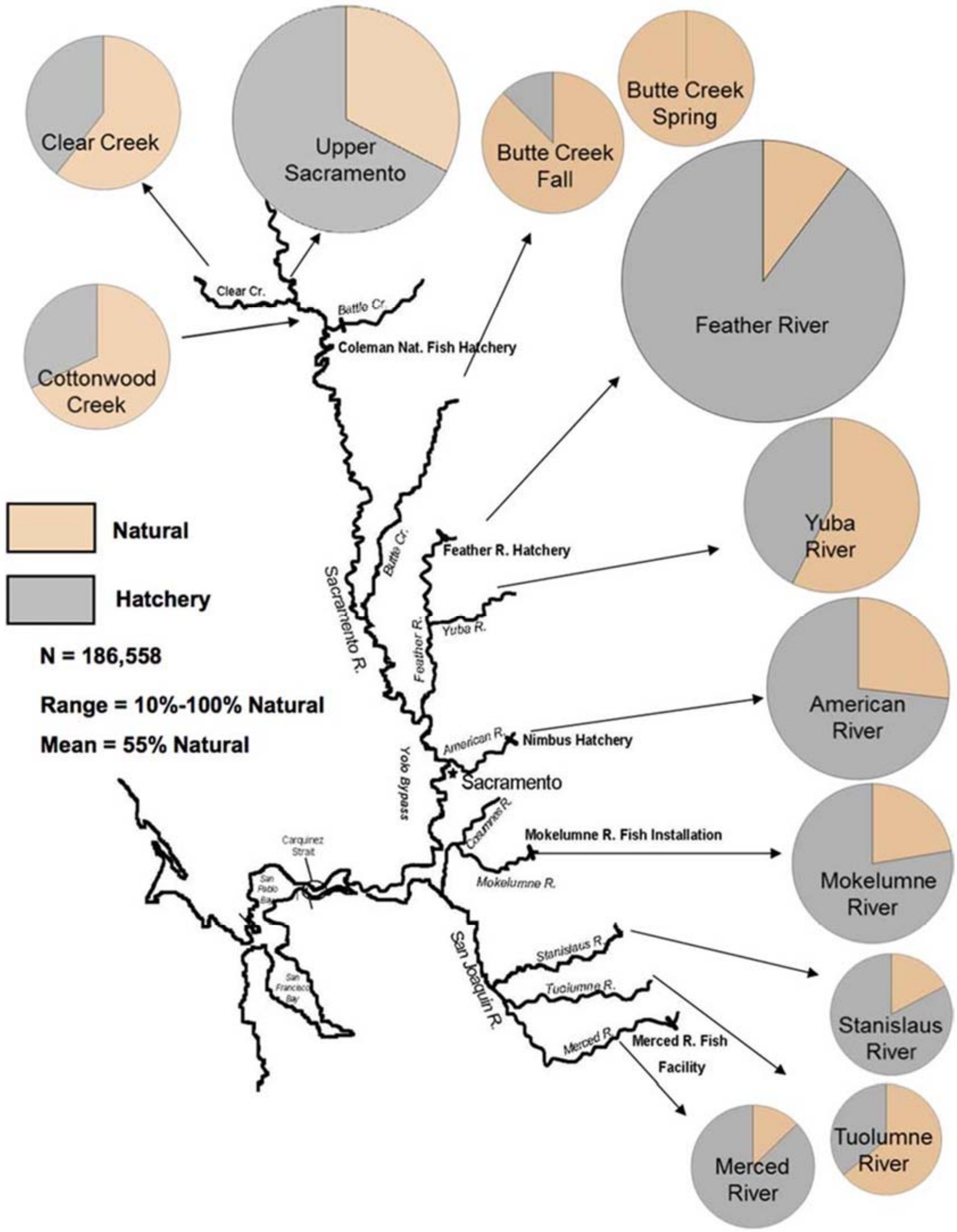


Figure 2. 2012 Fall-run Chinook salmon escapement, hatchery and natural proportions. From: DFW 2015, Fig. 3 pg. 18.

Agriculture. There are a large number of agricultural diversions along the Sacramento and San Joaquin Rivers and their tributaries, as well as in the Delta, which entrain juvenile salmon. Although most large diversions have been screened at considerable effort and expense to prevent entrainment, many small to medium diversions remain unscreened. Moyle and Israel (2005) noted that fish screens on rivers are subject to failure and may create holding areas for salmon predators (e.g., catfishes, striped bass, etc.). They also acknowledged that, despite their numbers, small diversions, even cumulatively, probably do not kill many salmon, unless they are on small tributaries. In general, the higher the proportion of flow taken by an unscreened diversion, the more likely the diversion is to have a negative impact on local salmon populations through entrainment.

The largest diversions in the Central Valley are those of the State Water Project (SWP) and the federal Central Valley Project (CVP) in the south Delta, which export water for both agricultural and urban use. Large pumping plants in the south Delta change the hydrology of the upper channels, resulting in substantial modifications in flow directions (Nichols et al. 1986). Pumping thus increases the likelihood of out-migrating smolts entering the interior Delta where longer routes, impaired water quality, higher predation rates, and entrainment lead to higher mortality rates (Perry et al. 2010). These pumping plants also entrain large numbers of fall-run Chinook salmon (as well as salmon of other runs), especially from San Joaquin River tributaries (Kimmerer 2008). Zeug and Cavallo (2014) found a strong positive correlation between the amount of water diverted and entrainment rates at diversion facilities. The largest diversions in the CV have louver screens that divert salmon to be "salvaged" by capture, trucking, and then released downstream in the Delta. However, both direct and indirect mortality associated with salvage operations is likely high because many stressed, disoriented salmon are eaten by predators immediately after release (Kimmerer 2008). Another cause of direct mortality is the high predation rates in Clifton Court Forebay, from which the SWP pumps water prior to it reaching the SWP salvage facility.

Agriculture also contributes to loss of juvenile habitat by limiting access, via an extensive network of flood protection levees, to the shallow, high-productivity habitats needed for rearing and protection from predators during migration. In addition to reduced access to floodplains, construction of levees to channelize rivers has had multiple effects, including simplifying bank structure through use of rip-rap, removal of large wood which creates cover, and reduction in shade. Recent studies have demonstrated the importance of floodplains for increased juvenile salmon growth and survival (Sommer et al. 2001, Jeffres et al. 2008). Inundation of floodplains increases water residence time, allowing it to warm compared to the relatively cool water in river channels. This in turn facilitates greater decomposition of terrestrial vegetation and increases primary production in the form of algal phytoplankton (Ahearn et al. 2006, Grosholz and Gallo 2006). Detrital decomposition and algal primary production are the primary sources of carbon that fuel aquatic food webs and supports zooplankton and other invertebrate populations, which, in turn, are the primary source of food for juvenile fish. Fish food densities are typically far greater on floodplains than in the river (Jeffres 2016, Corline et al. 2017).

In addition, juvenile salmonids may use less energy to maintain themselves on floodplains than they would in the mainstem Sacramento River, further reducing energy expenditure and increasing growth rates. For these reasons, growth rates for juvenile Chinook rearing in floodplain habitats exceed those rearing in riverine habitats (Sommer et al 2001, Jeffres et al. 2008, Katz et al. *in press*). In turn, larger out-migrants result in higher survival rates in the ocean (Unwin 1997, McCormick et al. 1998, Hayes et al. 2008, Williams et al. 2016). The rapid growth facilitated by inundated floodplain habitat is particularly important for fall-run which enter the lower river and Delta as small fish. However, there are very few floodplains now available to salmonid juveniles, which likely has a profound negative impact on survival and recruitment of naturally-spawned fall-run juveniles produced annually in the Central Valley.

A relatively new threat is the use of pyrethroid pesticides, which are particularly toxic to fish. Although mortality events are periodically recorded, the interacting effects of multiple pollutants on juvenile salmon survival are largely unknown. Even if pollutants are sublethal in concentration, they can stress both adult and juvenile fish, making them more vulnerable to disease, predation, and other stressors (Eder et al. 2008).

Urbanization. Urbanization can simplify habitats and degrade water quality conditions for Chinook salmon. Water diversions, levees (and their maintenance) and channel straightening all contribute to habitat simplification. Juvenile salmon are exposed to toxic materials discharged into rivers from urban and agricultural sources. Of particular concern is the poor water quality observed seasonally in the Stockton Deepwater Ship Channel. The channel serves as an area of concentration of pollutants from agricultural wastewater, discharges from the City of Stockton's sewage treatment facilities, storm drains, and other sources. Low dissolved oxygen levels in the fall have been shown to delay adult fall-run immigration into the San Joaquin basin.

Mining. Historical (and, to a lesser degree, ongoing) gold and gravel mining have dramatically altered many CV streams. Hydraulic and dredge mining in the 19th and early 20th centuries caused major morphological and hydrological changes in many rivers, degrading salmon spawning and rearing habitats. Many of these waterways are still recovering. In the past, Iron Mountain Mine, northwest of Redding, drained highly acidic water laden with heavy metals into the Sacramento River, resulting in acute mortality to Chinook salmon. Although discharge is now highly controlled, failure of the Spring Creek retention reservoir could result in severe impacts to aquatic life in large reaches of the Sacramento River.

Instream mining. Deep gravel pits in a number of rivers (e.g. Tuolumne, Merced, San Joaquin) reduce water velocities and allow for the aggregation of predatory fishes, potentially increasing mortality of juvenile salmon moving downstream.

Estuary alteration. Millions of naturally produced juvenile fall-run Chinook migrate into the estuary every year, but very few of them survive to return as adults (Williams 2012). Historically, juvenile fall-run Chinook salmon probably entered the estuary in February-June, and spent varying amounts of time there, in diverse habitats. However, loss of habitat diversity in the San Francisco Estuary has limited life history diversity and the best strategy for juvenile salmon today seems to be to move through the estuary as quickly as possible.

Despite long-term monitoring, causes of high mortality rates as fish pass through the estuary are poorly understood. General observations suggest that rearing conditions in the estuary are often poor. Juvenile fall run Chinook grow slowly in length (mean 0.33 mm per day) and hardly at all in weight during their passage from Chipps Island to the Golden Gate (MacFarlane 2005). Survival tends to be highest during wet years, when passage through the estuary is more rapid (Brandes and McLain 2001, Baker and Morhardt 2001) and fish are literally "flushed" through the system. Flooding in wet years also increases wetted off channel and floodplain habitats throughout the watershed resulting in greater floodplain contribution to river food webs and increasing rearing habitats in Sutter and Yolo Bypasses and the Delta, which likely has a positive effect on growth and survival.

Harvest. CV fall-run Chinook salmon support commercial and sport fisheries along the

California and Oregon coasts and freshwater sport fisheries in rivers of the Central Valley. Hatchery fish can sustain higher harvest rates than natural origin fish, but not all hatchery fish are visually marked, making it impossible to decipher between the two at the time of capture. It is, therefore, possible that existing recreational fisheries, in spite of being highly regulated and managed, may harvest natural-origin fish at unsustainable rates (Williams 2006). Wild-spawned fish, while a fraction of the overall fall-run, may be of particular importance in maintaining genetic attributes that increase life history diversity and adaptability to localized selection processes, particularly in the face of changing environmental conditions, such as those predicted under climate change (e.g., Hayhoe et al. 2004, Mote et al. 2005).

Fisheries also affect Chinook salmon populations through continual removal of larger and older individuals. This selection results in spawning runs made up primarily of two and three-year-old fish, which are smaller and, therefore, produce fewer eggs per female. The removal of older fish also removes much of the buffering that salmon populations have against natural disasters, such as severe drought, that may eliminate an entire cohort. Under natural conditions, four- and five-year-old fish residing in the ocean help buffer against population declines due to short-term environmental changes. In order to protect the low stock of CV fall-run Chinook salmon, ocean salmon fisheries were greatly restricted in 2006-2010 by the National Marine Fisheries Service and the Pacific Fisheries Management Council (*Congressional Record*, 50 CFR Part 660). The Chinook salmon sport fishery in the Sacramento River system was also restricted during this period. In 2011-2016 ocean and inland fisheries were not limited by low abundance of CV fall-run Chinook. However, the effects of the five-year drought (2011-2016) have likely impacted CV fall-run Chinook and the 2017 commercial ocean season has again been canceled.

Hatcheries. Over 2 billion juvenile salmon have been released from Central Valley hatcheries since 1946 (Huber and Carlson 2015). Analysis of coded wire tags recovered from returning adults indicates that hatchery-produced fish contribute the vast majority of total CV fall-run production (Kormos et al. 2012; Mohr and Satterthwaite 2013; Palmer-Zwahlen and Kormos 2013). Otolith analyses support these findings. Barnett- Johnson et al. (2007) showed that only 6% of 158 fall- run Chinook taken from the Pacific Ocean in 2002 were naturally produced, while Johnson et al. (2012) found that approximately 90% of fall-run Chinook returning to spawn in the Mokelumne River were of hatchery origin.

Increasingly juvenile Chinook salmon from Central Valley hatcheries are being trucked closer to the ocean and released downstream of the Delta in San Pablo and Grizzly bays (CDFW 2014, Huber and Carlson 2015). From 2007 to 2013, 54 percent of all hatchery fish in California were released off-site (PSMFC 2013). During the drought (2012-2016) the proportion of trucking juvenile Chinook increased dramatically. Transporting smolts improves survival, but it also increases straying rates in returning adults (Williams 2012). High straying rates contribute to homogenization of population structure and reductions in fitness by increasing gene flow among populations in different streams, thus reducing fitness of individuals, decreasing the reproductive capacity of populations, and eroding the biocomplexity of the entire CV fall-run Chinook salmon run. The homogenizing influence of hatcheries has made the fall-run more susceptible to adverse conditions, such as drought and corresponding low flows in freshwater habitats, or periods of reduced upwelling in coastal waters (Moyle et al. 2008, Carlson et al. 2011, Satterthwaite and Carlson 2015).

In general, the negative effects of hatchery production on wild stocks can be divided into ecological and genetic impacts, although the two interact considerably. Ecological effects include competition, predation, and disease transfer from hatchery stocks to wild populations

(Allendorf and Ryman 1987). Competition between hatchery and naturally-produced Chinook can reduce abundance (Pearsons and Temple 2010), growth rate (Williams 2006) and survival of wild juveniles in river, estuarine and marine habitats (Nickelson et al. 1986, Levin et al. 2001, Levin and Williams 2002, Nickelson 2003). Hatchery releases can even exceed the carrying capacity of ocean habitats, particularly in times of low ocean productivity (Beamish et al. 1997, Levin et al. 2001), resulting in high ocean mortality (Beamish et al. 1997, Heard 1998, Kaeriyama 2004).

Genetic effects include the loss of genetic diversity that facilitate adaptation to changing habitats by sustaining diverse behavior and life history strategies. Hatchery propagation has narrowed this behavioral variation in hatchery stocks (most fish are released over a short time period), leaving them vulnerable to climatic anomalies (ocean conditions, drought, etc.), and human alterations of the landscape. Hatchery propagation has also resulted in domestication of the stock, favoring a salmon genome that is well adapted to comparatively stable hatchery conditions, but may be less fit under variable natural conditions.

Alien species. For the past 150 years, numerous species have been introduced to the Central Valley. Probably most significant to these introductions are predatory fishes, including striped bass (*Morone saxatilis*), largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), and spotted bass (*Micropterus punctulatus*). Striped bass are known to prey on large numbers of juvenile salmon at diversion structures such as the Red Bluff Diversion Dam, or where hatcheries release large numbers of juvenile fish. The three bass species can also be important predators, particularly when they inhabit in-channel gravel pits or other obstacles to juvenile salmon migration.

Table 3. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Central Valley fall-run Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is high. See methods section for explanation.

Factor	Rating	Explanation
Major dams	Medium	Dams prohibit access to many historical spawning areas, alter flows, and simplify stream geomorphology; however, flow releases generally provide adequate water quality and temperatures below major dams.
Agriculture	Medium	Diverted water reduces stream flow and entrains juvenile salmon; levees protecting agricultural lands limit salmon access to floodplains, tidal marshes, and other important habitats.
Grazing	Low	Relatively little grazing takes place on the CV valley floor.
Rural/ residential development	Low	Generally minimal impact on large river systems (e.g., Sacramento), but increasingly connected to urbanized areas.
Urbanization	Medium	Urban areas widespread and growing in many portions of historical range; urban landscapes generally simplify habitats, impair aquatic ecosystem function and pollute streams.
Instream mining	Medium	Gravel pits in rivers are a problem in some locations, particularly in the San Joaquin River basin.
Mining	Medium	Legacy effects of hydraulic and hard rock gold mining remain; impacts may still be severe at a local scale. For instance, toxic run off from Iron Mountain Mine remains a perpetual threat.
Transportation	Medium	Most smaller Chinook streams have roads and railroads along them, often leading to habitat simplification.
Logging	Low	Little logging in the CV although logging may affect upper portions of CV watersheds.
Fire	Low	Fire may affect upper portions of CV watersheds and effects can be propagated downstream.
Estuary alteration	Medium	The Delta and Estuary are greatly altered and current physical and water habitat conditions impact effective migration of adults and juveniles in both river basins.
Recreation	Low	Recreation can disturb redds and spawners.
Harvest	Medium	Ocean and inland fisheries may harvest natural-origin (wild spawned) fish at unsustainable rates.
Hatcheries	High	CV fall-run Chinook are genetically homogenized.
Alien species	Medium	Introduced species increase predation, competition, and decrease food supply but do not threaten fall-run with extinction.

Effects of Climate Change: Climate change is one of the most significant emerging threats to the persistence of CV salmon (Williams 2006, Katz et al. 2012). At the southern edge of the Chinook salmon range, the CV fall-run already experiences environmental conditions near the limit of its tolerance, making it highly vulnerable to climate change (Moyle et al. 2008). Thus, small thermal increases in summer water temperatures will result in suboptimal or lethal conditions and consequent reductions in distribution and abundance (Ebersole et al. 2001, Roessig et al. 2004). Changes in precipitation in California may also significantly alter CV fall-run habitats. Climate change models predict that a larger proportion of annual precipitation will fall as rain, rather than snow, running off quickly and earlier in the season. With less water stored in snowpack, reservoirs will potentially have less water available for fishery releases, particularly during summer and fall months. The available water is also likely to be warmer. During summer and fall, high water temperatures will be exacerbated due to the lower base flows resulting from reduced snowpack (Hamlet et al. 2005, Stewart et al. 2005).

For fall-run Chinook salmon, adults may have to ascend streams later in the season and juveniles may leave earlier, narrowing the window of time for successful spawning and rearing. Snowpack losses are expected to be increasingly significant at lower elevations, with elevations below 3,000 m suffering reductions of as much as 80% (Hayhoe et al. 2004). Consequently, in the long-term, changes in stream flow and temperature are expected to be much greater in the Sacramento River and its tributaries, which are fed by the relatively lower Cascades and northern Sierra Nevada, than are changes in rivers to the south, which are fed by snowpack that is expected to remain more consistent in the higher elevations of the southern Sierra Nevada (Mote et al. 2005).

One of the least understood effects of climate change is the impact on ocean conditions. However, the implications of predicted rises in sea level and temperature, along with changes in wind patterns, ocean currents, and upwelling, all suggest major impacts to CV salmon populations while in the ocean environment. Ocean survival rates in California salmon have been closely linked to several cyclical patterns of regional sea surface temperature, such as the Pacific Decadal Oscillation, El Niño Southern Oscillation (Beamish 1993, Hare and Francis 1995, Mantua et al. 1997, Mueter et al. 2002), and the North Pacific Gyre Oscillation (Di Lorenzo et al. 2008). With increasing temperatures, the concentration of zooplankton (the primary food source for juvenile salmonids entering the ocean) may decrease, resulting in lower salmon survival (McGowan et al. 1998, Hays et al. 2005). Smolt-to-adult survival is also strongly correlated with upwelling in the Gulf of the Farallones, driven by strong winds during the spring and fall (Scheuerell and Williams 2005). Between 2005-2008, short-term anomalies in ocean conditions, resulting in decreased upwelling during critical times of year, were the likely proximate cause of low ocean survival for CV Chinook salmon (Barth et al. 2007, Lindley et al. 2009). Thus, as climate change results in more variable upwelling conditions, salmon populations may fluctuate more widely.

Status Score = 2.7 out of 5.0. High Concern. The Central Valley fall-run Chinook is listed as a species of special concern by NMFS and CDFW. The NMFS status review concluded that "…high hatchery production combined with infrequent monitoring of natural production make assessing the sustainability of natural production problematic, resulting in substantial uncertainty regarding this ESU (Myers et al. 1998)."

Table 4. Metrics for determining the status of Central Valley fall-run Chinook salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is high. See methods section for explanation.

Metric	Score	Justification
Area occupied	2	Most populations sustained by hatcheries with some indication of natural self-sustaining populations in the upper Sacramento River watershed, Clear and Butte creeks.
Estimated adult abundance	4	Annual spawning returns generally exceed 100,000 fish.
Intervention dependence	2	The majority of remaining spawning and rearing habitat is dependent on instream flow releases from major dams, gravel augmentation and other ongoing efforts; population appears largely dependent on hatchery augmentation.
Tolerance	3	Moderate physiological tolerance.
Genetic risk	2	High hatchery production has resulted in genetic homogenization reducing overall fitness.
Climate change	3	Least vulnerable of CV Chinook runs to extirpation although models suggest dramatic changes to lower elevation CV rivers and streams will have negative effects.
Anthropogenic threats	2	1 High, 9 Medium threats.
Average	2.7	19/7.
Certainty	4	Well studied although still much uncertainty about ocean stage.

Management Recommendations: There are three general directions where management actions could improve status: (1) improving information (2) improving habitat, and (3) and changing hatchery practices.

Improving information. All hatchery winter- and late-fall Chinook are adipose fin-clipped and implanted with coded-wire tags (CWTs) in the cartilage of their snout. Beginning in 2007, 25% percent of the roughly 32 million fall-run Chinook produced annually in CV hatcheries have been similarly marked. When an adipose fin-clipped salmon is harvested or a carcass recovered, CWT data can be read with the origin of the fish surmised. This fractional marking allows fishery managers to estimate the relative proportion of hatchery to naturally produced fish caught in commercial and recreational fisheries and returning to Central Valley rivers and hatcheries. Fractional marking data can also be used to determine straying rates of groups of tagged fish released from different hatcheries or compare stray rates between fish from the same hatchery released in different locations or at different times. However, because only 25% of hatchery fish are marked, most hatchery fish and natural origin (wild) salmon are still visually indistinguishable, which makes implementing management actions to specifically benefit wild Chinook salmon nearly impossible.

A major step towards improving management of Central Valley Chinook salmon stocks would be to mark all hatchery fish with an adipose fin clip. A fraction of these fish should be CWT tagged. Total marking is supported by the American Fisheries Society California-Nevada Chapter which argues that the practice would provide the following benefits:

- All hatchery fish would be instantly and visibly distinguishable from wild salmon;
- Implanted internal tags in a fraction of adipose-clipped fish would continue to allow data collection on hatchery of origin, stock, race, and age as is done with current fractional marking programs;
- Wild fish could be distinguished and allowed selective access to limited spawning habitats;
- Visual determination allows improved targeting of wild stocks for genetic monitoring;
- Hatcheries could better manage their broodstocks.

Marking of all hatchery fish has been implemented successfully in Idaho, Oregon and Washington, and in the Great Lakes. There are many that believe marking all hatchery fish would inevitably lead to a fishery in which only marked (hatchery) fish are kept, as it has in these locations. Potentially elevated mortality rates of released fish due to stress and increased marine mammal predation make mark-selective fisheries controversial. Mark selective fisheries appear to be particularly inappropriate here in California, where naturally produced Central Valley fall run are a major component of the catch that support commercial and sport fisheries. Any proposal to implement total marking in California, therefore, should include assurances that managers will not move to a mark selective fishery.

Ocean data. There is also a need to improve monitoring of salmon in the marine ecosystem off the California coast. Currently, our understanding of how ocean conditions affect salmon is largely educated guesswork with guesses often made long (sometimes years) after an event affecting the fish has occurred. An investment in better knowledge should have large pay-offs for better salmon management and population recovery.

Habitat diversity and a return to resiliency. Habitat diversity is essential to maintaining life history diversity in salmon populations. In the Central Valley, where a majority of salmon habitat has been lost behind either dams or levees, conservation strategies that restore and improve physical habitat quality, extent, and connectivity are essential tools in improving the resiliency of wild anadromous salmonid populations in a rapidly changing world. Attaining volitional passage to spawning and rearing habitat above Central Valley rim dams is important for the long-term viability of Central Valley salmon populations as wild fish. Likewise, implementing ecologically designed flow regimes, which mimic natural flow patterns, would improve conditions for salmon and other native fish below dams. On Putah Creek (Solano County) a flow regime designed to mimic the seasonal timing of natural increases and decreases in stream flow helped reestablish native fishes and reduce the abundance of alien (nonnative) fishes 20 km below the Putah Creek Diversion Dam. Importantly, restoration of native fishes only required a small increase in the total volume of water delivered downstream (Kiernan et al. 2012). Better management of New Melones reservoir to increase San Joaquin River fall-run Chinook smolt survival could include increasing releases during wet years to better mimic natural spring releases. Such actions would also benefit downstream water quality.

Today, the Central Valley is a patchwork of agricultural lands and communities located on former floodplains and wetlands, which are now mostly separated from rivers by high, steep levees. As a consequence, access to ancestral floodplain habitats by juvenile salmon and other native fishes has been greatly diminished. Restoring connectivity between river channels and seasonal habitat such as oxbows, side channels, riparian terraces, and especially floodplains should be a high priority for restoration projects. Redesigning and managing the last large-scale floodplains still connected to the Sacramento River (i.e., Sutter and Yolo Bypasses) for salmon would be an obvious place to start. On the San Joaquin River, reconnection of Paradise Cut and wildlife refuge floodplains will be essential steps to expanding life history diversity, increasing growth, and increasing seasonal habitat for salmonids south of the Delta. Setting back levees where feasible – such as the Lower Elkhorn project which expands the Sacramento and Yolo bypasses – improves flood protection while simultaneously enhancing floodplain habitat for fish and wildlife. Floodplain farmlands on the "dry side" of levees can also be managed to benefit of fish and aquatic ecosystems by intentionally inundating fields during winter. This mimics natural flood patterns and allows farm fields to serve a similar function to natural floodplains where shallow water and longer residence times combine to create engines of natural productivity where vast quantities of invertebrates are produced. This shallow water now teeming with fish food can be drained back to river channels to help support downstream aquatic food webs and fish populations in the river and Delta.

Particularly important are restoration strategies that connect all necessary habitat types for salmon to complete their freshwater life stages. For example, Butte Creek, which flows through Sutter Bypass, is essentially the last functioning floodplain system in the Central Valley. When restoration efforts addressed spawning habitat, passage and flow issues, Butte Creek became the only watershed in the Valley to have every link in the salmon habitat chain intact. When that happened, the fish population responded quickly and robustly, making Butte Creek a model for other watersheds. Restoration tools that restore natural riverine processes, such as ecological flow regimes and floodplain reconnection, help to recreate the dynamic mosaic of Central Valley habitat types under which our native fish evolved and to which they are adapted. But restoration actions alone will never result in recovery of abundant self-sustaining, naturally produced populations of Central Valley fall-run Chinook, as long as large numbers of straying hatchery fall-run continue to spawn in the wild.

Improving hatchery practices. Where large numbers of "domesticated" hatchery fish interbreed with naturally spawning fish, populations average only about half the number of returning offspring per spawner compared with wild counterparts (Christie et al. 2014). Decades of hatchery releases in the Central Valley have essentially domesticated the gene pool and today, fall-run populations breeding in different Central Valley rivers are genetically indistinguishable from each other or from hatchery fish (Williamson and May 2005). Current hatchery practices depress the reproductive and adaptive potential of naturally produced fall run and severely inhibit their potential for recovery. Stray hatchery fish, which spawn in the wild, also limit the ability of wild populations to adapt to changing habitat conditions through natural selection. Emerging understanding of the dangers of domestication selection indicates that every effort must be made to segregate natural spawning and hatchery stocks. This means changing practices to limit straying of hatchery fish. At the same time, hatchery practices can also be improved so that when introgression does inevitably occur, the fitness costs to naturally reproducing populations will be minimized.

Drastically reducing (or ideally ending) gene flow between hatchery and naturally reproducing spawning groups is essential for recovery of naturally reproducing, locally adapted stocks. The following management reforms, if implemented *along with* landscape-level habitat restoration, will help alleviate the demographic and genetic impacts of hatcheries on natural populations and increase the likelihood of recovering naturally produced, self-sustaining populations of Central Valley fall-run Chinook. They could also help sustain and rejuvenate California's commercial and sport fisheries. Management alternatives for segregating hatchery and naturally reproducing gene pools include:

- *Tidewater imprinting of hatchery releases.* A salmon's ability to "home" to a natal river is a result of juvenile fish being exposed to sequential stream odors and other cues on their downstream migrations to the ocean. When returning as adults, they follow these olfactory signposts in reverse. Hatchery-reared fish imprinted to upstream hatcheries and trucked to off-site release locations will attempt to return to their natal hatcheries in the upper watersheds without the benefit of the olfactory map, leading to high rates of straying and interbreeding with wild stocks. Instead, all trucked hatchery salmon should be imprinted to estuarine and coastal locations by holding them in net pens before release. Carefully selecting imprinting locations away from important spawning streams will minimize straying while maximizing harvest. A pilot program carried out at Pillar Point Harbor just north of Half Moon Bay has demonstrated that hatchery fish held in net pens for 5 days prior to release into the coastal ocean are caught in commercial and sport fisheries at much higher rates than either hatchery fish trucked but not imprinted or those released at the hatchery. Part of this seems to be a tendency of the imprinted fish to mill around the imprinting location exposing them to increased harvest.
- *Harvest all hatchery fish.* Establish highly efficient terminal fisheries to harvest all hatchery fish returning to estuarine imprinting locations. Increased harvest efficiency is a win-win, increasing the return on hatchery investment while simultaneously removing hatchery fish before they reach upper watersheds where they negatively impact and interbreed with wild stocks.
- *Switch to hatchery brood stocks that are highly different from local genomes.* This would be a marked departure from current practices which aim to make hatchery broodstocks as similar as possible to naturally produced populations. While intended to reduce the ill effects of domestication selection on "wild" populations, the outcome of current practice may in reality produce the opposite effect by allowing domesticated alleles to thoroughly permeate the naturally-reproducing population and therefore significantly reduce the reproductive capacity of the entire population. When hybridization between naturally produced individuals and hatchery strays inevitably occurs, their hybrid progeny inherit a genome adapted to the hatchery but unfit for local river conditions. Broodstock for artificial propagation under our divergent broodstock approach should be selected from distant locations where life history characters (especially migratory timing) are incompatible with California streams. Individuals carrying these domesticated genes would experience extremely high mortality rates in the wild, and be culled from the naturally reproducing gene pool with in generation or two. In this way hatchery genes would be rapidly purged by natural selection before they could permeate the naturally-reproducing population and begin to depress the reproductive capacity of the wild population.
- *Physical segregation.* Where hatchery fish do return to upper watersheds, use physical segregation via active sorting at weirs or dams so that only wild fish are passed upstream above barriers. This active sorting would only be possible if all hatchery fish were visually marked or if other "real-time" tests of hatchery origin become available.

CENTRAL VALLEY LATE FALL-RUN CHINOOK SALMON
Oncorhynchus tshawytscha

High Concern. Status Score = 2.1 out of 5.0. Central Valley late fall-run Chinook salmon are vulnerable to extinction by 2100. They have been extirpated from the majority of their native spawning habitat.

Description: Central Valley (CV) late fall-run Chinook are morphologically similar to other CV Chinook salmon, but tend to be larger, reaching 75-100 cm TL and weighing 9-10 kg or more. Like other Chinook salmon, late fall-run have numerous black spots and larger 'chevron' or irregular dark blotches on the back, dorsal fin, and both lobes of the tail in both sexes. This spotting on the caudal fin and the black coloration of their lower jaw make them distinguishable from other sympatric salmonid species. They have 10-14 major dorsal fin rays, 14-19 anal fin rays, 14-19 pectoral fins rays, and 10-11 pelvic fin rays. There are 130-165 scales along the lateral line. Branchiostegal rays number 13-19. They possess more than 100 pyloric caeca and have rough and widely spaced gill rakers, 6-10 on the lower half of the first gill arch.

Taxonomic Relationships: The four runs of Chinook salmon found in the Central Valley–winter, spring, fall and late fall–are more are named for season of adult entry into fresh water from the ocean. They are more closely related to each other than they are to populations outside the Central Valley (Williams 2006). All four runs are genetically distinct (Meek et al. 2016) and can be consistently differentiated based on fin clip samples run for genetic differences in Single Nucleotide Polymorphisms (SNP). Late fall-run Chinook were the last of the four distinct CV Chinook salmon runs to be recognized because their spawning migration was obscured by the more abundant and widespread fall-run Chinook with whom they share a similar arrival time. In 1966, after completion of Red Bluff Diversion Dam, two distinct peaks in run timing were observed passing over the fish ladder, the latter was recognized to be late fall-run.

Late fall-run Chinook salmon life histories differ from other Central Valley runs in maturity of fish entering fresh water, time of spawning migrations, spawning areas, incubation times, and migration timing of juveniles (Moyle 2002). For management purposes, CV juvenile Chinook salmon have been assigned to winter, spring, fall, and late fall-runs by length-at-date criteria, reflecting different spawning times and rearing conditions. Unfortunately, actual distributions of lengths of juvenile Chinook salmon in the Central Valley violate the basic two basic assumptions needed for these criteria to work: (a) juvenile length distributions do not overlap among runs, and (b) steady growth rates keep these size differences constant (Harvey et al, 2014). Although the use of body morphometrics can improve length-at-date identification of runs albeit with considerable error (Merz et al. 2014), genetic techniques have revealed large overlaps in sizes and growth rates of juveniles of different runs and make it clear that length-at-date data should be viewed with skepticism. Today, genetic methods are the most reliable techniques to separate runs (Harvey et al, 2014, Meek et al. 2016).

The National Marine Fisheries Service (NMFS) currently groups late fall-run Chinook with the fall-run ESU, though there are important life history differences between the two runs. Yoshiyama et al. (1998), Moyle (2002), and Williams (2006) and others recognize the Central Valley late fall-run to be a distinct taxonomic entity with a unique evolutionary trajectory (as evidenced by its distinct life history strategy) and with its own specific management concerns. The NMFS Southwest Fisheries Science Center considers late fall-run and fall-run as two separate races under a single ESU for management purposes (NMFS 2010). CDFW recognizes late fall-run Chinook as a unique life history strategy (Moyle et al. 2015).

Late fall-run Chinook are also spawned, and raised at Coleman National Fish Hatchery on Battle Creek. This fish are released as yearlings, at a time that there is low overlap with presence of other rearing salmon in the Sacramento River.

Life History: Late fall-run fish exploit extensive spawning and rearing habitat on the Central Valley floor during the cooler months and avoid temperatures that exceed thermal tolerances during summer and early fall. Historically this meant they largely used habitat at or above the present region covered by Shasta Reservoir. Late fall-run Chinook male their up-river spawning migrations in November-January, peaking in December; spawning is typically in January-March (Williams 2006). Juveniles emerge from spawning gravels in April-June and remain in fresh water 7-13 months before out-migrating to sea (Williams 2006). Peaks of emigration of smolts appear to be in fall (October-November) and spring, but juveniles apparently can out-migrate at younger ages and smaller sizes during most months of the year (2011-2015 passage data at Red Bluff Diversion Dam;
http://www.cbr.washington.edu/sacramento/data/query_redbluff_graph.html).

However, late fall-run Chinook life history is much less well-known than that of other CV Chinook runs because of its comparatively recent recognition as a discrete run and its tendency to migrate and spawn when the Sacramento River is high, cold, and turbid. This combination of factors makes this run particularly difficult to study, although advances in genetic differentiation should improve data specific to late-fall run Chinook. Historically, the annual spawning run was comprised of a mixture of age-2 to age-6 adults. Based on coded wire tag recovery in 2011, age-4 (51%) adults made up the majority of the run, followed by age-3 (30%), and age-5 (14%). Of only four age-6 fish recovered in the entire CV from all runs, all were brood year 2006 late fall-run Chinook (Palmer-Zwahlen and Kormos 2013).

Habitat Requirements: The specific freshwater habitat requirements of late fall-run adult and juvenile Chinook salmon have not been determined, but they are presumably similar to juveniles of the other runs of Chinook salmon. It is likely that optimal conditions fall within the range of physical and chemical characteristics of the unimpaired Sacramento River above Shasta Dam.

Late fall-run Chinook salmon are likely highly variable in their use of habitats from spawning grounds to the ocean. Some fry spend more time in the lower reaches of rivers and in the Sacramento-San Joaquin Delta, while others rear on the floodplains of the Central Valley before undergoing smoltification (Williams 2006). Juveniles that emigrate as yearlings are more likely to become smolts on the downstream migration and not spend as much time in the San Francisco Estuary (Williams et al. 2006). Seasonally inundated floodplain habitat historically provided rich foraging and rearing opportunities for juveniles, increasing size and survival (Jeffres et al. 2008).

Michel et al. (2013) found that tagged juvenile late fall-run Chinook salmon emigrated to the San Francisco Estuary/Delta at rates comparable to juveniles of other Central Valley runs, ranging from 14.3 km/day to 23.5km/day; migration rates were most closely associated with river width-to-depth ratios. These migration rates, and subsequent survival rates, depend on the route through which juveniles navigate the complex channels the Delta. These channels have flow patterns that can be influenced by the operation of Central Valley Project and State Water Project pumping facilities in the south Delta (Perry et al. 2010). Singer et al. (2013) studied juvenile Chinook and steelhead survival in 2009 and 2010 in the Delta and Estuary and found survival to the Pacific Ocean was less than 25% in both years, but that survival rates varied by habitat and year.

In brackish and salt water, significant juvenile growth occurs to help prepare them for the first critical month at sea (Williams et al. 2016). MacFarlane (2010) found that juvenile CV Chinook salmon grew relatively little (0.07 g/day median) while in the estuary, but gained 0.8 g/day - representing 4.3-5.2% of their body weight at ocean entry per day - in the rich coastal ocean off California. Considering the highly altered state of the San Francisco Estuary and the Delta today (Williams 2006), it is likely that smolt habitat use in the San Francisco Estuary has changed considerably over time (MacFarlane and Norton 2002). Currently, the nursery function typically ascribed to estuaries appears to largely absent but possibly can be deferred to initial ocean residence, at least in years where ocean conditions are productive at the time when smolts arrive at sea.

Distribution: The historical distribution of late fall-run Chinook salmon is not well documented. They most likely spawned in the upper Sacramento and lower McCloud rivers in reaches now blocked or covered by Shasta Dam and reservoir, as well as in Sacramento and San Joaquin River tributaries that provided adequate cold water in summer. There is also some evidence they once spawned in the San Joaquin River near where Friant Dam now stands and in other large San Joaquin tributaries (Yoshiyama et al. 1998). NMFS (Williams et al. 2016) defines the boundary of CV late fall-run Chinook as extending east from Carquinez Strait into the Sacramento and San Joaquin rivers and their tributaries. Currently, late fall-run Chinook are found primarily in the Sacramento River, and most spawning and juvenile rearing occurs in the mainstem between RBDD (Rkm 391) and Keswick Dam (Rkm 483). This range overlaps to some extent with spawning spring-run and winter-run Chinook salmon (NMFS 2016). Small but varying percentages of the total late-fall run spawn downstream of RBDD in some years. In 2003, for example, 3% of the late fall-run spawned downstream of the dam, while in 2004, no spawning occurred below the dam (Kano 2006a, b). R. Painter (CDFW, pers. comm. 1995) indicated that late fall-run Chinook have been observed spawning in Battle Creek, Cottonwood Creek, Clear Creek, Mill Creek, the Yuba and Feather rivers, but these are presumably a small fraction of the total population (Figure 1). The Battle Creek spawners are likely derived from fish that strayed from Coleman National Fish Hatchery (USFWS 2016).

Late fall-run Chinook juveniles presumably used the mosaic of historical aquatic habitats found in the central valley including, side channels, oxbows, floodplains and tidal marshes. In the ocean, late fall-run Chinook juveniles, presumably stay close to the coastal shelf, where upwelling provides rich foraging opportunities (Williams 2006) that enable rapid growth (MacFarlane 2010) and increase survival to adulthood (Williams et al. 2016).

Trends in Abundance: The CV late fall-run Chinook salmon was only recognized as a distinct run in 1966 after RBDD was completed and its fish ladder finally permitted accurate accounting of late-arriving fish. Historical abundance is not known. Total late fall-run estimates from 1974 to present have varied considerably (CDFW GrandTab 2017, Figure 1). Estimation of the abundance of the late fall-run has been more difficult since 1992 when the RBDD gates were opened to provide free passage of the listed winter-run Chinook salmon. Escapement estimates have remained below about 20,000 with the notable exception of 1997 and 2001 (Figure 2). An average of about 700 adults have returned to Battle Creek each year since 1992 (Figure 3).

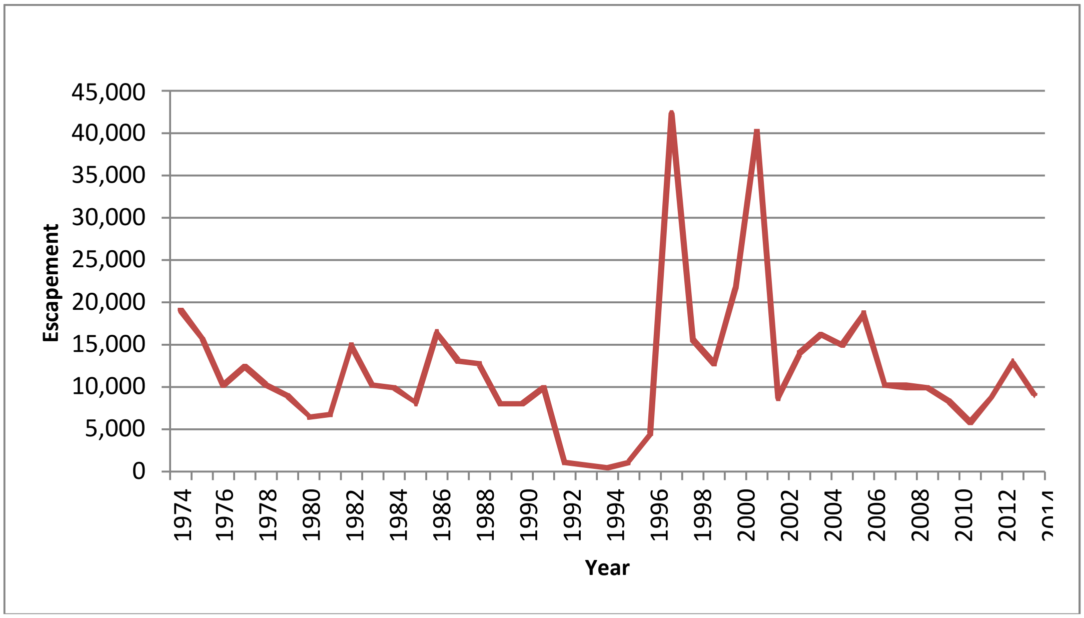


Figure 1. Central Valley late fall-run Chinook salmon escapement estimates, 1974-2014. Data from: CDFW GrandTab.

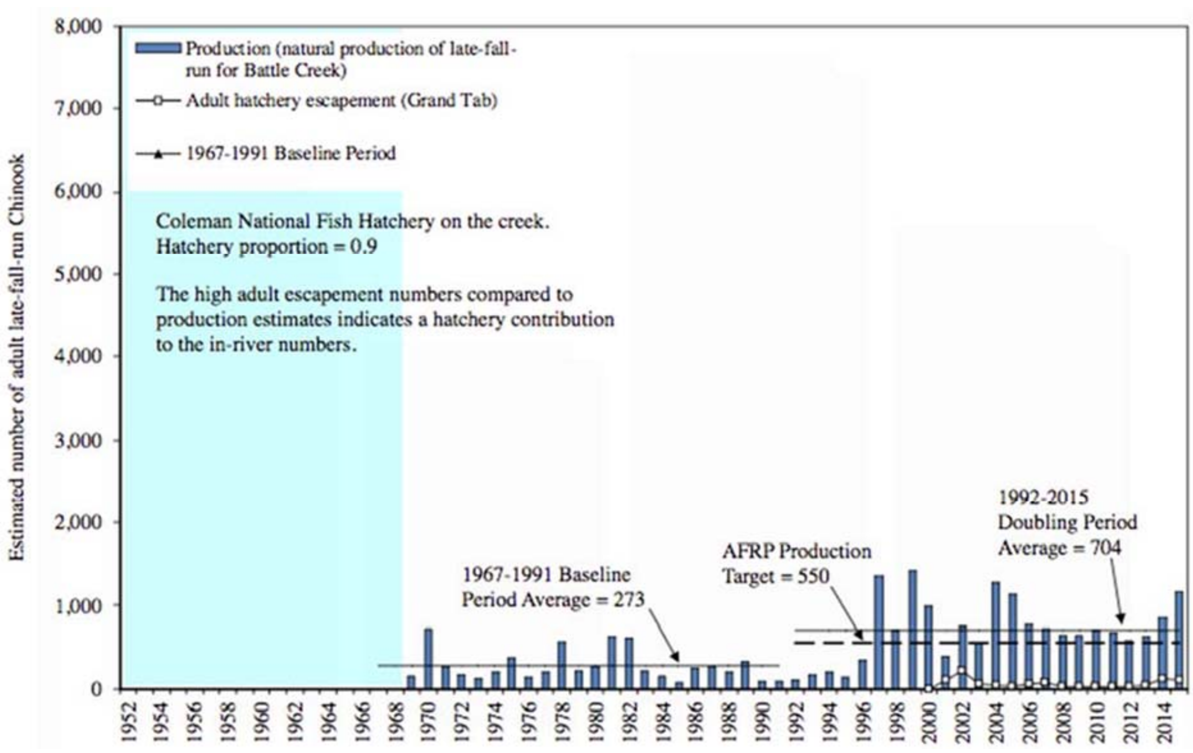


Figure 2. Estimated natural and Coleman National Fish Hatchery escapement of adult late fall-run Chinook salmon in Battle Creek. From: USFWS 2016 Anadromous Fish Restoration Doubling Fall-run is Goals Graphs, Fig. 14, pg. 14.

Fall-run Chinook salmon are by far the most abundant run in the Central Valley, but because of similarities in run-timing and spawning areas with late fall-run, the two are often misidentified and this confounds accurate estimation of abundance for both runs. By erroneously grouping the two runs, potential declines in late-fall abundance may be masked by the much more abundant fall-run (Meek et al. 2016). In order to increase accuracy of run classification, CDFW and the USFWS conduct joint aerial redd surveys to determine the extent and temporal distribution of late fall-run Chinook spawning. They also conduct carcass surveys on the mainstem Sacramento River (Bergman et al. 2012). While these studies have improved abundance estimates somewhat, accurate identification of late-fall run Chinook remains difficult; expert judgment is relied upon - based on timing of spawning and physical appearance - to differentiate between the carcasses and redds of the two runs (Bergman et al. 2012). To better understand spawning distribution and late fall-run Chinook abundance, CDFW has been upgrading fish counting technology and monitoring on Antelope, Bear, Cow, Deer, and Mill creeks (Bergman et al. 2012, Killam et al 2016).

Factors Affecting Status: The causes of population decline late fall-run Chinook salmon are poorly documented. A general overview of anthropogenic threats is provided below; for more detailed discussion of factors affecting Chinook salmon in the Central Valley more generally, see the CV spring-run Chinook salmon account.

Dams. When Shasta and Keswick dams were built in the 1940s, they presumably blocked late fall-run Chinook access to a majority of their historical spawning areas, where spring water from Mt. Shasta and snow-melt kept water temperatures cool enough for successful spawning, egg incubation, and juvenile survival year-round. Today, late fall-run Chinook living in the mainstem Sacramento River are largely dependent on coldwater releases from Shasta. Dams on the Sacramento River and its tributaries have reduced or eliminated recruitment of spawning gravels into downstream spawning areas below dams and altered temperature regimes. Loss of spawning gravels in the Sacramento River below Keswick Dam is regarded as a serious problem; large quantities of gravel are now trucked and placed in the river where it is used by all four runs for spawning. A temperature control device (TCD) was installed in 1991 on Shasta Dam to provide cooler water in summer for endangered winter-run Chinook; it has presumably also has benefited late fall-run Chinook.

Between 1966 and 2013, RBDD delayed Chinook passage to upstream spawning areas and concentrated predators, increasing mortality on out-migrating smolts of all runs. Kope and Botsford (1990) documented that the overall decline of upper Sacramento River salmon was closely tied to the construction of RBDD. The gates are now open year-round, allowing uninhibited passage of both adult and juvenile CV Chinook salmon.

Agriculture. Outmigrant mortality of both fry and smolts is a major factor affecting all runs of salmon in the Sacramento-San Joaquin drainage. Small numbers of outmigrants are presumably entrained at unscreened irrigation diversions along the Sacramento River or its many tributaries and sloughs that are operating during the migration period. The large State Water Project and federal Central Valley Project pumps that divert Sacramento River water in the southern Delta to holding reservoirs and eventually to conveyances such as the California Aqueduct also have a major negative impact on CVS Chinook salmon populations through entrainment and increased vulnerability to predation. Extensive bank alteration to benefit agricultural operations has reduced the amount of cover from alien striped bass (*Morone saxatilis*), terns, herons and other predators. It is also likely that warm, polluted agricultural return waters negatively affect water quality, even in rivers as large as the Sacramento River. Levees to protect agricultural fields from flooding have substantially degraded riparian habitats and eliminated connectivity of main stem river channels to historically widespread (and ecologically important) floodplain habitats (Sommer et al. 2001, Jeffres et al. 2008).

Urbanization. Urbanization can simplify and pollute Chinook salmon habitats although the impacts from the City of Redding are unknown on late fall-run Chinook salmon.

Mining. Existing gravel mining operations and legacy effects of past gravel mining, as well as placer and hydraulic mining for gold, may continue to affect late fall-run Chinook salmon; however, the effects are largely unknown. Lasting legacy impacts may be especially acute in the middle to upper portions of watersheds (preferred late fall-run spawning areas) where hydraulic mining was most prevalent and caused dramatic changes to river geomorphology and hydrology and severely degraded aquatic habitats.

In the past, Iron Mountain Mine, northwest of Redding, drained highly acidic water laden with heavy metals into the Sacramento River, resulting in acute mortality of Chinook salmon. Although the discharge is now highly controlled, unlikely failure of the Spring Creek retention reservoir could result in impacts to all aquatic life in the entire Sacramento River.

Estuarine alteration. There is growing appreciation of the importance of "biocomplexity" for the persistence of salmon in highly variable, complex environments (Hilborn et al. 2003), including those in the CV (Carlson and Satterthwaite 2011). Biocomplexity is defined as multiple variations in life history that improve the ability of populations to persist in changing environmental conditions. There is a direct link between salmonid access to diverse habitats and the life history diversity that develops over time to allow them to exploit those habitats (Carlson and Satterthwaite 2011); access to diverse habitats increases salmon resiliency to change and stressors and is a major goal in salmonid restoration. Loss of once-diverse, productive rearing habitats in the San Francisco Estuary has changed the way juvenile Chinook use this habitat (MacFarlane and Norton 2002). An alternative strategy that appears to have become common is to move through the estuary as quickly as possible (MacFarlane 2010). Pumping plants in the South Delta influence movement by increasing the likelihood that out-migrating smolts will inadvertently enter the interior Delta, where longer migration routes, impaired water quality, and increased predation result in higher mortality rates (Perry et al. 2010, Singer et al. 2013).

Better understanding of the use of migration corridors (Michel et al. 2013) and the timing and causes of smolt mortality before ocean entry (Singer et al. 2013) remain elusive but important parts of improved management of late fall-run CV Chinook salmon. General observations suggest that rearing conditions in the estuary are often poor; highest survival occurs during wet years, when passage through the estuary is likely most rapid and water quality is higher (Brandes and McLain 2001, Baker and Mohrhardt 2001). Flooding in wet years also increases available rearing habitat in the Delta and Yolo Bypass, which can have a positive effect on foraging opportunities, growth potential, and survival (Jeffres et al. 2008, Corline et al. 2017, Katz et al. 2017). Additionally, recent studies found that the further downstream a group of hatchery-reared late fall-run smolts is released, the longer the group takes to reach the ocean (Michel et al. 2013). These findings suggest that environmental cues that trigger migration in the upper watershed may be subdued, delayed, or absent in the lower river

Harvest. The effects of ocean harvest on the mixed CV Chinook salmon fishery, in general, are poorly understood due to lack of a robust genetic stock identification program and lack of marking of all hatchery fish, which would enable wild fish to be distinguished at sea. The harvest rates of late fall-run Chinook are not known, but it is likely that they are harvested at similar rates as spring-run Chinook salmon. Palmer-Zwahlen and Kormos (2013) collected over 8,700 coded wire tags from Chinook salmon harvested at sea in 2011, and found that 86% belonged to fall-run, followed by late-fall-run (2%), spring-run (1%), and winter-run (< 0.4%).

In general, Satterthwaite et al.(2013) found that tagged winter-run and late fall-run Chinook salmon were relatively restricted to the ocean south of San Francisco Bay compared to the fall- and spring-run Chinook, which were more concentrated to the north. This study highlights the potential utility of using spatial management, such as time-area closures, to control impacts of harvest on the endangered winter-run Chinook, but also late fall-run Chinook with low abundance. Although hatcheries are operated to sustain fisheries and hatchery fish can sustain higher harvest rates than wild fish, fisheries at sea do not currently have a reliable and practical method to discriminate between them. Fisheries may, therefore, be taking a disproportionately large number of natural-origin late fall-run Chinook salmon than is currently estimated. Fisheries also select for larger fish, leaving younger, smaller, and less fecund fish as spawners, which reduces resiliency of the populations over time. The effects of freshwater recreational harvest of late fall-run Chinook salmon is unknown, and estimates are likely unreliable as it is difficult to identify Chinook carcasses or redds by run (Bergman et al. 2012). Other effects of harvest are discussed in the CV spring-run Chinook salmon account.

Hatcheries. Late fall-run Chinook salmon have been reared at Coleman National Fish Hatchery on Battle Creek since the 1950s, even though the run was not formally recognized until 1973 (Williams 2006). The current production goal is one million smolts per year, which are all adipose fin-clipped and released into Battle Creek from November through January (Palmer-Zwahlen and Kormos 2013). Hatchery broodstock selection for late fall-run fish includes both fish returning to Coleman National Fish Hatchery and those trapped below Keswick Dam based on run-timing and appearance (Bergman et al. 2012). Large numbers are needed because survival rates are very low (0.78% at Coleman), even compared to other hatchery-reared salmon smolts (Kormos et al. 2010). Hatchery production is likely to have impacts on the naturally-spawning population through predation, competition, and hybridization (Kormos 2013, NMFS 2016). The genetic influence of hatchery fall-run Chinook on natural spring- and late fall-run production is also unclear at this time but remains a concern (NMFS 2016). Hatchery release practices are correlated with increased straying rates of adults, posing genetic risk to other Central Valley salmon populations (Kormos et al. 2012, Palmer-Zwahlen and Kormos 2013). To help combat these threats and reduce hybridization and interbreeding between fall- and late fall-run Chinook, the California Hatchery Scientific Review Group (CHSRG 2012) recommends collection of natural-origin adult fish at alternate locations instead of only the fish trap at Keswick Dam, including collecting and retaining fish from Battle Creek, to enhance genetic diversity in the broodstock.

Alien species. Chinook salmon have always had to live with predation being a major source of mortality, including that by steelhead rainbow trout and other native fish and numerous birds and mammals. Over the past 150 years, numerous predatory fish species have been introduced to the Bay-Delta system, replacing native predatory fish. Several species of introduced fishes prey upon Chinook salmon, including striped bass and largemouth, smallmouth, and spotted bass (*Micropterus* spp.). Striped bass can consume large numbers of juvenile salmon, particularly at diversion structures, or retaining ponds such as those near the south Delta pumping facilities, or where hatcheries release large numbers of juvenile fish. Predation may be limiting some populations of Chinook salmon, especially in areas with manmade structures that alter the environment such as those found near the pumping and diversion stations in the highly altered Delta (F. Cordoleani, NMFS, pers. comm. 2017).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Central Valley late fall-run Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is low. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Dams block access historical spawning grounds; current spawning depends on operation of Shasta Dam.
Agriculture	Medium	Levees reduce access to floodplains; diversions and agricultural return water decrease water quantity and quality.
Grazing	Low	Little grazing on valley floor.
Rural/ residential development	Low	Source of minor changes to riverbanks and pollution.
Urbanization	Low	Urban areas along Sacramento River and tributaries may restrict habitat and decrease water quantity and quality.
Instream mining	Low	Gravel mining and legacy effects of placer and gravel mining may continue to impair habitats.
Mining	Low	Catastrophic discharge from Iron Mountain Mine can potentially negatively impact all fish in the Sacramento River.
Transportation	Low	Roads have minor impact on Sacramento River.
Logging	Low	Generally low impact; occurs at higher elevations.
Fire	Low	Few impacts on mainstem river likely.
Estuarine alteration	High	San Francisco Estuary has increasingly limited capacity to support juvenile salmon.
Recreation	Low	Boating, wading, and angling can disturb spawners and migrants.
Harvest	Medium	Ocean and inland fisheries may harvest natural-origin (wild spawned) fish at unsustainable rates in mixed fishery at sea.
Hatcheries	Medium	One million+ juveniles of hatchery origin are released each year but their impact on wild fish is not well understood.
Alien species	Medium	Predation and competition from introduced fishes is a growing concern, especially near water infrastructure and in severely altered habitats such as the Delta.

Effects of Climate Change: Moyle et al. (2013) found late fall-run Chinook salmon to be "critically vulnerable" to extinction from the effects of climate change because of the run's dependence on cold water released from dams, their low abundance, and their limited range due to dam construction. For a general discussion of climate change impacts on CV Chinook salmon, see the spring-run Chinook salmon account.

The maintenance of a cold water pool in Shasta Reservoir to keep water in the Sacramento River cold enough to support late fall-run habitat requirements year-round are particularly critical for winter- and late fall-run Chinook. Maintaining the cold water pool will become increasingly difficult during periods of extended drought and in the face of predicted increasing air and water temperatures across northern California (Williams et al. 2016). Less snowfall, and potentially reduced overall precipitation, is expected across California in the coming decades (Lindley et al. 2007). Extended droughts are also predicted for California under most climate change models and they could render remaining habitat unusable, either through temperature increases or lack of adequate flows from Shasta Reservoir (Williams et al. 2016). Increased frequency and/or intensity of drought will lead to less water available behind dams, reducing the cold water pool available for releases to support adult fish in warmer summer months. Thus, spring-fed Battle Creek may be crucial as a refuge during periods of drought, as was encountered from 2012-2016.

Status Summary Score = 2.1 out of 5.0. High Concern. Late fall-run Chinook have been extirpated from a considerable portion of their historical spawning grounds. In the past 10 years, numbers have fluctuated, but appear to mirror numbers seen in the 1970s and 1980s. According to NMFS, they "continue to have low, but perhaps stable, numbers" (NMFS 2010). Nevertheless, CV late fall-run Chinook are vulnerable to anthropogenic threats, stochastic events, and climate change because of their relatively small population size and limited spawning distribution. Lack of access to (and degradation of) spawning and rearing habitats may make this population exceptionally vulnerable to changes in water quality and flow in the Sacramento River, as in the case of an extended drought (2012-2016), changes in water management, or a major spill of toxic materials (Iron Mountain Mine). Their persistence depends on favorable operation of water projects (Shasta Dam) and hatchery operations (Coleman National Fish Hatchery). As a result, both the fall- and late fall-run Chinook salmon are considered a Species of Concern by the National Marine Fisheries Service (NMFS 2010) and CDFW (2015). At the very least, based on recent genetics, late fall-run Chinook salmon should be acknowledged as a discrete run and managed accordingly to aid in their monitoring and management.

Table 2. Metrics for determining the status of Central Valley late fall-run Chinook salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is low. See methods section for explanation.

Metric	Score	Justification
Area occupied	2	Only one primary population concentrated in the upper Sacramento River; some tributary spawning and rearing.
Estimated adult abundance	3	Total escapement has averaged approximately 10,000 spawners in recent years, while contribution from Coleman National Fish Hatchery is low (~10%).
Intervention dependence	2	Primary population is dependent on Shasta Dam operation for flows and gravel injection for spawning habitat improvement.
Environmental tolerance	3	Moderate physiological tolerance, multiple age classes.
Genetic risk	2	Hybridization with other runs occurs as evidenced by significant mixing of in-river spawners of natural and hatchery-origin on Battle Creek and potential hybridization with fall-run Chinook.
Climate change	1	Extremely vulnerable due to dependence on cold water releases from Shasta Reservoir, snowpack, and spring-fed flows.
Anthropogenic threats	2	2 High and 4 Medium threats.
Average	2.1	15/7.
Certainty	2	Least studied of Central Valley Chinook runs.

Management Recommendations: Currently, less monitoring, research, and management effort is directed to benefit late fall-run Chinook salmon than for any other run in the Sacramento River. A key to conserving late fall-run Chinook is to develop and implement specific monitoring and restoration measures tailored to its distinctive life history. Recent advances in genetic monitoring that allow more accurate run assignment should help in gathering life history data.

This run should benefit considerably from measures being taken to enhance winter- and spring-run Chinook salmon populations in the upper Sacramento River, such as maximum temperature thresholds and minimum flow requirements for releases from Shasta Reservoir and Delta pumping restrictions. However, specific studies should be undertaken to better understand the environmental requirements specific to the late fall-run, because this run needs protection at all stages of its life cycle. The U.S. Fish & Wildlife Service Anadromous Fish Restoration Program has set a goal in their final restoration plan of an average production (escapement plus catch in fishery) of 44,000 fish per year, although the official doubling goal (required in the Central Valley Project Improvement Act) is 68,000 natural-origin fish (USFWS 2016). Whether or not remaining accessible habitat is adequate to sustain a population at either level is uncertain. Spawning and rearing ground monitoring specific to the run is needed, as are habitat use and juvenile life history studies.

Restoration will require: (1) continuing to provide improved passage of adults to suitable holding and spawning areas, (2) protecting adults in spawning areas, (3) establishing additional spawning populations (e.g., Battle Creek) that are not supported by hatchery production, (4) providing passage flows that allow migrating juveniles to move through the Delta as rapidly as possible, (5) maintaining and expanding rearing habitats for juvenile fish in the mainstem river and floodplains (floodplain restoration in lower Battle Creek should be high priority), and (6) ensuring ocean and inland fisheries regulations (such as time-area closures) minimize impacts to the extent possible.

Habitat restoration is critical to maintaining long-term population stability in late fall-run Chinook salmon, particularly in the face of climate change. Enhancement of the Battle Creek population through restoration of floodplain and off-channel habitat in Lower Battle Creek and in the mid-Sacramento River and restoration of the San Joaquin watershed populations are important aspects of late fall-run Chinook conservation because they could increase the geographic range of the run, genetic adaptation and diversity, and resiliency to climate change.

These efforts will require continuous, adaptive management as well as improved monitoring and population evaluation programs for both hatchery and naturally-produced fish. A comprehensive monitoring plan for Central Valley salmon is a promising sign that efforts are being made to focus on better understanding and protecting all Chinook salmon stocks. However, the late fall-run will not be afforded the same level of attention as their ESA-listed counterparts until they are recognized as an independent ESU based on recent genetic work and likely categorized as a threatened species under state and federal Endangered Species Acts.

CENTRAL VALLEY SPRING CHINOOK SALMON
Oncorhynchus tshawytscha

Critical Concern. Status Score = 1.7 out of 5.0. Vulnerable to extinction in the next 50-100 years, or less. Small, self-sustaining populations remain in only a few watersheds.

Description: All California Chinook salmon are similar in morphology and other characteristics. They are categorized into various Evolutionary Significant Units (ESUs) for management purposes. These "runs" are distinguished mainly by genetic and life history traits (e.g., run timing, maturation, and rearing patterns), although there are often statistical differences in size among Chinook salmon ESUs (see UKTR fall-run Chinook salmon account for a description of some minor differences). Central Valley spring-run Chinook salmon (CVS Chinook) are mainly differentiated by their late maturation and geographic and spatial separation from the more abundant Central Valley fall-run Chinook salmon, but they also display considerable plasticity in their age at spawning, the juvenile life history stage, and migration behavior.

Taxonomic Relationships: Chinook salmon are most closely related to coho salmon (*O. kisutch*), with which they occasionally hybridize (Moyle 2002). Chinook runs are named for the season in which they begin their freshwater spawning migration but are delineated phenotypically, genetically and geographically. In California's Central Valley, there are four ESUs of Chinook: fall, late-fall, winter, and spring. While each run is distinct in timing of spawning and migration, location of spawning areas, and rearing strategies based on instream conditions, their juveniles that mix together in the Sacramento River are commonly differentiated by length-at-date size class projections when captured at rotary screw traps and salvaged at Delta pumps. These projections are often inaccurate, which can confound accounting of returns from each run (Meek et al. 2016, M. Johnson, CDFW, pers. comm. 2017). Spring-run fish are distinguished in part by their delayed maturation, which enables them to delay spawning for 4-5 months after entering fresh water. Prince et al. (2016), using genomic techniques, showed that the delayed maturation life history arose as a single evolutionary event in Chinook salmon and spread throughout the species' range, primarily through straying. Where conditions favored this late-maturing life history strategy, reproduction by strays was followed by positive selection for the spring-run phenotype over time. Presumably the rare early possessors of this distinctive gene complex would have spawned with early-arriving fall-run Chinook salmon, but their offspring possessed both the spring-run genetic mutation and the locally adapted genes of the dominant fall Chinook salmon. There would then have been strong selection for spring-run fish that could use streams at times and places not available to fall run Chinook (see life history section).

Based on molecular genetic techniques, there are two distinct self-sustaining populations of CVS Chinook salmon: Deer and Mill creeks (Tehama Co.), and Butte Creek (Tehama Co., Garza and Pearse 2008). Small populations of spring-run Chinook are consistently observed in Big Chico, Cottonwood, Antelope, Clear, and Battle creeks, but it is not known if these streams support self-sustaining populations or if these fish are strays from the Butte/Deer and Mill populations (M. Johnson, CDFW, pers. comm. 2017). In addition, there is a putative population of CVS Chinook in the Feather River, which is supported by releases of nearly 2 million juvenile fish from Feather River Fish Hatchery (FRFH) each year (www.rmpc.org, NMFS 2016). These fish are nearly identical genetically to fall-run Chinook salmon (Williams 2006), yet are included somewhat controversially in the ESU designation due to concerns of inbreeding and outbreeding depression among other Chinook populations in the Central Valley. Feather River CVS Chinook have hybridized with hatchery fall-run Chinook in the watershed over decades due to operations at FRFH that mixed fish of different run timings (NMFS 2016). An alternative hypothesis is that spring-run salmon recently diverged from fall-run salmon that re-colonized the Feather River after hydraulic mining ended (J. Williams, pers. comm. 2008). Small runs of spring salmon in the Yuba River, tributary to the Feather River, are too data-deficient for conclusive analysis of origin (NMFS 2016), but are most likely derived from Feather River fish and partly supported by strays from the Feather River Hatchery.

Life History: CVS Chinook salmon get their name for their spring migrations upstream during high springtime runoff events that swell rivers. These seasonal flows allow passage into higher elevation, smaller tributaries that are generally inaccessible to salmon at other times of the year. Once in headwater streams, adults hold in deep pools and spawn in early fall. Juveniles exhibit three distinct emigration strategies: 1) emigrating after a few months in fresh water; 2) emigrating after spending an entire year in fresh water; or 3) leaving spawning areas soon after hatching during high flows. After emigrating, juvenile spring-run Chinook may rear in the Sacramento River and other downstream habitats, such as the Sutter or Yolo bypasses, and migrate to the ocean as smolts during the spring or remain in their natal stream for an entire year and outmigrate the following fall, winter, or spring as yearlings. CVS Chinook exhibit considerable plasticity in their life history strategies and do not fit easily into the life history categories identified for most other Chinook salmon populations (Box 1).

CVS salmon begin their spawning migration in January-February and arrive in the Sacramento River primarily during April through June, with migrations peaking in mid-April in Butte Creek and in mid-May in Deer and Mill Creeks (Williams 2006, NMFS 2016). Johnson and Merrick (2012) found adult spring-run Chinook entering Deer and Mill creeks as early as late February and as late as July, with a peak in late April through early May, while historical documents indicate peak adult returns used to occur nearly a month later from late May to early June (M. Johnson, CDFW, pers. comm. 2017). They migrate as silvery, immature fish that only reach sexual maturity after reaching summer holding areas, which are generally higher in the watershed than those of other runs. They seek out deep pools with hyporheic flow, subsurface springs, or inlet streams that provide suitable cool summer water temperatures to allow them to survive high daytime summer temperatures. CVS Chinook often do not stay in the same pool for the duration of summer, but move upstream from pool to pool as stream flow conditions allow, often spawning in the tailwaters of their final holding pool (Moyle 2002). Spawning behavior is similar to that of coho salmon, with females digging redds in the appropriate substrate and large, dominant males fighting off other males prior to spawning. The gametes of the dominant males are often supplemented, however, by those from one or more jacks (two-year-old males) that spawn by sneaking into the nest with the mating pair from downstream and releasing their milt just as the female releases her eggs (Moyle 2002, Williams 2006). The "sneaker" male spawning strategy supports greater genetic diversity in the population by reducing one-male to one-female mating and contributes to variable life history strategies.

CVS Chinook maintain a high degree of plasticity in their age at spawning. A small proportion of the run can be made up of jacks, or small male fish that return to the rivers to spawn after only a single year or two in the ocean, instead of the more typical three or four years at sea for most Chinook in the Central Valley (NMFS 2014). Age at spawning for CVS Chinook salmon generally varies from age 2 to age 4 (NMFS 2014); approximately 69% of the spawners returning to Butte Creek in 2005 were estimated to be age-4 adults (McReynolds et al. 2006), while in 2006, age 4 fish accounted for 75% percent of returning spawners to Butte Creek (McReynolds et al. 2007). Pre-spawn mortality studies on Butte Creek from 2001-2006 estimated average spring-run FL to be 735 mm and 785 mm for females and males, respectively (McReynolds et al. 2007). Observations of sexually mature 1-year old male salmon parr (those never going to sea) have also been made in watersheds supporting spring-run life histories (C. Jeffres, UC Davis, unpubl. obs.). These fish spawn in much the same way as jacks (Quinn 2005). It is thought that some of these "precocious parr" - whose enormous testes account for ~21% of their body weight - may actually survive to spawn a second time (Moyle 2002). The variability in male reproductive strategies ensures that around 90% of eggs are fertilized and that genetic diversity of the population is maintained (Moyle 2002).

Box 1. Chinook salmon life history strategies

Chinook salmon have a variety of life history adaptations that allow them to persist through variable environmental conditions, but are most often divided into two main life-history strategies: 1) stream-type and 2) ocean-type. Initially, these types simply distinguished salmon that spent a few months or a winter in fresh water before migrating to sea, as revealed by growth patterns in their scales (large spaces between rings indicated fast growth while at sea, while closely-spaced rings indicate slower growth while in fresh water, Gilbert 1913). Later, other characteristics were associated with these types, which have been named based on their timing of maturation. Generally speaking, ocean-type refers to a population in which juveniles begin their migration to sea soon after emerging from redds, spend less than a year in fresh water, return as mature adults, and spawn soon after reaching spawning grounds. The overwhelming majority of CVS Chinook juveniles exhibit this life history strategy (F. Cordoleani, NMFS, pers. comm. 2017). Alternatively, stream-type Chinook stay in streams for over a year before initiating seaward migration, and re-enter fresh water in spring as sexually immature fish. These fish mature in the stream over summer before spawning in early fall.

While at sea, ocean-type Chinook tend to forage close to the coast, whereas stream-type Chinook tend to venture farther out to forage in the open ocean. Stream-type Chinook displaying these characteristics predominate north of 55º latitude (near the southern edge of Alaska), while ocean-type Chinook predominate south of 55° latitude (Healey 1991). However, Williams (2006) noted that juvenile CVS Chinook from populations south of the Columbia River, including in California, often migrate to sea in their first year, and tend to forage in coastal waters. This is consistent with the development of a stream-type life history from an ocean-type lineage, which Healey (1991) recognized as a possibility and which has been demonstrated with CV Chinook that were transported to New Zealand (Unwin et al. 2000). It is also consistent with experiments showing that normally stream-type juvenile Chinook will behave like ocean-type fish if they are exposed to a short day photoperiod when they emerge (Clarke et al. 1992; Williams 2006). Healey (1991) postulated that the steam-type life history represents an Asian or Beringian lineage that had been separated from a Cascadian ocean-type lineage during the last glaciation event. There remains some confusion in the literature over how to apply the stream-type/ocean-type nomenclature, but it seems safest to use it only in reference to juvenile migration patterns, because they are not necessarily linked to adult behavior (Williams 2006).

Embryos incubate in the gravel for 40-60 days (temperature dependent) and remain in the gravel as alevins for an additional 4-6 weeks until the yolk-sac is absorbed and fry begin to forage (Williams 2006). Juveniles feed mainly on zooplankton, benthic invertebrates, terrestrial drift, and larvae of other fishes, especially suckers (Moyle 2002). Rearing and migration timing is extremely variable in CVS Chinook, ranging from 3 to 15 months, presumably as a result of limited rearing habitat available in the upper watersheds and variable flow conditions (Stillwater 2006, McReynolds et al. 2007). Some juveniles begin their emigration as fry mere hours after emerging from the gravel; in Butte Creek, most juveniles have already migrated downstream as fingerlings by the end of February (McReynolds et al. 2007). Most begin smoltification after a few months of stream rearing and move to salt water as sub-yearlings. A third type remains in the stream for a year, over-summering in natal streams before beginning downstream emigrations (Hill and Webber 1999, Stillwater 2006). This complex of life history strategies can generally be thought of as bet-hedging behavior. Fish that remain in freshwater habitats near where they were hatched may have a reduced food supply due to comparatively lower productivity or intraspecific competition, but may enjoy increased survival rates until emigration. They are also likely to be larger at smolting, resulting in increased survival in downstream habitats. Another option to increase size and survival while migrating downstream is to spend time on floodplains, where water is often slower, warmer and more turbid, and full of abundant prey (see below).

Outmigrating juveniles from Mill, Butte, and Deer creeks average approximately 40mm FL between December and April, and reflect a prolonged emergence of fry (Lindley et al. 2004). Conversely, Ward et al. (2003) found the majority of spring-run migrants to be fry moving downstream primarily during December through February in Butte Creek, and that these movements appeared to be influenced largely by flow. Small numbers of spring-run juveniles remained in Butte Creek to rear and migrate as sub-yearlings later in the spring. Juvenile emigration patterns in Mill and Deer creeks typically exhibit a later young-of-the-year migration and an earlier yearling migration than juveniles in Butte Creek (Lindley et al. 2004). There are generally more yearling juveniles in Mill and Deer creeks than in Butte Creek, which is probably due to the colder spring and summer water temperatures found there (F. Cordoleani, NMFS, pers. comm. 2017). By contrast, in the Feather River the bulk of juvenile emigration occurs during November and December (USACE 2012).

As they move downstream, young Central Valley Chinook use the lower reaches of non-natal tributaries and the shallow edges of the mainstem to seek relief from high flows, forage, and refuge from predators. Downstream migration serves not only to disperse juveniles to the ocean, but it gives them access to temporary habitats with warmer temperatures and abundant food, such as aquatic invertebrates and larval fish, that allow for rapid growth. Sommer et al. (2001), Jeffres et al. (2008), and Katz et al. (2017) indicate significantly higher growth rates for juvenile Chinook rearing on floodplains as opposed to those rearing in riverine habitats. Floodplain production and temperatures are considerably higher in these lowland habitats and provide an important resource for out-migrating juveniles. The extensive levee building along the Sacramento River over the last century has prevented Chinook juveniles from accessing these habitats, except in a few places such as the Yolo and Sutter bypasses, when high winter and spring flows provide access. Juveniles can rear for 1-3 months in the bypasses, putting on significant weight, which is directly tied to increased survival at sea (Williams 2006).

CVS Chinook apparently rear on available floodplains and tidal marsh habitat of the Sacramento-San Joaquin Delta and may also utilize the shallow habitats of San Pablo Bay (Williams 2006). Juvenile usage of tidal marshes, mudflats, and bays of the San Francisco Estuary is not well understood or studied (Williams 2006). There is considerable inter- and intra-annual variation in rearing habitat use that varies in part with fish size. Young-of-year juvenile spring-run Chinook are almost all ocean-type fish; these fish enter the estuary at a smaller size than do the much less common yearling, or stream-type juveniles that spend more time growing in the upper watershed (F. Cordoleani, NMFS, pers. comm. 2017). The smaller ocean-type juveniles need to spend more time foraging and exploit available resources in the estuary before migrating out to sea (Williams 2006). Food type and availability varies with habitat, but aquatic and drift insects, amphipods, copepods, and small crustaceans are generally available throughout the brackish regions of the estuary (MacFarlane and Norton 2002). Studies from the early part of the 20th century in the lower estuary indicate that young Chinook frequently appeared in trawls and beach seines, but the San Francisco Bay has changed significantly since then and the current ecosystem does not resemble the historical ecosystem (Scofield 1913, Williams 2006). As a result, it is likely that smolt habitat usage in the San Francisco Estuary has changed correspondingly.

We can infer from other studies, such as from the Columbia River Basin, that estuaries can play an exceedingly important role in both salmon and steelhead smolt growth and survival; size of smolts upon ocean entry appears to be a strong determinant of survival in the first year at sea (Williams 2006). Juvenile CVS Chinook that rear on the Sutter Bypass floodplain will likely emerge from that habitat at sizes larger than 70 mm FL, nearly double their size at emigration from Butte, Mill, and Deer creeks, and can then proceed to the estuary quickly without needing to delay and rely on what rearing habitat may be available in the Delta (Hill and Webber 1999). Tagging data of spring-run Chinook smolts in Sutter Bypass in May indicate that floodplain-reared fish average around 107 mm, compared to the much smaller 70mm juveniles emigrating at the same time that had not reared on the floodplain from Mill Creek (NMFS *In prep.*). While there have been few studies of juvenile CVS Chinook use of estuarine habitats, the low numbers of juveniles encountered throughout the bays and lower tidal marshes and the lack of growth observed in those reaches since the turn of the century reflect the immense changes and habitat alteration that have taken place. MacFarlane and Norton (2002) studied CVS Chinook migration through the Delta and San Francisco Estuary, and documented average juvenile migration durations of approximately 40 days through the estuary, at an estimated rate of 1.6 km/day. The bulk of tidal marsh and creek habitats have been leveed and channelized for drainage, water deliveries, and navigation. Meanwhile, water transfers at the federal and state Delta pumps have drastically altered the hydrology, salinity, and turbidity of the lower Delta, impacting migration duration. Additionally, it is possible that predation from fish and birds affect Chinook survival and behavior in these habitats (NMFS 2014).

The majority of a CV spring-run Chinook salmon's life, including its growth and weight gain, now typically occurs in the ocean, whereas a significant portion of growth presumably historically occurred in the San Francisco Estuary (MacFarlane and Norton 2002). Once smolts arrive at sea in the late spring/early summer, they begin to feed on a variety of crustaceans, euphausiids, and small prey fishes and larvae (MacFarlane and Norton 2002, MacFarlane et al. 2005). As young Chinook grow larger, their diet shifts from crustaceans to predominantly fish, such as herring, anchovies, juvenile rockfish, and sardines (Moyle 2002). Smolt body condition (K), which is a ratio of weight to length of fish, after the first summer of foraging in the ocean is thought to be a good predictor of juvenile survival over their first winter (Williams 2006). It appears that a certain threshold of food abundance must be reached to ensure high survival, although in the warmer regions surrounding San Francisco Bay, this threshold may not be as absolute as in the more northerly regions, due to less variety and abundance of food resources (MacFarlane and Norton 2002). MacFarlane and Norton (2002) found that mean growth in length (0.18 mm/day) and weight (0.02 g/day) of juvenile Chinook in estuary habitat paled in comparison to estimated daily growth of 0.6 mm/day and 0.5 g/day observed while these fish were in the ocean; at the same time, body condition declined while they were in the estuary, but improved markedly once these fish had a chance to feed in the ocean. This phenomenon can be accounted for by the California Current system that prevails outside San Francisco Bay, where upwelling, especially in the Gulf of the Farallones, creates a rich foraging area for young salmon.

It is probably for this reason that newly arrived salmon mostly feed near shore, rather than farther out in the open ocean.

Size at ocean entry differs between stream-type and ocean-type fish, with stream-type fish generally being larger than their ocean-type counterparts due to their longer growth in freshwater. Once in the ocean, growth rates are similar, but the sizes at entry can determine ultimate lengths of adults returning to spawn at a given age (Moyle 2002). Commercial and sport fisheries also select for smaller size by removing larger and older fish from the population, as current regulations stipulate adults must measure > 51cm North of Point Arena, and > 61 cm from Point Arena South, which results in smaller and younger returning adults (CDFW 2017a). Thus natural factors may favor survival of larger CVS Chinook salmon, while fisheries may favor survival of smaller individuals.

Habitat Requirements: Chinook salmon use a variety of habitats during their lives. In general, water temperature determines their presence in a particular stream segment in fresh water. Preferred holding habitat is characterized by maximum weekly average temperatures less than 21ºC (Moyle 2002), although there is evidence that CVS Chinook in some areas may tolerate slightly higher temperatures, such as in Butte Creek, tributary to the Sacramento River.
The upper limit of temperature tolerance for adult CVS Chinook appears to be between 21 and 24ºC (Williams 2006). Evidence from Butte Creek indicates consecutive days with daily mean temperatures >/= 21ºC increase adult mortality. Eggs are less tolerant and thus adults wait until stream temperatures drop to around 13-15°C in the fall before spawning, while juveniles are more tolerant than eggs or embryos (Williams 2006). Preferred spawning habitat seems to be at depths of 25-100 cm and at water velocities of 30-80 cm/sec, though CVS Chinook have been observed digging redds and spawning at depths from a few centimeters to several meters and at water velocities of 15-190 cm/sec (Williams 2006). Redds cover 2-10m^2, where the loosened gravels permit steady access of oxygen-saturated water.

Adult CVS Chinook require deep pools with good cover for holding over the summer; most of this habitat lies at elevations between 150 and 500 m, but from 500-1,000 m for Deer and Mill creeks (NMFS 2016, M. Johnson, CDFW, pers. comm. 2017). Most spawners reach these areas by July and select deep (> 2m) pools with bedrock bottoms and moderate velocities (15-18 cm/sec) and with abundant hiding places such as rock ledges, bubble curtains, and woody debris (Moyle 2002). Spawning begins once water temperatures decrease to around 15°C. Spawning gravel varies in size. The most important factor for spawning site selection appears to be good hyporheic (subsurface) flow that provides relatively cooler, oxygen-saturated water to support embryos in gravels (Moyle 2002).

Ocean-type fry spend more time in the lower reaches of rivers and in the Sacramento-San Joaquin Delta than stream-type fish, foraging in the shallows and rearing on the floodplains of the Central Valley before undergoing smoltification before ocean migration (Williams 2006). Juveniles that emigrate as yearlings are more likely to become smolts on the downstream migration, and not spend as much time in the San Francisco Estuary. In the ocean, fish from the Central Valley stay close to the coastal shelf, where upwelling provides rich foraging opportunities (Williams 2006).

Distribution: CVS Chinook salmon historically ranged throughout the Central Valley in both the Sacramento and San Joaquin watersheds. The National Marine Fisheries Service five-year status review for CVS Chinook (NMFS 2016) identified 18 or 19 independent historical populations of CVS Chinook ranging from the Pit River in the north to the southern reaches of the upper San Joaquin. These populations inhabited five distinct geologic/hydrologic regions: 1) Basalt and Porous Lava, 2) Northern Sierra Nevada, 3) Northwestern California, 4) Southern Sierra Nevada, and 5) Central Valley domains. For now, the ESU boundary for CVS Chinook salmon consists of the Sacramento River Basin downstream of impassible barriers; should a run of self-sustaining, CVS Chinook become re-established in the San Joaquin River, the boundaries could be updated to include this population (NMFS 2016).

In the Sacramento drainage, CVS Chinook once ranged into the Fall, Pit, McCloud, and upper Sacramento Rivers, from which they have been excluded since the 1940s by Shasta Dam. Western Sacramento River tributaries once inhabited by CVS Chinook include: Stony, Thomes, Cottonwood, and Clear creeks. Eastern tributaries that historically supported CVS Chinook include: Cow, Battle, Antelope, Mill, Deer, Big Chico, and Butte creeks as well as the Feather, Yuba, American, and Mokelumne rivers. Today, some CVS Chinook are still found in Battle Creek and in the Sacramento River below Keswick Dam, but current distribution of viable populations is limited to just a handful of streams in the northern Sierra Nevada Region (Lindley et al. 2016). This includes naturally reproducing populations in Mill, Deer, and Butte Creeks. CVS Chinook also occur on a regular basis in some of the smaller tributaries, such as Antelope, Big Chico, Little Chico, Beegum, and Clear creeks, but these populations are presumably not self-sustaining (Lindley et al 2007, Figure 1).

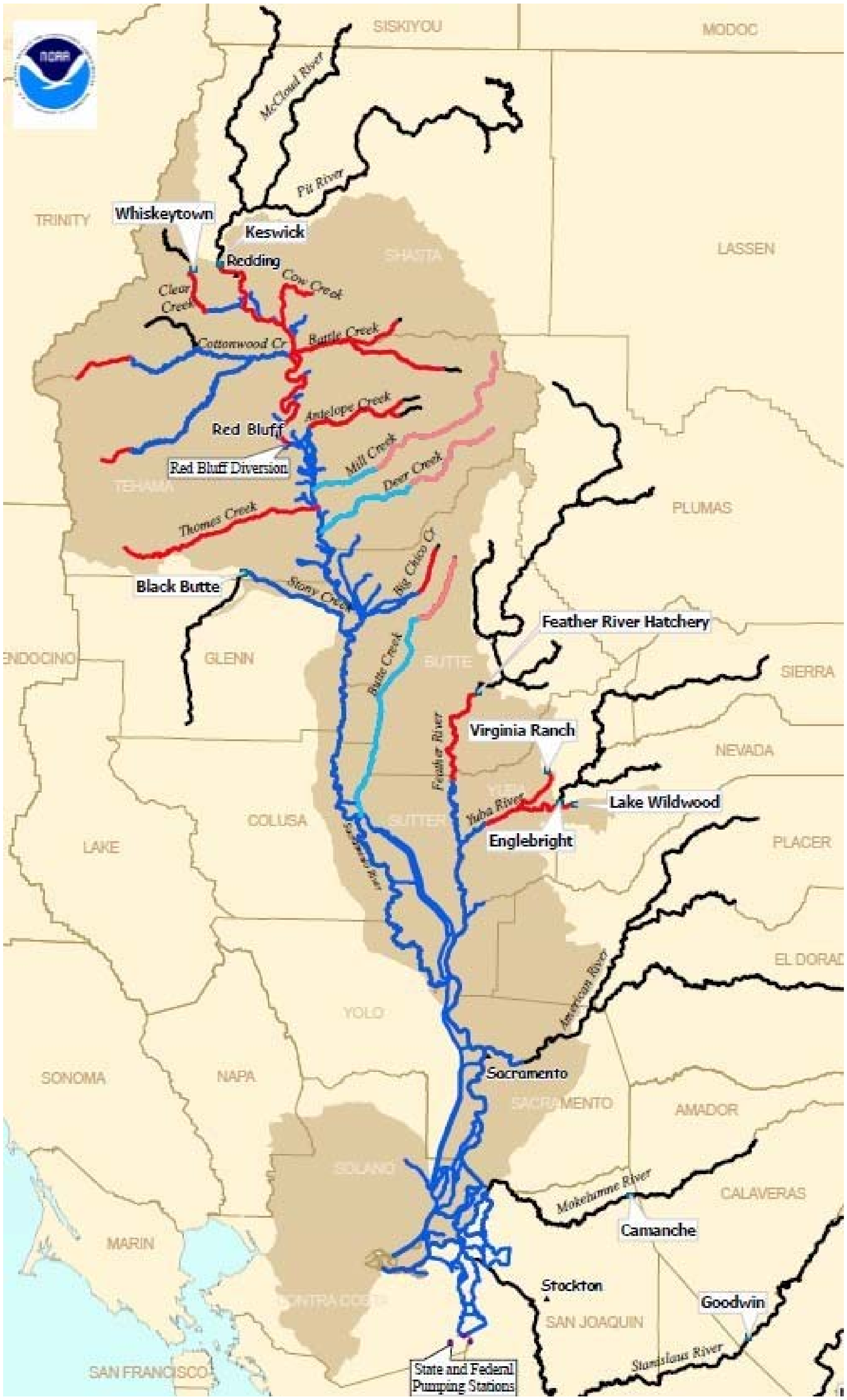


Figure 1. Historical and extant range of CVS Chinook salmon in the Sacramento Basin, California. Black = historical migration range; blue/light blue = current rearing or migration range; red/pink = current spawning range; blue squares = major dams, diversions, or hatcheries; purple squares = major pumping facilities. From NMFS 2014 Fig. 2-4, pg. 32.

In the San Joaquin drainage, lingering snow and glaciers at high elevations created a long spring hydrograph that favored CVS Chinook, making them the dominant run in the region. They apparently historically ascended the Kings, upper San Joaquin, Merced, Tuolumne, and Stanislaus Rivers, although pre-dam records for the latter three rivers are scarce (Yoshiyama et al. 1998). All San Joaquin drainage runs of CVS Chinook were thought to be extirpated as a result of a combination of factors including dam construction, water diversions for agriculture, and land use practices (NMFS 2014). More recent study (Franks 2015, FishBio 2015) has revealed Chinook exhibiting spring-run migration timing have been returning to the Stanislaus, Tuolumne, and Merced rivers, though their origin is unknown (Figure 2).

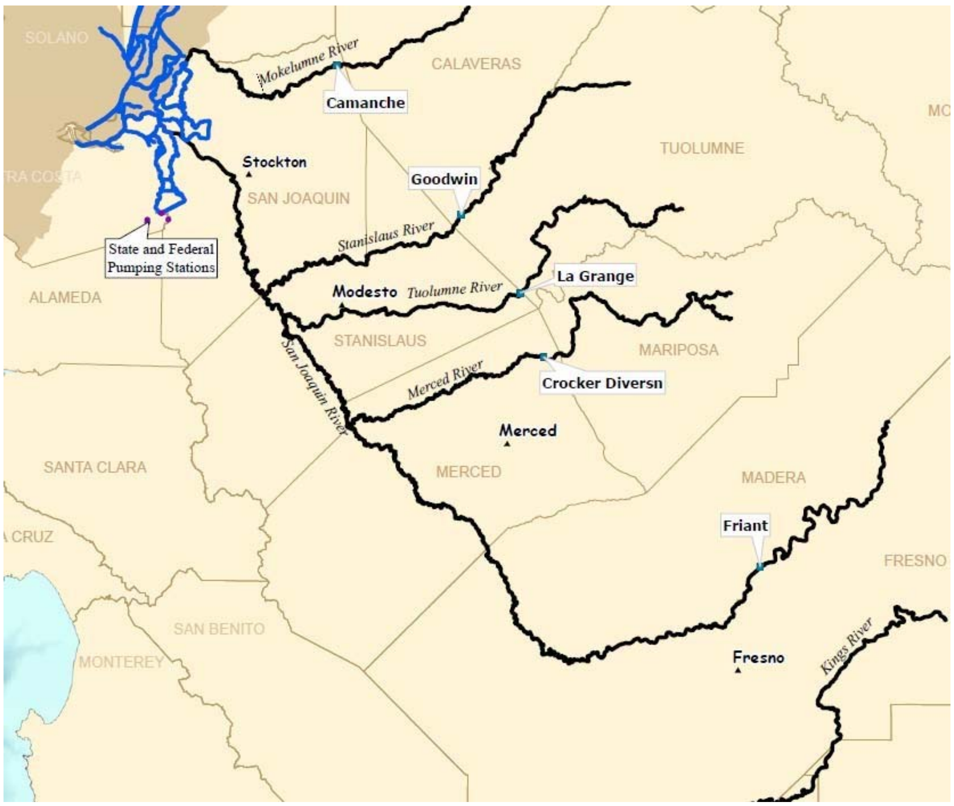


Figure 2. Historical and extant range of CVS Chinook salmon in the San Joaquin Basin. Black = historical migration range; blue = current rearing/migration range; blue squares = major dams, diversions, or hatcheries; purple squares = pumping facilities. From NMFS 2014 Fig. 2-4, pg. 32.

Within these regions, CVS Chinook distribution is limited significantly by both accessibility of habitat and water temperature. Many of the streams in the northern end of the ESU range are fed by springs, where precipitation falls mainly as rain rather than snow. These streams tend to have steady, year-round flows of cold water, which provided excellent habitat for over-summering CVS Chinook and decreased flow variability. Conversely, the bulk of the precipitation in the Sierra Nevada regions, especially in the south, falls as snow and spring-fed systems are less prevalent. This creates a sharp peak in the hydrograph during the late spring and early summer months when snowmelt is at its highest, then tapers off over the summer months into the early fall, when ambient and stream temperatures cool enough for spawning. The historical timing of high flows in the spring and early summer provided enough flow for CVS

Chinook to reach their summer holding areas. CVS Chinook upstream migrations were historically only truncated by natural impassible barriers, such as waterfalls and boulder fields that limited their access to higher-elevation, cooler reaches.

Trends in Abundance:

Overview. Annual abundance of CVS Chinook salmon populations has varied considerably from 1970 to 2012, from highs of around 31,000 to lows around 3,000 (NMFS 2014, Figure 3), including fish of hatchery origin in the Feather River. However, despite an upward trajectory in 2012, returns of fish from 2012-2014 again showed a downward trend, which carried over to very poor returns in 2015 (NMFS 2016).

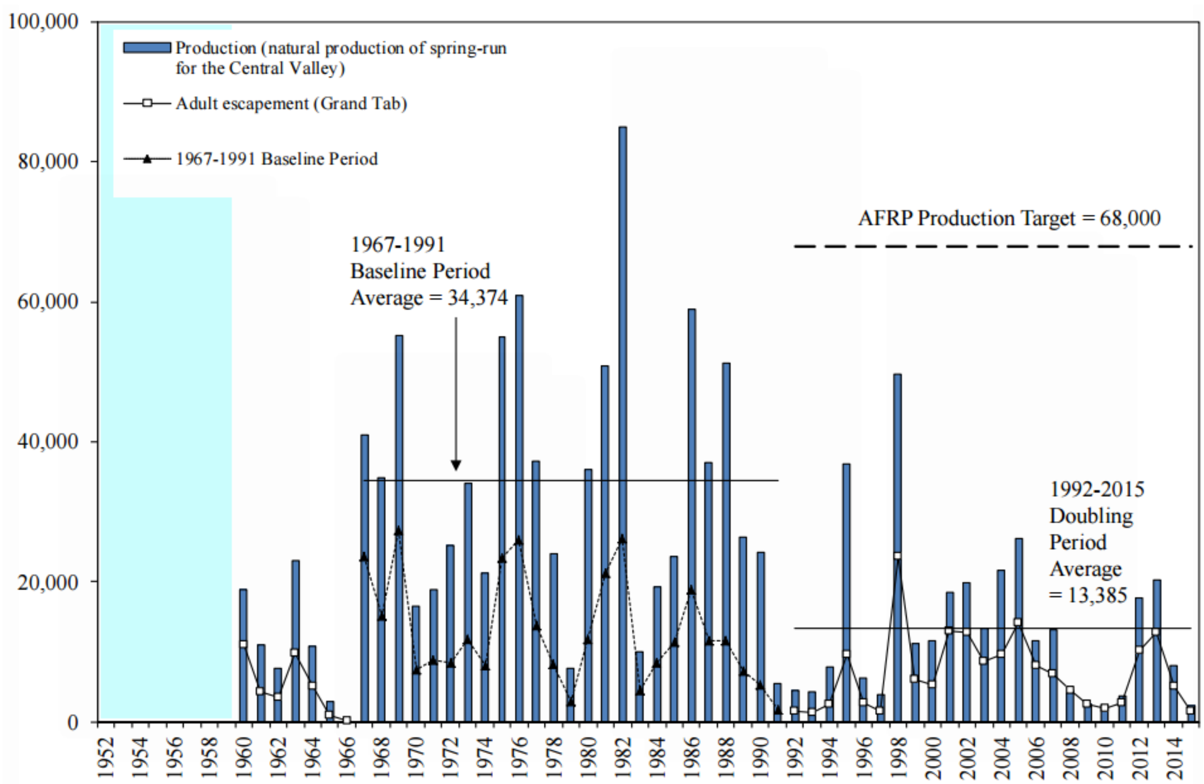


Figure 3. Estimated yearly adult natural production, and in-river adult escapement, of spring-run Chinook salmon in Central Valley rivers and streams, 1952-2015. Based on CDFW's Grandtab escapement estimates. From USFWS Anadromous Fish Restoration Program 2017, Fig. 5, pg. 5.

Sacramento Basin. CVS Chinook have been extirpated from the vast majority of their historical range in the Sacramento Basin due to habitat alteration. 19th century combined run sizes were probably in the range of 1 million fish per year, +\- 500,000 (Yoshiyama et al. 1998); CDFW (1998) estimated historical runs of around 600,000 CVS Chinook in the Central Valley from the 1880s to the 1940s. The three extant populations in Mill, Deer, and Butte creeks in the Northern Sierra Nevada diversity group saw an upward trend in abundance from 2010-2014. The Battle and Clear creek populations have been increasing recently and climbed into the moderate extinction risk category, (NMFS 2016) until ongoing drought brought significant declines across CVS Chinook populations (NMFS 2016, Figure 4), especially during 2015-16. The latest counts in 2016 (331 on Deer, 175 on Mill creeks) are likely the lowest ever recorded (M. Johnson, CDFW, pers. comm. 2017). NMFS (2016) points out that once populations dip below 500 individuals over two generations, as is likely the case for Deer and Mill populations, the extinction risk becomes high.

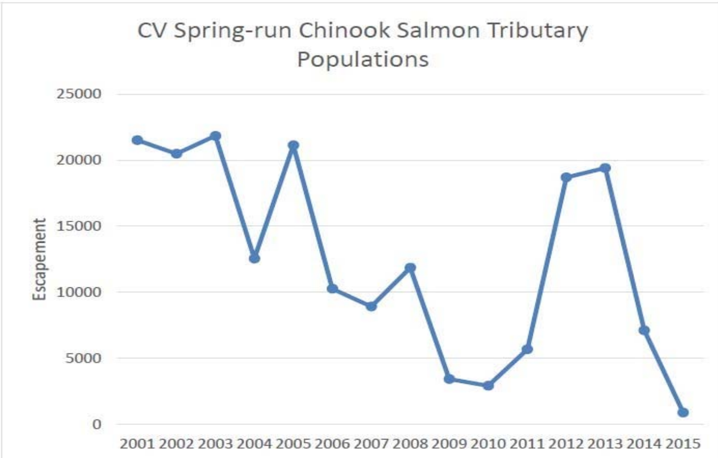


Figure 4. Combined escapement estimates for CVS Chinook salmon in Butte, Mill, Deer, Battle, and Clear creeks based on carcass surveys, 2001-2015. From NMFS 2016, Fig. 2, pg. 15.

Populations in Cottonwood, Antelope, and Big Chico creeks have remained at or near zero since 2007. Meanwhile, the Yuba River (with a few hundred to perhaps a few thousand fish) remains at high extinction risk as a result of FRFH influence. Aerial redd surveys conducted by CDFW indicate that a small number of CVS Chinook spawn in the mainstem Sacramento in September between Keswick Dam (Rkm 483) and Red Bluff Diversion Dam (Rkm 391) (NMFS 2016). In contrast, about 1,000-20,000 fish return to the Feather River annually, though these numbers are supported significantly by FRFH (NMFS 2016). The hatchery has been releasing about 2 million juvenile Chinook per year from 2006-2016 at various locations, including in-river and downstream locations in net pens within San Pablo and San Francisco bays (PSMFC 2017, Figure 5). These juveniles have been introgressed with Central Valley fall-run Chinook through hatchery practices over time (CHSRG 2012).

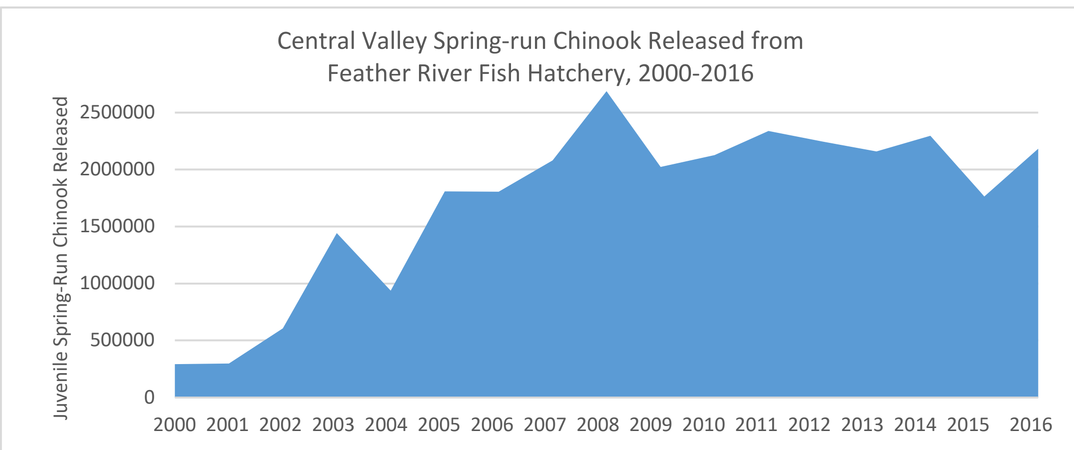


Figure 5. CVS Chinook salmon releases from Feather River Fish Hatchery, 2000-2016. Data from PSMFC 2017: http://www.rmpc.org/.

While hatchery releases show an increasing trend since 2000, returns of adults and jacks to the Feather River Fish Hatchery have remained fairly constant, ranging from about 1,000-4,000 fish per year (NMFS 2016). Since 2007, FRFH has adjusted their operations to tag and release fish that ascend the ladder from September 15 to June 30, ensuring that fish exhibiting spring run timing can be identified later when spawned. Nearly 1,000,000 juvenile fish are released into net pens in San Pablo Bay near Benicia, CA each year, which presumably increases in-river survival but leads to increased rates of straying to other Central Valley streams due to reduced opportunities for imprinting (NMFS 2016). From 2004-2010 on the Yuba River, hatchery-origin fish from FRFH accounted for an average of about 20% of the total annual run of CVS Chinook salmon passing upstream of Daguerre Point Dam (USACE 2012).

San Joaquin Basin. Fry (1961) estimated that approximately 50,000 CVS Chinook migrated up the San Joaquin River annually before completion of Friant Dam below Millerton Reservoir (Rkm 431) in 1942. By the 1950s, nearly the entire San Joaquin Basin run was extirpated, leaving only remnants in the Merced River (Yoshiyama at al. 1998). Since 2014, considerable investment has been made to attempt to re-establish a self-sustaining population of CVS Chinook in the San Joaquin River (www.rmpc.org). As part of the historic San Joaquin River Restoration Agreement, an average of 57,000 CVS Chinook originally from Feather River Fish Hatchery and raised at a temporary conservation hatchery facility at Friant Dam, have been released at the confluence with the Merced River (Rkm 189).

In addition, there have been reports of adult Chinook salmon returning from February through June to the Mokelumne, Stanislaus, and Tuolumne rivers that display spring-run life history strategies for holding and spawning (Franks 2014, Workman 2003, FishBio 2015, NMFS 2016). In 2013, 114 adult Chinook were counted on the video weir on the Stanislaus River between February and June, with only 7 fish (6%) without an adipose fin, indicating the majority are presumably natural-origin fish or offspring of natural-origin fish (FishBio 2015). The origins of these fish and viability of putative populations are unknown at this time.

Summary. Annual estimates of CVS Chinook salmon numbers in the past 25 years have varied from 3,000 to 30,000, including FRF Hatchery fish. Presumably such numbers would have large confidence intervals around them if the intervals could be calculated. If the FRFH component is ignored, numbers of wild CVS Chinook seem to range between 1,000 and 20,000 spawners, with a general downward trend.

Factors Affecting Status: Major factors affecting, or potentially affecting, the status of CVS Chinook in the Central Valley are discussed below. For a full discussion of more general factors, see Central Valley fall-run Chinook salmon.

Dams. Dams are the major cause of the widespread extirpation of CVS Chinook salmon. Currently, dams block access to over 90% of their historical spawning and summer holding areas, including almost all of the mainstem Sacramento, all of the San Joaquin drainage, the northern Sacramento basin, and central Sierra Nevada streams such as the Yuba, Feather, and American Rivers (Yoshiyama et al. 1998). All but three historical spawning areas are either behind impassable dams or are strongly impacted by dams (Mill, Deer, Butte creeks), and even these watersheds have been negatively affected over the last century by water diversions, small dams, and between-basin water transfers to support agricultural and urban water usage (Yoshiyama et al. 1998). Keswick (Rkm 483) and Shasta (Rkm 505) dams on the Sacramento River have fundamentally altered the hydrograph and character of the river downstream, significantly limiting the ability of the river to support salmon of any run timing (see winter-run

Chinook account). Historically, Red Bluff Diversion Dam (Rkm 391) utilized gates to divert water for agricultural and urban uses that created impediments to fish passage, but the gates have remained open since 2011 to allow passage for salmon, steelhead, and sturgeon in the river (USFWS 2014).

Large dams change the flow patterns of rivers, reducing the length of spring outflow events that help juveniles emigrate and attract adults upstream, and limit the ability of floodplains to properly function by providing opportunities for rapid growth of juvenile salmonids. Coldwater releases from dams provide cool temperatures to sustain CVS Chinook in some cases, but also facilitate easy hybridization with fall-run fish because of the loss of the historical geographic segregation between them. This relatively recent spatial overlap in spawning may drive the lack of genetic differentiation between fall- and spring-run Chinook in the Feather River.

The DeSabla-Centerville dams on Butte Creek operated by Pacific Gas & Electric Company are scheduled for relicensing in the near future, but PG&E has withdrawn their application. The future operation of these dams and associated diversions will have important consequences for the long-term persistence of CV spring-run Chinook.

Rural/ residential development. Development of all kinds along the Sacramento River and its tributaries impact CVS Chinook rearing in the main river and its tributaries through polluted run-off, sedimentation, loss of riparian habitat, and small diversions. The effects of such actions, however, are poorly documented.

Logging. Logging has and continues to be an important economic activity in the watersheds throughout the historical CVS Chinook range. Much of the headwaters of these watersheds lie within National Forest boundaries, including (from North to South) Shasta-Trinity, Lassen, Plumas, Tahoe, Eldorado, Stanislaus, and Sierra national forests. Legacy impacts to streams from logging and its associated road building have resulted in erosion, landslides, and loss of riparian vegetation and large woody debris inputs to streams that provide refuge for multiple life stages of salmonids across the ESU range (Meehan 1991).

Grazing. Cattle grazing occurs throughout most of the Central Valley watersheds where CVS Chinook reside. Basin-wide impacts include erosion from bank trampling, loss of meadow habitat, and loss of riparian vegetation with a resulting increase in water temperature and decrease in water quality. This in turn can alter the macroinvertebrate community that is relied on as food for both juveniles and over-summering spawners. Many of these impacts have been reduced in recent years (e.g., through the fencing of meadow streams such as in Deer Creek Meadows) by improved management of landowners and other partners, but livestock grazing is still a major land use throughout the ESU range.

Agriculture. There are numerous diversions along the Sacramento River to support agricultural uses, which can potentially entrain CVS Chinook fry and smolts. The larger diversions are all screened and presumably offer some degree of protection from entrainment, while smaller diversions on the main river mostly do not need to be screened (Moyle and Israel 2006). The large State Water Project and the federal Central Valley Project pumps in the South Delta are another problem for salmon. They ultimately pull Sacramento River water towards them to support agriculture in the southern Central Valley, as well as to provide water for Bay Area and Southern California cities. In 2009, the National Marine Fisheries Service issued a formal Biological Opinion on the impacts of this pumping on listed salmon, sturgeon, and killer whales in a jeopardy determination that triggered exploration and implementation of alternatives to business-as-usual operations (NMFS 2017). See Central Valley fall Chinook account for further discussion of this factor. Finally, obsolete fish screens, diversions, dams, fish ladders, inadequate flows, levee construction, floodplain simplification, and hydroelectric projects on Battle and Butte creeks and the Feather and Sacramento rivers significantly limit spawning and rearing habitat for CVS Chinook by fragmenting and altering flow and habitat regimes (NMFS 2016). In addition, the variation of water operations deviates from natural hydrographs. In winter and spring, dams hold back flows to store water, then deliver it at artificially high flows to consumers downstream. These low springtime flows harm juvenile and adult spring-run Chinook by reducing water quality (higher water temperatures and contaminants) and increased predation (from lower velocity flows and warm water that favors alien species, M. Johnson, CDFW, pers. comm. 2017).

Historically, the biggest impact of agriculture on CVS Chinook salmon was construction of the massive levee system in the Central Valley in the 19th and early 20th centuries to prevent flooding of agricultural fields and towns (Kelley 1989). Levees caused the lower river to down-cut as large sediment loads from hydraulic mining came downstream (which had raised historical river levels and exacerbated problems with flooding). The result was loss of floodplain and backwater habitat that was historically important for rearing juvenile Chinook salmon. Today this habitat is largely absent along the channelized Sacramento River and diked Sacramento-San Joaquin Delta. Patches of slow-moving rearing habitat are sparse outside the main river, where exposure to predators is high. In wet winters and springs when the Yolo Bypass is flooded, presumably some CVS Chinook take advantage of the favorable rearing conditions present. The impact of loss of this floodplain rearing habitat on CVS Chinook is poorly documented, but the combination of fewer opportunities for rapid growth and more constant exposure to predators in the main river channels may greatly reduce out-migrant survival.

On Deer Creek, Stanford Vina Ranch Irrigation Company (SVRIC) and Deer Creek Irrigation District (DCID) draw enough water out of the creek to stop upstream migration of adults, leaving less than 5cfs of flow during some intensive irrigation windows (CDFW 2014). On Mill Creek, Ward Diversion Dam reduces instream flows and increases stream temperatures, creating sub-optimal conditions for migrating and rearing spring-run Chinook (CDFW 2014b). Deer and Mill Creek spring-run juveniles out-migrate at later dates and smaller sizes compared to other CV Chinook stocks (Johnson and Merrick 2012). This likely puts Deer and Mill spring-run at greater risk to low flows and warm water temperatures in diverted stream sections and the mainstem Sacramento River during spring.

On the Feather River, water is diverted from Oroville Reservoir and warmed in a shallow reservoir (Thermalito Afterbay) for rice farming, with excess warm water returned to the river. The influx of warm water can raise instream temperatures to lethal levels for over-summering and juvenile CVS Chinook. Agricultural return waters also contain pesticides and other contaminants, which may negatively affect juvenile Chinook health and survival through reduced water quality.

On the San Joaquin River, diversion dams, levees, and similar water conveyance and control projects such as Chowchilla Bypass eliminated much of the rearing habitat for juvenile salmon that persisted through the summer by quickly draining coldwater flows from the high Sierra snowmelt and from artesian groundwater. These rearing habitats were found in the braided side channels and are still faintly visible on aerial photographs and in floodplains of the basin.

For more than sixty years, the mainstem San Joaquin has been dry as a result the cut-off of flows combined with overdraft of surface and ground water in the region, especially in the vicinity of Chowchilla Bypass. This has effectively eliminated the migration and rearing capacity of the mainstem San Joaquin River altogether, significantly reducing CVS Chinook abundance by removing the southernmost population. As a result of over eighteen years of lawsuits and a settlement agreement in September 2006, the San Joaquin River Restoration Program was adopted and has been implemented by NMFS, U.S. Fish and Wildlife Service, CDFW, water users, the U.S. Bureau of Reclamation, and other stakeholders. Under this program, flows have been secured from Friant Dam below Millerton Reservoir to provide access to spawning, rearing, and migration habitat for CVS Chinook. Using strategic releases of water from Friant Dam, the Settlement Parties aim to make volitional migration possible once again for an experimental population of CVS Chinook from the Dam site downstream to the confluence with the Merced River. CVS Chinook juveniles from broodstock at the Feather River Fish Hatchery are being raised at a temporary hatchery facility downstream of Friant Dam. Additional information on this program can be found at (San Joaquin River Restoration Program 2017).

Mining. Hydraulic gold mining was presumably an initial factor in CVS Chinook decline in the late 19th century, which radically altered holding areas across most of the Sierra Nevada foothills (Merced to Feather rivers, as well as other areas). Historical mining during the California Gold Rush resulted in the destruction of many of the streams used by all runs of Chinook, but especially CVS Chinook, which require high-quality habitat and cold water all year round. Hydraulic mining washed millions of tons of sediment into streams, covering spawning gravel and destroying habitat. Historical records indicate that runs in rivers subjected to hydraulic mining were extirpated for some time until conditions improved and the salmon were able to re-colonize areas not blocked by dams (Williams 2006). Significant scarring and habitat alteration resulting from mining 150 years ago can still be seen today in streams and rivers throughout the southern and northern Sierra Nevada areas (e.g. Yuba River); high sediment loads in rivers after winter storms are a continuing legacy impact of these activities.

Toxic mining wastes, such as heavy metals like mercury, mainly from abandoned mines, also have legacy effects on CVS Chinook. The principal threat today associated with mining in the ESU range is the potential for a major spill of highly toxic waste from Iron Mountain Mine, a U.S. EPA superfund site about 14.5km Northwest of Redding (Shasta Co.). If the check dam on Spring Creek that currently contains the contaminants should fail, a plume of toxic water could be flushed down the Sacramento River, with lethal consequences for many fish.

Estuarine alteration. The San Francisco Estuary is a very different ecosystem today than it was when Central Valley Chinook salmon evolved, leaving few opportunities for foraging and rearing salmon and the kind of growth that historically prepared smolts for the rigors of ocean migration (MacFarlane and Norton 2002). Historically, juvenile CVS Chinook would have arrived in an estuary that was a complex of tidal marshes, with many shallow channels, rich in food and refuge. In this system, they could physiologically adjust to changing salinities slowly, while finding abundant food and cover to compensate for the stress of emigration. Today, most of the tidal marshes are gone, food resources are diminished, and exposure to predators, especially in the form of invasive or alien species, is high. Thus, juvenile salmon move through estuary as rapidly as possible, at considerable cost in energy and vulnerability to predation (and the federal and state pumping facilities in the South Delta).

Harvest. In the nineteenth century, commercial fisheries decimated CVS Chinook populations statewide. The fisheries were reduced initially because numbers of salmon had become too small to make canning profitable, and then regulations helped to reduce harvest rates. There was some recovery in populations until the completion of major Central Valley rim dams blocked access to most spawning and rearing habitat for CVS Chinook.

The impacts of commercial and sport fisheries have been through incidental take in ocean fisheries, which largely depend on hatchery fish from FRFH. It is likely that such take has been a significant source of mortality for the diminished populations of CVS Chinook, but its impact is not well understood due to lack of a robust genetic stock identification program or marking of all hatchery fish for a defined period, which would enable wild fish to be distinguished from hatchery fish at sea. Fisheries also select for younger, smaller, and less fecund fish as spawners, reducing resiliency of the populations over time.

Recent breakthroughs in sequencing specific segments of DNA to determine life history strategy and run assignment, called ancestry-informative markers (AIMs), shows promise for stock identification of Chinook caught at sea. Meek and colleagues (2016) developed testing to reliably identify Central Valley Chinook salmon by run timing with an average accuracy rate of 96%. This and other experiments will be very important for monitoring and managing the health of individual populations, especially the imperiled spring-run ESU (Meeks 2016).

CVS Chinook salmon have a relatively broad ocean distribution from Central California to Cape Falcon, Oregon, that is similar to Central Valley fall-run Chinook salmon. However, they are thought to incur less fishing mortality and pressure than their fall-run counterparts since they migrate to freshwater in spring before the most intense fishing pressure occurs in the summer (NMFS 2016). Salmon fishing is prohibited on Mill, Deer, and Butte creeks (NMFS 2016), though low recoveries of coded wire tags from Butte Creek fish are inconclusive as to harvest impacts on CVS Chinook.

Hatcheries. There appears to be little obvious hatchery influence on Mill Creek, Butte Creek, and Deer Creek populations, but Battle Creek (Rkm 534) and Feather River (Rkm 128) populations are strongly influenced by hatchery operations through straying and genetic introgression of fall-run and spring-run Chinook (NMFS 2016). Coleman National Fish Hatchery (Battle Creek) and Feather River Fish Hatchery are the two main hatcheries currently operating in the CVS Chinook ESU range. While Coleman National Fish Hatchery only raises and releases fall- and late-fall Chinook (USFWS 2017b), FRFH releases an estimated 2 million CVS Chinook smolts per year to prop up abundance in the Feather River. The millions of smolts are released in relatively close proximity to one another (Rkm 504-518), and have the potential to stray and spawn with fall- and late-fall Chinook in other watersheds. These releases significantly prop up the population; 78 percent of spawners in the 2010/2011 CVS Chinook salmon carcass survey on the Feather River were estimated to be from the FRFH (Kormos et al. 2012, Palmer-Zwahlen and Kormos 2013). This facility has received criticism for mixing spring- and fall-run fish for broodstock in the past (Williams 2006). While Butte Creek and Feather River CVS Chinook appear genetically distinct, about 200,000 juvenile Feather River CVS Chinook were planted into Butte Creek in 1986 in response to extremely low numbers of returning fish. However, there is little evidence that this plant had any effect on Butte Creek populations.

In addition, until 2015, at least half of the CVS Chinook salmon production from FRFH has been trucked to release sites such as the San Francisco Bay, rather than releasing juveniles into the river. This practice increases straying rates of returning adults to other watersheds, posing genetic risk to their populations (Kormos et al., 2012, Palmer-Zwahlen and Kormos 2013). A prolonged influx of FRFH CVS Chinook salmon strays to other Central Valley salmon populations, even at levels of less than one percent, can increase extinction risk after only four generations (Lindley et al. 2007). The extent of interbreeding between hatchery-reared fall and CVS Chinook salmon in the Feather River is unknown due to the absence of spatial or temporal barriers to spawning in the river today, but is estimated to be significant (NMFS 2016). Also, a significant portion of the FRFH returning adults (~20%) have been recovered in the American (Cramer 1996) and Yuba rivers in recent years (USACE 2012). On Battle Creek, perhaps as much as 29% of CVS Chinook salmon in 2010 were estimated to have originated from the FRFH (USFWS 2014). In addition, a recent study suggests a high likelihood that FRFH Chinook may be interbreeding with natural-origin spring- or fall-run Chinook salmon in the Sacramento River after a fish trap below Keswick Dam captured dozens of FRFH-tagged Chinook in recent years (Rueth 2015). If such introgression occurs with fish in Mill, Deer, or Butte creeks, as seems likely, the negative consequences may be important because these fish retain unique genetic signatures among extant CVS Chinook salmon (NMFS 2016). A Hatchery and Genetics Management Plan that mandates in-river release of juveniles for imprinting and reducing straying, which began implementation in 2015, should reduce genetic impacts of hatchery fish on natural-origin CVS Chinook. See Central Valley fall Chinook salmon account for more discussion of hatcheries.

Alien species. Predation on juvenile salmon by predators such as striped bass (*Morone saxatilis*), smallmouth bass (*Micropterus dolomieu*), and largemouth bass (*M. salmoides*) may be limiting, especially in areas with man-made structures that alter the environment such as near the pumping and diversion stations in the highly altered Delta. For example, the striped bass spawning migration in the Sacramento River corresponds with the outmigration timing of spring-run and fall-run Chinook juveniles to the ocean, which may be a source of juvenile salmon mortality (F. Cordoleani, NMFS, pers. comm. 2017). However, there is little solid evidence that predation is actually limiting salmon populations, especially compared to other factors.

In addition to anthropogenic threats discussed above, forest fires, volcanic activity, drought, and climate change all have a high potential to affect CVS Chinook because three major populations are located closely together in the Mount Lassen foothill region. Catastrophic forest fire has become a major problem in the Sierra Nevada stemming from a century of fire suppression, fuel accumulation, and housing development in the urban-forest interface. All three of the extant spring-run Chinook creeks have their headwaters in public and private forestland that has high potential for large, destructive fires. Lindley et al. (2007) examined fire risk and demographics in the spring-run watersheds and determined that a fire of 30km width could simultaneously burn the headwaters of all three populations, leading to heavy potential impacts on CVS Chinook. Such a fire has a 10% chance of occurring in any given year, increasing the vulnerability of these populations to such catastrophic stochastic events (Lindley et al. 2007).

Likewise, all CVS Chinook populations are vulnerable to volcanic eruptions from Mount Lassen, an active volcano located at the headwaters of Mill, Butte, and Deer Creeks. All three streams are located within the estimated reach of pyroclastic and debris flows from a potential volcanic eruption. The USGS has classified Mt. Lassen as "highly dangerous" (Lindley et al. 2007).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Central Valley spring-run Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Dams block access to over 90% of historical habitat and alter the quantity and quality of remaining habitat for CVS Chinook.
Agriculture	High	Diversions, mostly for agricultural uses, have reduced availability of cold, clean water throughout most of the ESU range.
Grazing	Low	Cattle grazing relatively common throughout the ESU, though impacts are likely small.
Rural /residential development	Low	Development at the forest-rural interface may reduce availability of cold water, especially during critical holding periods in summer and fall.
Urbanization	Low	Water diversion and groundwater pumping may negatively impact coldwater flows.
Instream mining	Low	Some legacy impacts from gravel mining possible.
Mining	Low	Legacy impacts are likely small; toxic spills from Iron Mountain Mine could be a major threat to all fish in the Sacramento River.
Transportation	Low	Watersheds have many logging or agricultural roads; impacts likely to be local and related to sedimentation.
Logging	Medium	Holding/spawning areas are within National Forests or private timberlands, which have been subjected to decades of logging and associated road building.
Fire	Medium	Fish in adjacent watersheds (Mill, Deer, Butte creeks) are susceptible to depletion from a single large fire.
Estuary alteration	High	The San Francisco Estuary is heavily altered, to the detriment of all migratory fish.
Recreation	Low	Boating, swimming, and fishing may disturb holding CVS Chinook, but magnitude of impacts unknown.
Harvest	Low	Recent estimates of commercial fishing pressure on CVS Chinook are low and do not contribute to overutilization; recreational fishing in freshwater is banned for CVS Chinook.
Hatcheries	High	Feather River Fish Hatchery fall- and spring-run Chinook are closely related and stray to other watersheds, reducing genetic integrity of natural-origin stocks.
Alien species	Medium	Predation from fishes such as striped and smallmouth bass may be locally significant.

Effects of Climate Change: CVS Chinook salmon are rated "critically vulnerable" to extinction by Moyle et al. (2012). This vulnerability is the result of the added effects of climate change on top of restricted distribution and depressed annual abundance. Climate change represents a special conservation challenge for these fish due to their reliance on cold spring water and snowmelt to sustain them through warm summer months. Lindley et al. (2007) indicate that climate change will likely lead to elimination of suitable thermal habitat in much of the currently accessible extant range by 2100. In a detailed field and modeling study of Butte Creek, Thompson et al. (2012) found that CVS Chinook salmon will be extirpated from the creek in 50-100 years or less, as the result of loss of coldwater habitat from climate change. Less snowfall, more variability in total precipitation, and potentially reduced overall precipitation, is expected across California in the coming decades (Lindley et al. 2007). Reduced snowpack and snow\water equivalents are expected to decrease over time, especially in Northern California, where snowpack is generally shallower than in the southern Sierra. Dettinger (2005) indicates that a 5°C rise in average temperatures is likely. Such higher temperatures, especially in lower elevation streams such as Butte, Cottonwood, and Big Chico creeks, will continue to stress CVS Chinook salmon. On top of greater variability in precipitation and warmer temperatures, more frequent and intense wildfires are expected statewide, which can burn riparian vegetation, lead to sedimentation and landslides, and reduce habitat for salmon. The chances of catastrophic fire under a warmer, dryer climate in California are very likely to increase over time (NMFS 2016).

This means CVS Chinook will need to spawn and hold higher in the watersheds than current infrastructure (dams and diversions, natural barriers) allows, making experimentation with trap-and-haul or other fish passage programs likely; this concept is currently being explored by the Yuba Salmon Partnership Initiative, a collection of agencies and other partner groups around Englebright Dam on the Yuba River (CDFW 2017b). Lusardi and Moyle (*In press*) however indicate that two-way trap and haul programs do not have a good record of success and that there are many uncertainties associated with the efficacy of such programs.

Restoration of former habitat is critical to maintaining long-term population stability, particularly in the face of future climate change. Enhancement of the Battle Creek population and restoration of the San Joaquin River population, in particular, are very important aspects of CVS Chinook conservation because both have good sources of cold water. In addition, the San Joaquin River is distant from other populations and could bolster spatial stability and increase genetic adaptation and diversity in putative San Joaquin tributary populations.

Habitat simplification and degradation have reduced resiliency of CVS Chinook populations and rendered them more susceptible to additional environmental stressors, such as drought (NMFS 2016). Extended dry or drought periods are predicted in the future in California under most climate change models, and could easily render most existing CVS Chinook habitat unusable, either through temperature increases or lack of adequate flows (Williams et al. 2016). Increased frequency and/or intensity of drought will lead to less water available behind dams, reducing cold water pool available for releases to support adult fish in warmer summer months. These reduced flows will also limit the ability of groundwater aquifers to recharge and to supply the cold spring flows that are characteristic of the Basalt and Porous Lava Domain. The impacts of recent drought years (2012-2016) and warm ocean conditions on the juvenile life stage will only be realized in potential low returns of CVS Chinook adults from 2015 through 2018 (Williams et al. 2016). This is already being realized with very low returns of CVS Chinook in 2015 to almost all watersheds in the ESU. Lethal water temperatures in traditional and non-traditional CVS Chinook salmon holding habitat during the summer in Butte and Big Chico creeks have been documented since 2013, causing elevated pre-spawn mortality in some populations. A large number of adults (about 900 and 230, respectively) were estimated to have died prior to spawning in the drought years of 2013 and 2014 (Garman 2015). Additionally, pre-spawn mortality in adult CVS Chinook salmon in Mill, Deer, Butte, and Battle creeks was observed in addition to longer duration of warm temperatures (Garman 2015). Thus, while NMFS (2016) considers CVS Chinook to be at moderate (Mill and Deer) or low (Butte Creek) risk of immediate extinction, these populations are likely to decline as impacts from the 2012-16 drought are realized in returning adults (NMFS 2016) and as climate change impacts become more severe (Thompson et al. 2012).

Status Score = 1.7 out of 5.0. Critical Concern. There is high likelihood of naturally spawning CVS Chinook going extinct in the next 50 years. Both state and federal governments list CVS Chinook as threatened. Our evaluation does not count fish from FRFH (and naturally spawning strays from the hatchery) that are currently included in the CVS Chinook ESU because of their hatchery dependence and hybridization with fall-run Chinook. However, the score would be only slightly higher if these fish were counted. Given the high likelihood of extirpation of the naturally spawning, independent populations, the spring-run life history strategy in the future may only exist in the Central Valley in rivers with cold water releases from dams, where natural reproduction is enhanced with fish from FRFH. We also recognize that some straying of CVS Chinook from FRFH into Deer, Mill, and Butte creeks probably occurs in most years, with unknown effects.

Recent management efforts and restoration projects have somewhat reduced their vulnerability to extinction, but the probability of populations plummeting in the future is high, especially considering recent downward trends (2014-2016). NMFS (2016) suggested CVS Chinook in Butte, Mill, and Deer Creeks were at low risk of extinction in the short term, but impacts of drought have caused populations to plummet. All three independent populations are in adjacent streams subject to natural and human-caused disasters, populations have been extremely small in the recent past, and all three streams are small and could become marginal for salmon as the climate continues to shift. The small populations in Battle and Clear creeks show increasing trends since 2011 and could potentially become sustainable populations, as could the experimental population in the San Joaquin River with active management and significant restoration, but drought impacts will likely reduce the number of returning adults in 2017-2018 (Williams et al. 2016, F. Cordoleani, NMFS, pers. comm. 2017).

While NMFS (2016) reports that the status of the ESU has probably improved since 2008 due to extensive restoration (e.g., Clear Creek) and increases in spatial structure of populations (Battle and Clear creeks), this evaluation does not take into account the more recent impacts of drought and climate change. A downward trend across Butte, Mill, and Deer creek populations has already been observed since 2014 (NMFS 2016). In general, low abundance, diversity, and poor spatial structure among independent populations of this ESU are troubling (NMFS 2016). The recent declines due to high pre-spawn and egg mortality, uncertain juvenile survival to smolting, and poor ocean conditions during the 2012 to 2016 drought are all causes for concern. In addition, straying of FRFH CVS Chinook salmon to other watersheds and introgression with other populations may also be working against the resiliency of remaining natural-origin populations (NMFS 2016). Thompson et al. (2012) therefore predict that the large population of unhybridized Chinook salmon in Butte Creek is likely to go extinct in the near future.

Table 2. Metrics for determining the status of CVS Chinook salmon, where 1 is poor value and 5 is excellent. Scores 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Self-sustaining populations are mainly in Deer, Mill, and Butte creeks, with potential for populations in Battle and Clear creeks, as the result of restoration.
Estimated adult abundance	2	Total numbers have periodically dropped below 5,000 fish even when FRH fish are counted. When FRH fish are not counted, most of the remaining fish are in Butte Creek.
Intervention dependence	2	Without significant restoration and intervention, CVS Chinook would likely already be extinct in the Central Valley.
Environmental tolerance	2	Narrow physiological tolerances in summer for both adults and juveniles considering streams they inhabit.
Genetic risk	2	Butte Creek and Deer-Mill Creeks populations appear to be distinct, but effect of hybridization with FRH fish unknown.
Climate change	1	Extremely vulnerable given small population sizes and range, as well as already high temperatures of streams.
Anthropogenic effects	1	Four High threats.
Average	1.7	12/7.
Certainty (1-4)	4	Well-studied.

Management Recommendations: CVS Chinook declined from once being as abundant as fall-run Chinook salmon to a few hundred to perhaps a few thousand fish in each population (NMFS 2014). The most recent annual population trends indicate pending extinction and that past management efforts have not been sufficient to protect them from extinction. Climate change models show an increased need for large-scale management, with predicted increases in average air temperatures and more variable precipitation patterns expected by 2100. Lindley et al. (2007) found that at a minimum, California water temperatures are expected to increase 2°C by 2100, but perhaps as high as 8°C, which would remove all currently available thermal refuges for CVS Chinook in California. The largest independent run, in Butte Creek, already faces near-lethal temperatures in summer and has a high probability of being extirpated in the not-too-distant future from climate change effects alone (Thompson et al. 2012). These factors all need to be taken into account when managing CVS Chinook salmon.

Our management recommendations fall into three distinct categories: self-sustaining populations, hatchery-supported populations, and restored populations. Most of this account focuses on self-sustaining populations because the population supported by the Feather River Fish Hatchery is a hybrid stock genetically close to CV fall-run Chinook. However, a substantial part of this population does maintain the spring-run life history and so is worth continuing to manage for that life history, especially if the hatchery contribution can be effectively managed to reduce influence on selection, survival and behavior. Restored populations are works in progress, with recent work on the San Joaquin River, Clear and Battle creeks, the Yuba River, and the McCloud River.

Self-sustaining populations. The most important actions for preventing extinction of CVS Chinook salmon lie in protecting/enhancing coldwater habitat in Butte creek and restoring unimpaired flows in the lower reaches due to diversions in Deer and Mill creeks. The fact that self-sustaining populations of CVS Chinook salmon have managed to persist in Mill and Deer creek is mostly a matter of luck: the streams were too small for economically feasible dams yet supplied cool water year round due to mainly to their geology and geography. The present limited current distribution of the 'independent' CVS Chinook makes them vulnerable to localized stochastic events (fire, volcanic eruption, etc.) in which the entire run may be jeopardized by a single incident. Fortunately, all three watersheds are the focus both of large-scale restoration projects (e.g. Deer Creek Meadows) and citizen groups that want to protect the fish and their streams.

Seizing an opportunity to protect an important source of diversity in Chinook life histories in the Central Valley, agencies and landowners in the Central Valley, in coordination with the Deer Creek and Mill Creek conservancies and Friends of Butte Creek, began protecting CVS Chinook salmon. For Deer and Mill Creeks, cooperative agreements among ranchers, lumber companies, agencies, and other partners to do business in a fish-friendly manner, while public agencies similarly managed their own lands and waters. The most important immediate actions involve reducing diversions and modernizing fish passage facilities on Mill, Deer and Antelope creeks to increase flows for adult and juvenile fish passage through water acquisition, conjunctive use of wells, and water use efficiency plans and improvements (NMFS 2016). In 2014 and 2015, NMFS and CDFW developed a Voluntary Drought Initiative program (NMFS 2017) to provide minimum flow protection in lower Deer and Mill Creeks. Where voluntary cooperation by diverters was not obtained, the California State Water Resources Control Board (SWRCB) provided a "back-stop" for flow protection by the adoption of emergency regulations (M. Johnson, CDFW, pers. comm. 2017). Similar Voluntary Drought Initiative cooperation or SWRCB flow regulation should be permanently secured for Mill, Deer, and Butte Creeks. Johnson (2016) lists other recommendations to benefit spring-run in these watersheds.

On Butte Creek, Pacific Gas and Electric (PG&E) has indicated (February 2017) that they will no longer operate their hydroelectric project or be responsible for implementing flows beneficial to CVS Chinook starting in about a decade. What impact this forfeiture of the operating license for the hydroelectric project and water operations will have on the state of Butte Creek and plight of CVS Chinook there remains to be seen. This may provide an opportunity for developing new approaches to protecting Butte Creek fish from predicted extirpation from climate change effects (Thompson et al. 2012).

Hatchery-supported populations. FRFH salmon express the spring-run life history, but are of a mixed genetic background. Many of these fish wind up spawning in the Feather River below the hatchery and contributing to returning salmon numbers, although there is only one population. In the hatchery itself, a genetic management program should be aimed at reducing domestication trends and to enhancing the spring-run life history, while hatchery operations should be focused on minimizing the potential for fall-run and spring-run fish to interbreed.

Restoration: Battle Creek. Restoration in many watersheds, such as Battle Creek, have begun to improve the spatial structure of the ESU overall and provide some hope of recovery (NMFS 2016). We therefore encourage the full implementation of a project on Battle Creek to turn it back into a major CVS Chinook stream. In 1999, PG&E, National Marine Fisheries Service, U.S. Fish and Wildlife Service, U.S. Bureau of Reclamation, and California Department of Fish and Wildlife reached an agreement to restore salmon and steelhead in Battle Creek. Under this arrangement, nearly 80km of habitat will be opened up for salmon spawning and rearing. The Battle Creek Salmon and Steelhead Restoration Project will remove dams, install fish screens, and end the diversion of water from the North Fork to the South Fork specifically to benefit listed salmon runs (NMFS 2016), with full implementation expected by 2020.

Restoration: San Joaquin River. Restoration of the San Joaquin River to support CVS Chinook salmon is a heroic effort that should continue, given that the river was illegally dried up when it still supported a substantial run of CVS Chinook salmon. In 1948, virtually all water behind Friant Dam on the San Joaquin River was sent down the Friant-Kern and Madera canals to support agricultural uses, with a small release for riparian landowners immediately below the dam. CDFG officials attempted to rescue the 1948 run by trucking some 1,915 CVS Chinook around the dry stretch to the tailwaters at the base of Friant Dam. There the fish successfully over-summered and spawned, but the outmigrating smolts were stranded in the dry river downstream; in just a few years thereafter, the run was extirpated, as was the companion run in the Kings River (Moyle 2002).

In 2006, a settlement agreement was reached for the San Joaquin River, known as the San Joaquin River Restoration Program, to provide minimum instream flows to create a permanent flow of water all year round, plus additional water for migration, spawning, and rearing of an experimental population of CVS Chinook salmon. Among the project goals are extensive habitat restoration to riparian and instream habitats, necessary after 50+ years of complete neglect and abuse of the channel. Restoring continuous flows to the approximately 240km of often dry and heavily altered river channel will take place in a series of phases.

Toward that end, interim and experimental flows started in 2009 with the goal of establishing a self-sustaining population of CVS Chinook by 2025. Restoration flows designed to restore connectivity from Friant Dam to the Merced River began in January 2014; the first release of CVS Chinook salmon from the Feather River Fish Hatchery into the San Joaquin River occurred shortly thereafter in April, 2014. A second release occurred in 2015, and future releases are planned to continue annually in the spring. A conservation hatchery and captive broodstock program was initiated in 2012 to support the reintroduction with limited impact on source populations by utilizing excess broodstock from FRFH. The 2016 release included the first generation of CVS Chinook salmon reared entirely in the San Joaquin River in over 60 years (NMFS 2016). Such watershed-scale reconciliation and restoration actions are essential for keeping CVS Chinook from going extinct in the next 50 years. Barrier removal or some kind of trap and haul operation will also likely be a major part of CVS Chinook conservation in the near future, until more permanent solutions to restore passage to historical habitat can be found.

General recommendations. Dam and water conveyance operations throughout the salmonid migration corridor from rim dams to the valley floor and the ocean must be re-thought and adjusted to help recover CVS Chinook and avoid local extirpations in the short-term. Instream flows should be restored to mimic natural hydrographs to the extent practicable on the Sacramento River and its tributaries. Additional cold water should be secured through shifts in land use practices and management, in cooperation with private and public entities, to help bolster populations in places such as Antelope, Clear, Deer, and Mill creeks (NMFS 2016). Oroville Dam and related operations must be evaluated in the context of long-term salmonid persistence in the Central Valley. Fish passage improvements in all CVS Chinook watersheds should also be prioritized to foster increases in spawning success and abundance.

In February 2017, an unprecedented rainfall event caused catastrophic failure of the spillway at Oroville Dam, just upstream from the FRFH. About 200,000 people were evacuated from downstream cities including Oroville, Yuba City, and Marysville. Meanwhile, 6.5 million fall- and spring-run Chinook at FRFH were put into trucks and transported to the hatchery annex near Thermalito Afterbay, several kilometers downstream (CDFW 2014b). The mobilization of large amounts of water, sediment, and large concrete structures associated with the ruptured spillway likely removed spawning and rearing habitat immediately downstream of the dam, but the effects of this event on CVS Chinook in the Feather River will not be known for years.

Third, floodplain management must be altered to restore ecologically based flows in the Sacramento River to allow breaching and volitional immigration and emigration of juvenile salmonids to floodplains such as Sutter and Yolo bypasses to help increase survival of smolts. At the same time, NMFS (2014) noted the need to explore alternative water operations and conveyance systems in the Delta to improve prospects for all Central Valley salmonids, including allowing more water to flow downstream to the Delta to enhance water quality and provide more suitable migration and rearing habitat that favors native fishes. Such a task will require an enormous investment of resources and novel ways of thinking to restore more natural hydrology and ecosystem function of the Delta. Such improvements will be essential to leveling the playing field for native fishes in the heavily altered Delta and San Francisco Estuary. Ongoing tidal marsh restoration throughout Suisun, San Pablo, and Grizzly bays could be most effective in conjunction with changes to water operations and management of the Delta. Multi-benefit projects that leverage incentives for stakeholder participation should be expanded where possible to help build momentum and scale up successful projects to region-wide initiatives. Many of these recommendations come directly out of the NMFS (2009) Biological Opinion on the continued operations of the state and federal pumping facilities in the Delta.

In addition to the specific restoration and reconciliation work occurring across the ESU, there is an important need for continued monitoring of watershed conditions and populations to gauge progress toward recovery of CVS Chinook. Drought impacts are likely to drive remaining populations again into the high extinction risk categories between 2015-2018, reinforcing the need to document changes that could impact viability and persistence of populations.

In addition to the actions discussed above, the NMFS (2014) Recovery Plan for Central Valley winter and CVS Chinook and steelhead identifies more specific actions that can be taken in the short term to benefit CVS Chinook:

1) Augmenting spawning gravel on Clear Creek
2) Enhancing riparian habitat and spawning gravel on the Yuba River
3) Restoring access to high elevation habitat in the Yuba River upstream of New Bullards Bar Dam and in the Sacramento River upstream of Shasta Dam
4) Reducing harvest of CVS Chinook in ocean salmon fisheries
5) Implementing temperature reduction at the DeSabla Forebay (Butte Creek)
6) Modernizing fish passage facilities at Weir 1 in the Sutter Bypass;
7) Finalizing and implementing the Hatchery and Genetics Management Plan for the FRFH to reduce interaction between hatchery and wild fish
8) Providing passage at Sunset Pumps weir (Feather River)
9) Implement fish passage improvements at Stanford Vina Irrigation Company and Deer Creek Irrigation District dams on Deer Creek and Upper Dam on Mill Creek
10) Increase instream flows between March 1 and July 15 to protect migrating spring-run Chinook salmon.

SACRAMENTO RIVER WINTER-RUN CHINOOK SALMON
Oncorhynchus Tshawytscha

Critical Concern. Status Score = 1.3 out of 5.0. Sacramento River winter-run Chinook salmon (winter-run) face immediate risk of extinction. The ESU is extirpated from its native spawning range and has been reduced to a single small spawning population, which is wholly dependent on artificially-created spawning habitat and cold water releases from Shasta Dam.

Description: There are few obvious morphological differences separating the four runs of Central Valley Chinook salmon, though winter-run tend to be smaller than fall Chinook. For a full description of Chinook salmon, see the North Central Coast Chinook salmon account.

Taxonomic Relationships: While Sacramento River winter-run Chinook salmon are genetically distinct from other Central Valley runs, the four runs of Central Valley Chinook salmon are more closely related to each other than they are to fish from outside the Central Valley. Historically, there were four separate populations of winter-run Chinook that spawned in headwater reaches of the upper Sacramento, McCloud, and Pit rivers and Battle Creek (Tehama Co.). For a more complete discussion of taxonomic relationships among Central Valley Chinook salmon, see the Central Valley spring Chinook salmon account.

A major problem, however, has been identification of members of each run when they are juveniles, especially when mixed in downstream reaches. The standard method for separating runs has been a simple length-at-date measurement, based on the assumption that fish hatching at different times would retain the difference in size as they grew larger and moved downstream. Harvey et al. (2014) found that, in fact, many genetically identified fall, late-fall, and spring-run were being identified as winter-run because they were bigger than expected under length-at-date criteria. Part of the reason for these inaccuracies is likely due to the fact the growth rates differ according to the habitats in which juveniles rear. However, the growth rates, which underlie the length-at-date criterion, were based on solely on models for in-channel rearing, therefore when individuals gain access to more productive off-channel or floodplain habitats growth rates can be accelerated leading to possible misidentification. "The two central assumptions of the length-at-date approach (i.e. segregated [length] ranges and a constant shared growth rate among races) were not supported by the [length] data for genetically identified juveniles (Harvey et al. 2014, p. 1182)." Merz et al. (2014), found that using morphometric measurements decreased the rate at which genetically identified fall-run and late-fall-run Chinook juveniles were counted as winter-run or spring-run.

Life History: For the basic life history of Chinook salmon, see the North Central Coast Chinook account. Winter-run Chinook, which are known only from the Sacramento Valley, have a life history strategy that differs from all other Chinook salmon. Winter-run life histories were shaped in response to access to year-round, spring-fed, cold-water stream reaches, a rare hydrologic feature among salmon bearing streams, of the headwaters of the Sacramento River watershed. Eventually, breeding in isolation from other runs led to the evolution of distinctive strategies affecting all winter-run life stages including adult migration timing, spawn timing, egg incubation duration and juvenile outmigration.

Winter-run enter fresh water as sexually immature adults in January through May with runs peaking in mid-March. Prior to dam construction, winter-run migrated to the headwaters of the Sacramento River Watershed, where they exploited spring-fed stream reaches in the basalt and porous lava region of the northeastern part of the state. They would hold in these perennial, cold water reaches for several months until spawning in April through early August (Williams 2006).

The spring through mid-summer spawn and egg incubation period (the most temperature-sensitive of Chinook salmon life stages) typically occurs during the hottest part of the year when water temperatures in many California rivers exceed the lethal range for Chinook embryos. The consistently cold temperatures of winter-run streams presumably lengthened incubation duration of winter-run eggs relative to eggs of other runs incubating in warmer water reaches.

In the post-dam era in the reach below Keswick Dam, fry emerge from the gravel in July through mid-October (Yoshiyama et al. 1998, Williams 2006) and juveniles rear for approximately 5-10 months before moving downstream (Yoshiyama et al. 1998). Thus, spawning in summer gives winter-run an advantage over the spring- and late-fall runs by providing longer rearing times in the stream, without juveniles having to over-summer in the following year (Stillwater 2006).

Peak movement for juveniles of all runs of Chinook salmon in the Central Valley tends to be at night, thus reducing predation risk. Juvenile entry into the Sacramento-San Joaquin Delta occurs from January to April (Stillwater 2006). According to Williams (2006), most fry migrate past Red Bluff diversion dam in summer or early fall, but many apparently rear in the river below Red Bluff for several months before moving to the Delta in early winter.

Habitat Requirements: For general Chinook salmon habitat requirements, see the North Coast Chinook salmon account.

Adult migration and holding. Winter-run historically migrated high into the spring-fed reaches of the McCloud, Pit, and Sacramento rivers and Battle Ck to spawn, which required migration during winter-run and early spring when flows were high enough enabling passage. Once they reached their spawning grounds, they held for several months in deep pools with good cover until they were ready to spawn. Today, winter-run Chinook still attempt to migrate to the highest upstream spawning habitats available to them but are stopped by Keswick Dam (Stillwater 2006). Optimal temperatures for holding range from 10-16°C (see thermal tolerance table in North Coast Chinook account); optimal holding velocities range from 0.47 to 1.25 m/s, significantly higher than selected by the other runs (Table 1, USFWS 2003).

Spawning and egg incubation. Winter-run Chinook require water temperatures that must be cold enough during summer to enable successful embryo incubation, but warm enough in winter-run to support juvenile rearing (Stillwater 2006). Little is known of winter-run spawning habitats prior to dam construction. From observations below Keswick Dam, they appear to spawn in deeper water than the other runs of Central Valley Chinook, generally from 1-5 m (USFWS 2003), but have been observed spawning in water as deep as 7 m (Moyle 2002). Stream temperatures that exceed the 13.3°C (56°F) daily average temperature limit winter-run egg survival (NMFS 2016a):

Juvenile rearing and outmigration. Winter-run juveniles appear to occupy fresh water nearly all year round. In 1898 Rutter (1903) observed extensive numbers of juvenile salmon of multiple size classes over-summering in pools of the upper Sacramento River near Simms. That same year in Battle Creek Rutter captured hundreds of small salmon (between 35-50 mm) in October and November suggesting a midsummer emergence from the gravel that would match up with the winter-run life history strategy. Today winter-run run are first detected in July close to their natal grounds in the Sacramento River near Red Bluff, and smolts are last detected at Chipps Island leaving the Delta as late as May (del Rosario et al. 2013). In a seine survey in July 1898, Rutter found no salmon juveniles rearing in the river downstream of the mouth of Battle Creek (Rutter 1902). In the reaches below Keswick Dam, juveniles emerge from the gravel in mid-summer and are restricted to reaches that maintain cool summer temperatures (generally upstream of the mouth of Deer Creek at Rkm 354) between July and September for rearing (Stillwater 2006). Once water temperature cools in the downstream reaches in the early fall, the rapidly growing parr use more of the river for rearing.

Because of their distinctive emergence time, winter-run Chinook fry generally have little competition from other juvenile salmonids during the first few months of life, although the extent to which juveniles of other runs held over the summer in the McCloud River and other historical habitats is unclear. As winter-run Chinook move downstream, they share rearing habitat with spring-run Chinook juveniles entering the Sacramento River from the Mill, Deer, and Butte Creek drainages, which may be as much as a year old and are thus considerably larger than winter-run Chinook juveniles (Williams 2006, Stillwater 2006). While this may result in a competitive advantage for spring-run, there is some indication that the two runs use habitat differently based on their sizes and thus do not directly compete (Stillwater 2006). They also mix with juveniles from other runs during their outmigration, making identification difficult (Merz et al. 2014); thus any general description of the migratory behavior of juvenile winter-run Chinook, such as the one that follows, should be viewed with caution.

Outmigration appears to be tightly correlated with the first high flows of the migration season. Winter-run tend to migrate past Knights Landing on the Sacramento River (Rkm 144) *en masse* shortly after flows reach a threshold of 400 cubic meters per second measured at the Wilkins Slough gage (del Rosario et al. 2013). These "first flush" events may occur anytime between October and April, depending on the water year. However, in most years outmigration is concentrated in November and December with the majority of the population passing Knights Landing by early January (del Rosario et al. 2013).

Residence times in the lower river and estuary (between Knights Landing and Chipps Island) average about 3 months (ranging from 41 to 117 days) with juveniles that arrive earlier in the migration season apparently residing for longer periods in the lower river and Delta. Winter-run-sized fish have been detected at Chipps Island as early as December and as late as May, with peak outmigration from the Delta in mid-March (del Rosario et al. 2013).

Winter-run juveniles would have historically benefitted from the typical winter-run flooding which would have made hundreds of thousands of acres of floodplain and off-channel habits available for rearing in the Sacramento River basin. Inundation of floodplains increases residence times, allowing water to warm compared to the relatively cool river channel; increases in residence time and water temperature facilitate greater decomposition of terrestrial vegetation and increase primary production in the form of algal phytoplankton (Ahearn et al. 2006, Grosholz and Gallo 2006). Detrital decomposition and algal primary production are the primary sources of carbon that fuels aquatic food webs and supports zooplankton and other invertebrate populations, which are the primary source of food for juvenile fish. Fish food densities are typically far greater on the floodplain than in the river (Conrad et al. 2016, Katz et al. *in press*). In addition, juvenile salmonids may use less energy to maintain themselves on floodplains than they would in the mainstem Sacramento River, further increasing growth rates. For these reasons, growth rates for juvenile Chinook rearing in floodplain habitats tend to far exceed those rearing in riverine habitats (Sommer et al 2001, Jeffres et al. 2008, Katz et al. *in press*). Rapid growth results in larger out-migrants and higher survival rates in the ocean (Unwin 1997, McCormick et al. 1998, Hayes et al. 2008, Williams et al. 2016). However, there are very few floodplains now available to salmonid juveniles in the Sacramento River Watershed, which may have a profound negative impact on winter-run survival and recruitment, in addition to the loss of spawning habitat upstream of Shasta Dam.

Distribution: All four winter-run Chinook populations have been extirpated from their historical spawning areas in the Upper Sacramento, Pit, and McCloud Rivers and Battle Creek (Lindley et al. 2007). The closing of Shasta Dam in 1945 halted migration into the Upper Sacramento, Pit and McCloud River drainages. The Battle Creek population was extirpated by hydropower dam operations. Additionally, the weir at Coleman National Fish hatchery was a barrier to upstream migration until recently (NMFS 1997, Lindley et al. 2007). Battle Creek remains unsuitable for holding and spawning due to high summer water temperatures, particularly during dry years (Lindley et al. 2007, NMFS 2016a).

Currently, there is only a single winter-run Chinook population, which spawns in the Sacramento River below Keswick Dam near Redding (NMFS 1997). This population holds and spawns at the base of Keswick Dam, where cold-water releases from Shasta Reservoir, combined with artificial gravel additions, have created headwater-type habitat on the valley floor (NMFS 2016a).

Trends in Abundance: Historical abundance of winter-run Chinook is thought to have been approximately 200,000 spawners per year (NOAA 2005). Winter-run have been in decline for the last decade with the 2011 escapement of 827 spawners being the lowest run since the construction of the LSNFH conservation hatchery in 1997 (NMFS 2016b, Figure 1). In 2015, an estimated 3,500 winter-run Chinook salmon returned to the Sacramento River to spawn (NMFS 2016b, Figure 2). Drought conditions and warm water releases from Shasta Reservoir contributed to extremely low fry-to-egg survival in the 2014 and 2015 brood years. Low returns are expected from these brood years, with the additional concern that the proportion of hatchery spawners will also likely increase. Recent abnormally warm conditions in the ocean may also negatively impact winter-run returns for the next 2 to 3 years (NMFS 2016a, Williams et al. 2016).

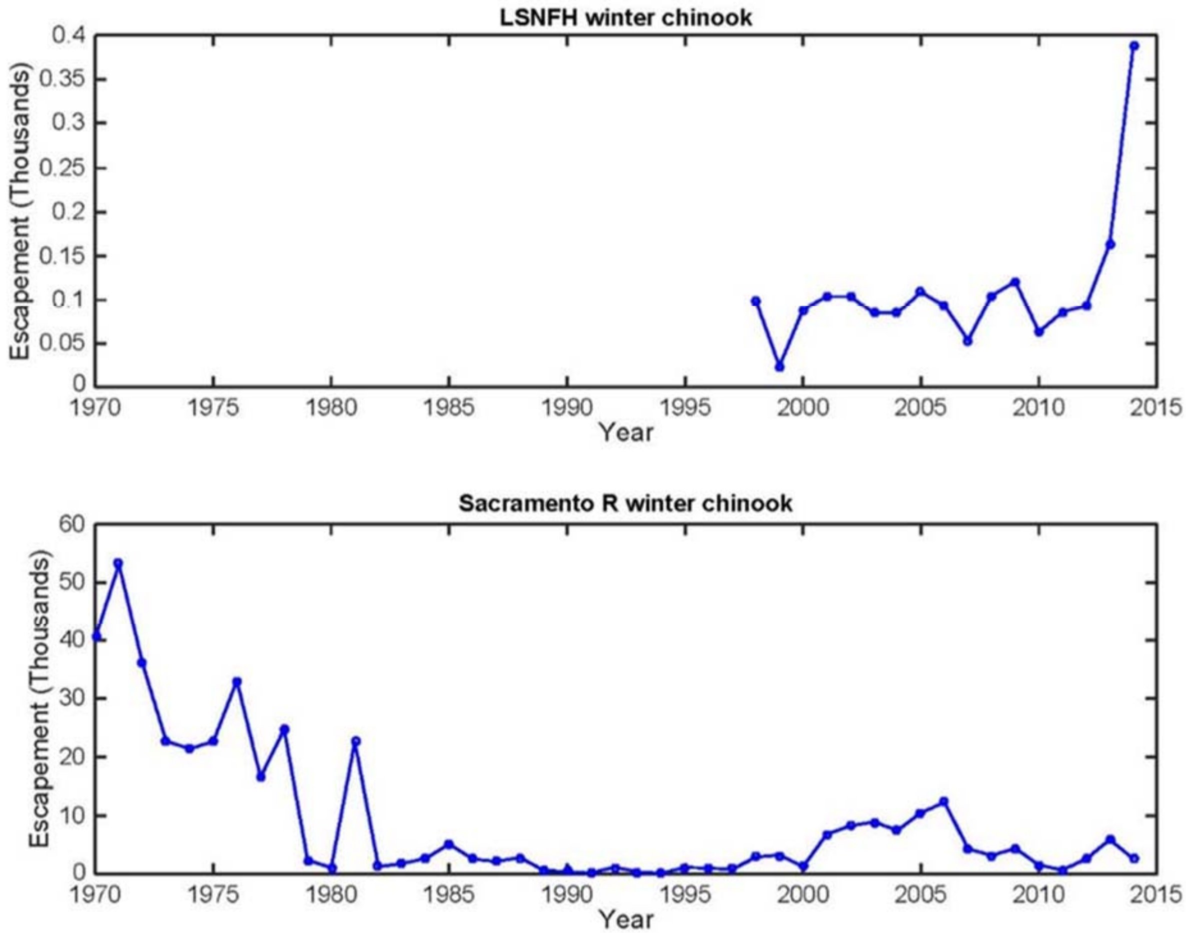


Figure 1. Time series of escapement for SR winter-run Chinook salmon populations (a) used as broodstock at LSNFH and (b) spawning in-river. Estimates for in-river spawners is the average number of adults counted at Red Bluff Diversion Dam and the carcass survey mark-recapture estimates (when available). Note: only mark-recapture estimates used beginning in 2009; From NMFS 2016b, based on Azat 2014.

Accurate abundance data has been difficult to collect, and there have been numerous instances (e.g. Williams et al. 2006) in which putative winter-run were discovered to be either spring or late fall-run fish. Livingston Stone Hatchery produces approximately 200,000 winter-run smolts per year that are marked and tagged before release (Williams 2006), and percentage of hatchery fish spawning below Keswick Dam in recent years has increased to an overall trend

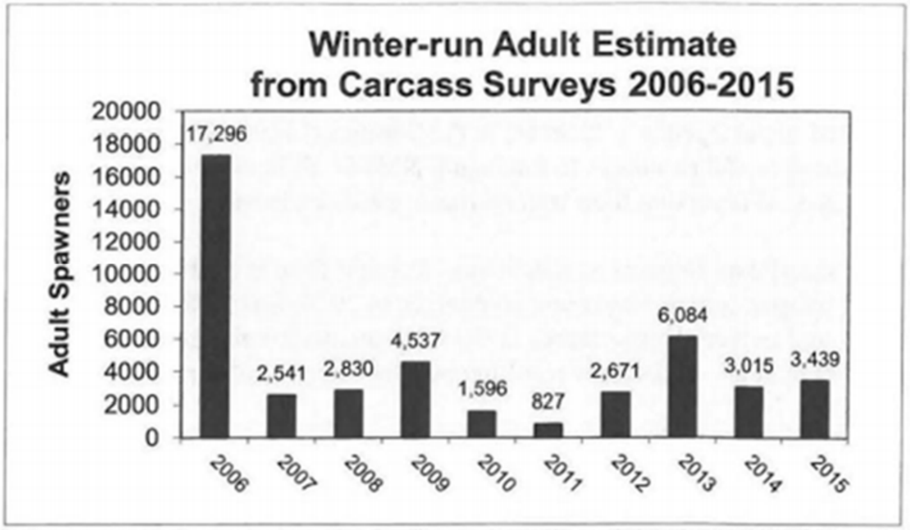


Figure 2. Winter-run Chinook salmon spawner escapement in the Sacramento River, 2006-2015. Data from CDFW GrandTab, 2015. From NMFS 2016b, Fig. 1. pg. 2.

Factors Affecting Status: Major factors affecting or potentially affecting the status of Sacramento winter-run Chinook salmon in the Central Valley are discussed below.

The biggest single cause of decline of winter-run Chinook salmon was the construction of Shasta and Keswick Dams in the 1940s, which blocked access to historical spawning habitat. The subsequent steep decline of winter-run abundance in the late 1980s-early 1990s was precipitated by a combination of 1) excessively warm water released from Shasta Dam, 2) barriers to passage of both juveniles and adults, 3) entrainment in diversions, 4) possibly heavy metal contamination and acid mine drainage from Iron Mountain Mine (NMFS 1997), 5) loss of floodplain rearing habitat, and 6) commercial and recreational fisheries, which do not discriminate between hatchery fall-run Chinook and wild fish of any run (NMFS 2016a). NMFS (1997) has also expressed concern over climatic factors further affecting habitat-based issues, primarily through extended droughts, low flows, and higher temperatures. The effects of poor conditions for growth and survival in fresh water are also exacerbated when poor conditions naturally occur in salt water (Satherthwaite et al. 2014). Unfavorable ocean conditions from periodic El Nino events in the Pacific Ocean can reduce salmon survival by altering upwelling and decreasing productivity, thus reducing food availability (NMFS 1997).

Dams. Shasta and Keswick Dams effectively prevented all upstream spawning migration for winter-run Chinook, denying access to all historical spawning and most rearing areas (NMFS 1997, Williams 2006). Ironically, the cold water releases from the dam also kept the run from going extinct. It was not expected that winter-run would survive after Shasta Dam was built (Moffet 1949), but the cold water releases allowed spawning to occur in a previously unsuitable river reach directly below Keswick Dam (NMFS 1997). Keswick Dam, located 14 km below Shasta Dam, regulates the releases from Shasta Reservoir, as well as flows diverted from the Trinity River. Initially, water temperatures were cold enough below Keswick Dam for annual spawning of winter-run Chinook. However, drought years and high levels of water removal rendered water temperatures below Keswick Dam unsuitable (up to 27°C) with enough frequency that the population all but disappeared in the late 1980s and early 1990s (NMFS 1997, NOAA 2005). The high temperature water released from Shasta Dam was credited by NMFS as one of the main factors that led to their listing as endangered under the federal Endangered Species Act (NMFS 1997). A temperature control device (TCD) was installed on Shasta Dam in 1997 to provide a continuous supply of cold water, as well as to improve dissolved oxygen and turbidity levels. The TCD was built for winter-run Chinook in particular, but has also benefited other runs.

During the recent drought, management of the cold water releases from Shasta Dam was among the most controversial of all water issues in the state. Flows released early in the season (May-June), encouraged adults to spawn over a wide area below the dam. However, flows then abruptly decreased in July and August because of the depleted cold-water pool in the reservoir, resulting in extremely high mortality rates of developing embryos, presumably from a combination of warmer temperatures and reduced hyporheic flow, reducing oxygen delivery to embryos (Martin et al. 2017).

An additional impact of Shasta and Keswick Dams has been coarsening of the substrate in spawning areas from large releases from the dam. Such releases move spawning gravel downstream, while preventing new gravel inputs from upstream (Stillwater 2006). This has led to a decrease in available spawning habitat over time and requires continuous gravel augmentation in the reaches below the dams to provide spawning habitat.

Red Bluff Diversion Dam (RBDD) is widely credited with causing 30 years of significant passage impairment to both upstream migrating adults and out-migrating juveniles due to inadequate fish passage (i.e., poorly designed fish ladders). In addition, predatory fish gathered at the base of the dam and historically devoured many out-migrating juveniles with the assistance of the RBDD's lighting system, which made the juveniles visible at night. This has since been changed. Initially a NMFS Biological Opinion required that the dam gates be raised for six to nine months of the year; since 2012, the gates at RBDD have been open year-round to allow unimpeded upstream and downstream fish passage and minimize the impacts of predation at the dam.

Agriculture. Agricultural diversions along the Sacramento River presumably have some impact on out-migrating juvenile winter-run salmon (Moyle and Israel 2006). There has been a concerted effort at improving diversions through screening and most large diversions, such as the Anderson-Cottonwood Irrigation District, Glenn Colusa Irrigation District, Reclamation District 108, and Reclamation District 1004 are now screened (CDFG 2004).

A more important driver of both direct and indirect mortality are the Central Valley Project and State Water Project pumps in the southern Delta. Kimmerer (2008) estimated that the loss rate as the result of project pumps was "on the order of 10% or less" (p. 24), a rate which varies according to numbers of fish entrained as well as pre- and post-entrainment mortality (which are poorly understood). The tendency to increase pumping in the winter-run in order to reduce pumping at other times of year (for protection of Delta smelt and other species) may further increase entrainment mortality rates for winter-run Chinook salmon (Kimmerer 2008).

NMFS partners with the U.S. Fish & Wildlife Service, California Department of Fish & Wildlife, the State Water Resources Control Board the Department of Water Resources, and others to ensure that water operations do not put the continued existence of winter-run Chinook in jeopardy. They do so through extensive monitoring near the massive Delta pumps, and rely on coded-wire tags and genetic samples from winter-run Chinook daily salvage and loss counts. The baseline for these assessments is an estimate of survival to the Delta, which was estimated at 42% for 2012-2015 (NMFS 2016a). Since the 2009 Biological Opinion on the operation of the pumping stations in the Delta was released, NMFS authorizes incidental 'take,' as defined in the

Endangered Species Act, of up to 1% of the total juvenile production estimate (JPE) for winter-run Chinook salmon per water year before adjustments to the State Water Project and Central Valley Project operations must be made (NMFS 2016a). For water year 2015, these mortality allowances were about 1,000 wild and 1,500 hatchery fish (NMFS 2016a). There is some flexibility, as water operations managers can choose another operational trigger if 2.5 winter-run juveniles become entrained per thousand acre-foot of water released in the Delta (NMFS 2016a). How well these measures protect winter-run Chinook salmon population is not known.

Urbanization. Current winter-run spawning and holding habitat is proximate to heavily developed urban areas near the city of Redding. Streams in urban settings often face elevated risk of pollution from surface runoff, sewage discharges and storm drains, which can degrade water quality during vulnerable life history stages such as adult holding and egg incubation. The effects of the City of Redding on winter-run Chinook salmon are largely unknown.

Mining. Iron Mountain Mine is a superfund site which drains highly contaminated heavy metal contaminate effluent into Keswick Reservoir and has severely impacted water quality in the Sacramento River in the past by discharging toxic metals and acid mine drainage. It is an EPA Superfund Site and millions of dollars have been spent on remediation and clean up. A dam on Slickrock Creek has reduced 95% of the release of toxic metals, resulting in low levels of dissolved heavy metals in Sacramento River water. Despite ongoing funding, the mitigation solutions must be regarded as temporary, given the potential for dam failure and other factors causing massive pollution of the river. A particularly severe problem would be failure of the dirt dam holding back toxic waste from Iron Mountain Mine, which could wipe out fish and aquatic life in an extended reach of river below Keswick Reservoir (NMFS 2016a).

Transportation. The Sacramento River flows through urban Redding. Through this stretch, the river is crossed by numerous auto and train bridges, which also increase the possibility of toxic spills into the river. A disaster similar to the 1991 spill of pesticides at the Cantara Loop Bridge on the upper Sacramento River could impact an entire year class of winter-run Chinook and other aquatic life.

Estuarine alteration. There is growing appreciation of the importance of "biocomplexity" for the persistence of salmon in a variable environment (Hilborn et al. 2003, Carlson and Satterthwaite 2011). Biocomplexity (AKA the Portfolio Effect) is defined as multiple variations in life history that improve the ability of populations to persist in changing environmental conditions. Historically, juvenile fall-run Chinook salmon probably entered the estuary in different months and spent varying amounts of time there. Loss of habitat diversity in the San Francisco Estuary has limited life history diversity and the best strategy for juvenile salmon today seems to be to move through the estuary as quickly as possible. Large pumping stations in the south Delta divert approximately 40% of historical Delta flows, resulting in substantial modifications in flow direction (Nichols et al. 1986). This pumping also increases likelihood of out-migrating smolts entering the interior Delta where longer routes, impaired water quality, higher predation and entrainment lead to higher mortality rates (Perry et al. 2010).

Despite long-term monitoring, causes of apparent high mortality rates as fish pass through the estuary are poorly understood. General observations suggest that rearing conditions in the estuary are often poor; highest survival occurs during wet years, when passage through the estuary is likely most rapid (Brandes and McLain 2001, Baker and Morhardt 2001). Flooding in wet years also increases rearing habitat in the Delta and Yolo Bypass, which may also have a positive effect on growth and survival. To improve survival, most hatchery juveniles are transported and released downstream of the Delta in San Pablo and Grizzly bays (CDFW 2014).

Transporting smolts improves survival, but it also increases rates of straying upon return as adults. High straying rates contribute to homogenization of population structure and reductions in fitness by facilitating gene flow between populations in different streams, thus reducing biocomplexity within the CV Chinook salmon complex.

Recreation. The upper Sacramento River through the city of Redding supports substantial angling pressure for rainbow trout. Angler pressure tends to be greatest in locations and at times where winter-run Chinook are actively spawning and therefore more susceptible to impacts. Some are inadvertently hooked in this section of river must be released without being removed from the water, however, the process of hook-and-release likely still negatively impacts adults. An additional concern is that wading anglers could inadvertently trample redds while pursuing rainbow trout. To reduce these impacts, the Sacramento River from Keswick Dam downstream to the Highway 44 Bridge in Redding was closed by emergency regulations adopted by the California Fish and Game Commission in 2016 (CDFW 2016).

Harvest. Myers et al. (1998) examined harvest impacts and found that freshwater harvest was negligible, but that ocean harvest had considerable impacts on winter-run Chinook, because fishermen cannot distinguish between hatchery fall-run Chinook and endangered winter-run- and spring-run Chinook, and taking only a small number of fish from such a small population can have a large impact. Winter-run Chinook salmon are primarily impacted by fisheries south of Point Arena, California, due to their more southerly distribution in the ocean compared to other CV salmon stocks (NMFS 2016a). For the years 2000-2013, omitting 2008-2010 when the fishery was closed, approximately 19% of the age-3 winter-run were taken annually by the ocean fishery (PFMC 2015). The recreational fishery south of Point Arena has been closed since the early 2000s in February and March to protect winter-run Chinook, which are known to congregate in that area at that time of year (NMFS 2016a). The entire ocean fishery was halted in 2008-2010 because of the rapid decline of the fall Chinook population. Fall-run Chinook numbers have since increased sufficiently to reopen the fishery, but the winter-run Chinook population has not rebounded, indicating that even present incidental harvest rates may be excessive, when combined with other mortality factors (Kimmerer 2008).

Hatcheries. The long-term negative impacts of hatcheries on wild salmon populations are discussed in the Central Valley fall Chinook account. Two major concerns are (1) the effects of hatchery rearing of winter-run Chinook salmon on their behavior and genetics, because hatchery fish are increasingly contributing to the in-river spawning population (Williams 2006), and (2) the effects of competition from large numbers of hatchery fall-run Chinook when ocean productivity is low (Levin et al. 2001). The concerns boil down to the likelihood of hatcheries accelerating the decline of naturally-spawning Chinook of all runs. Hatchery proportion of LSNFH-origin spawners in the river has been steadily on the rise, increasing from an already alarmingly high of ~20% in 2005 to greater than 30% in 2012. NMFS places the population at a moderate risk of extinction from what they term "excessive hatchery influence" (NMFS 2016a).

In 2015, in response to extreme drought conditions the USFWS, NMFS, and CDFW collectively decided to re-initiate a captive broodstock program using juvenile hatchery fish from the LSNFH Conservation Hatchery Program. Program fish which will only be reared to maturity at the LSNFH and not released. This is a last resort safety measure with the stated goals, listed in order of priority, to provide: 1) a genetic reserve of winter-run Chinook salmon in a safe and secure environment to be available for use as hatchery broodstock in the event of a catastrophic decline; 2) a future source of winter-run Chinook salmon to contribute to multi-agency efforts to reintroduce them upstream of Shasta Dam and into restored habitats of Battle Creek; and 3) a future source to fulfill the needs of research projects (NMFS 2016a).

Alien species. Predation on juvenile salmon by non-native predators such as striped bass (*Morone saxatilis*), smallmouth bass (*Micropterus dolomieu*) and largemouth bass (*Micropterus salmoides*) and native species such as Sacramento pikeminnow (*Ptychocheilus grandis*) may be a major source of mortality of out-migrating juveniles. This is true especially in areas where structures concentrate large predators, such as Clifton Court Forebay at the South Delta pumps near Tracy (San Joaquin County). Preliminary results of 2013-2015 studies, which used hatchery winter-run smolts implanted with acoustic tags, showed survival rates to the ocean that varied from 5-12% (A. Ammann, NMFS, pers. comm. 2015). The highest mortality rates were observed in the middle reaches of Sacramento River, presumably due to predation (NMFS 2016a).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of the Sacramento River winter-run Chinook population. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years; a factor rated "high" could push the species to extinction in 10 generations or 50 years; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact. Certainty of these judgments is high based on peer reviewed and gray literature, direct observation, expert judgment, and anecdotal information. See methods for explanation.

Factor	Rating	Explanation
Major dams	Critical	Dams have blocked access to entire native spawning range, and reduced ESU to a single population below Keswick Dam that is dependent on coldwater releases from Shasta Reservoir.
Agriculture	High	Competition for limited cold water during drought is a major stressor. Levees protect and drain farmland but have reduced floodplain habitats by 95%. Diversions and drains may cause juvenile and adult mortality. Most locations have been screened, but south Delta water diversion and pumping remains one of the most pressing factors contributing to extinction risk.
Grazing	Low	Impacts likely small.
Rural /residential development	Low	Generally minimal impact on large river systems (e.g., Sacramento), but increasingly connected to urbanized areas.
Urbanization	Low	Water diversion and groundwater pumping can negatively impact cold water flows.
Instream mining	Low	Some legacy impacts from gravel mining.
Mining	Medium	Could be rated as "High" if Iron Mountain Mine contamination remains a major threat to all fish in the Sacramento River.
Transportation	Medium	Toxic spills from roads or the railway, while unlikely, represent a clear threat to the single population holding and spawning in an urban area bisected by multiple bridges.
Logging	Low	Potential impacts of logging could increase if winter-run are reintroduced into the upper portions of their historical watersheds.
Fire	Low	Little threat of fire on Sacramento River.

Estuary alteration	Medium	The San Francisco Estuary is heavily altered, to the detriment of all native species of fish migrating through it.
Recreation	Low	Redds can be disturbed by wading anglers. Boating, swimming, and fishing may impact holding and spawning behavior of winter-run but magnitude of impacts is unknown.
Harvest	Medium	The ocean fishery may harvest winter-run at unsustainable rates.
Hatcheries	High	NMFS places the population at a moderate risk of extinction from what they term "excessive hatchery influence."
Alien species	Medium	Predation from diverse predators is a source of juvenile mortality.

Effects of Climate Change: A recent NMFS summary of climate change impacts found that freshwater and marine productivity and survival tend to be lower in warmer years for most west coast salmon and steelhead populations (Williams et al. 2016). Winter-run are likely among the most 'at risk' salmonids because of their unique life history in which spawning and incubation takes place at the most thermally challenging time of the year. This makes them especially vulnerable to climate change and drought. Reduced availability of cold water leads to increased water temperature conditions degrading adult holding, spawning, and rearing conditions, but egg development is most acutely impacted. Climate change is likely to make maintaining cold water conditions on which the only naturally spawning population depends increasingly difficult. California's recent, historic drought (2012-2016) reduced the cold water pool in Shasta reservoir. Stream temperatures that exceed the 13.3°C (56°F) daily average temperature limit winter-run egg survival (NMFS 2016a): warm water during the periods of egg incubation and fry emergence were particularly severe during in 2014 and 2015 below Keswick Dam when the egg-to-fry survival was estimated to be 5.6% and 4.2%, respectively (NMFS 2016a), dramatically lower than the average of 26.4% for brood years 2002-2012 (Poytress et al. 2014).

Lindley et al. (2007) provide thermal suitability maps based on several warming scenarios that show that without passage around dams to cooler headwater areas, impacts to winter-run (and all other runs of Central Valley Chinook) may be severe. The hatcheries, too, are not immune from increases in drought frequency and severity associated with a warming climate, where low flows and high temperatures will likely increase the threat of disease (NMFS 2016a). Continued monitoring is critical, as well as developing adaptive management strategies should warming in their current habitat approach or exceed winter-run Chinook thermal tolerances.

Status Score = 1.3 out of 5.0. Critical Concern. Sacramento River winter-run Chinook salmon have a high likelihood of extinction, as reflected in their listing as an endangered species by both state and federal governments. Abundance has declined from having perhaps 200,000 fish divided among four populations, to having a few thousand (once a few hundred) in just one population. Lindley et al. (2007) indicate that catastrophic events in the region such as prolonged drought, catastrophic forest fire, or volcanic activity, could have extremely detrimental impacts on the population, particularly because there is now only a single population with no geographic redundancy. The proportion of hatchery-produced fish spawning in the wild is also on the rise. Winter-run salmon's continued persistence shows their remarkable resiliency, but they remain extremely vulnerable to loss or alteration of artificially maintained habitats, especially with respect to a warming climate that will make maintaining cold water conditions on which they depend increasingly difficult. Multiple critically dry years in a row or an extension of the most recent severe drought could potentially devastate the population.

In 1985, the California-Nevada Chapter of the American Fisheries Society (AFS) petitioned the National Marine Fisheries Service (NMFS) to list Sacramento River winter-run salmon as a threatened species under the Endangered Species Act (ESA) (NMFS 1997). In 1987, NMFS concluded that, while the decline was alarming, the conservation efforts that had already been implemented, in addition to those planned for the future, should enable recovery of the species without formal listing. This elicited a lawsuit by the Sierra Club Legal Defense Fund on behalf of AFS and eventually winter-run were listed as threatened in 1990. They were subsequently reclassified as endangered in 1994 (NMFS 1997) a status that was reconfirmed in 2005, 2011 and 2016. The State of California listed winter-run Chinook as endangered in 1989.

Table 2. Metrics to determine the status of Sacramento River winter-run Chinook, where 1 is poor value and 5 is excellent and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	1	A single population in a reach below dams; extirpated from their historical range.
Estimated adult abundance	2	The recent (2007-16) assessments indicate an average of < 3,000 returning spawners, therefore an effective population size of 600. In some years, the number of adults is below 1,000.
Intervention dependence	1	A captive broodstock program has been initiated to augment the conservation hatchery program at Livingston Stone Fish Hatchery. The naturally spawning population depends entirely on cold water releases from Shasta Dam.
Tolerance	1	Winter-run Chinook spawn in the most thermally challenging times of the year and are particularly at risk to high temperatures and low dissolved oxygen levels.
Genetic risk	2	Considerable genetic drift resulting from consolidation of winter-run populations into a single population is likely exacerbated by the large influence of hatchery broodstock.
Climate change	1	Extremely vulnerable because of reliance on cold water habitat and releases from Shasta Reservoir.
Anthropogenic threats	1	1 Critical, 2 High, 5 Medium threats.
Average	1.3	9/7.
Certainty	4	Well-studied population.

Management Recommendations: Since the ESA listing in 1990, there have been a great number of conservation measures instituted to help support populations of winter-run Chinook salmon, including:

- Opening the gates year-round at Red Bluff Diversion Dam to allow passage of adults and juveniles and minimize predation.
- Installation of a temperature control device on Shasta Dam in 1997 to provide a continuous supply of cold water and improve dissolved oxygen and turbidity levels.
- Construction and operation of the Livingston Stone Conservation hatchery.
- Screening almost all large water diversions on the Sacramento River.
- Construction of barriers at the Knights Landing Outfall gates (2015) and Wallace Weir

(2016) to block adults from straying into the Colusa Basin Drain.

- Attempts to stabilize mainstem flows so as to not dry out winter-run redds.
- Ocean harvest restrictions.
- Attempts to reduce entrainment and increase survival for entrained individuals at the south Delta export pumps.
- Dumping of large quantities of gravel below Keswick Dam to improve spawning habitat.

Improving habitats for spawning and rearing of Chinook salmon in the Central Valley remains an ongoing process. Two major initiatives will help with the recovery of winter-run: 1) reintroduction to Battle Creek and 2) reconnection of off-channel/floodplain rearing habitats.

Battle Creek. In 1999, Pacific Gas and Electric Company (PG&E), National Marine Fisheries Service, U.S. Fish and Wildlife Service, U.S. Bureau of Reclamation, and California Department of Fish and Wildlife reached an agreement to restore salmon and steelhead in Battle Creek, one of the original four watersheds of winter-run Chinook. Restoration and re-colonization of Battle Creek could provide 67 km (42 mi) of additional spawning habitat for all five of the anadromous salmonid runs as well as higher instream flows and cooler temperatures. The project will remove five dams, install fish screens and ladders on three dams, and end the diversion of water from the North Fork to the South Fork specifically to benefit listed salmon runs (NMFS 2016a), with full implementation expected by 2020. Restoration of floodplains in lower Battle Creek will offer critical support to establishing and sustaining a winter-run population should they be reintroduced to the watershed. Reconnecting the floodplains of Rancho Brisgau, once known as Bloody Island, located at the confluence of Battle Creek and the Sacramento River, would benefit all runs of Chinook from both streams.

When cold-water habitat is restored, Battle Creek will represent the only location, other than the mainstem Sacramento River where a winter-run population can swim through their entire life cycle. Such is not the case for proposed reintroduction to the McCloud River, which will involve relocation of adults over Shasta Dam (likely in so a so-called "trap and haul" program) and capture of juveniles in the McCloud River or Shasta Reservoir for trucking and relocation back to Sacramento River below Keswick Dam. NMFS has proposed that winter-run Chinook should be re-introduced to the McCloud River using two-way trap and haul (TH2), in which adults are transported around dams to spawn and juveniles are then captured as they migrate downstream and released in the Sacramento River below Shasta Dam (NMFS 2016a). Lusardi and Moyle (*in press*) indicate that any such trap and haul program should proceed with extreme caution. They view such aggressively interventionist efforts not as restoration efforts but as extinction prevention efforts and as such they should only be implemented after all other restoration measures have failed. If a reintroduction above Shasta were to be carried out it move forward under an experimental framework to designed to resolve potential uncertainties in adaptive fashion. Analysis of existing TH2 programs in the Pacific Northwest suggests they rarely, if ever, work for effectively increasing salmon abundance. Typically the greatest difficulties involve capturing out-migrating juveniles, which is likely to prove particularly challenging in the McCloud arm of Shasta Reservoir where surface water elevation can fluctuate dramatically.

Other concerns of a winter-run Chinook trap and haul program include its high cost and need for perpetual funding, making the species particularly vulnerable to future economic trends. Another danger is moving precious few adults above the dam creates a "population sink," whereby above dam populations that cannot sustain themselves actually drag down the below dam populations as well, and the major changes to the McCloud River since winter-run were extirpated from it including diversion of over 75% of its summer base flow into the Pit River as part of PG&E's Pit-McCloud hydropower project (DFW 2004).

Floodplain reconnection. Del Rosario (2013) has shown that winter-run spend about three months on average rearing in the lower river, estuary, and Delta. This is a considerably more time than other runs, which makes sense when you consider that the majority of winter-run juveniles have entered the Delta before most fall run hatch from gravel. The extended rearing in the food-rich, off-channel habitats of the lower Valley was likely one of the drivers of evolution of the unique winter-run life history. Since European settlement, over 95% of floodplain and tidal marsh habitats have been drained, which drastically diminishes the benefits to fish rearing in the lower system. Accordingly, restoration of floodplains and tidal marshes is viewed as a critical recovery action for the species. To mitigate for the ecological impacts of water development, the legally binding Biological Opinion for the operation of the State and Central Valley Water projects calls for restoring 17,000 to 20,000 acres of floodplain and tidal marsh habitat for juvenile salmon rearing in the Lower Sacramento River and Delta. These actions are primarily aimed at increasing the frequency with which the Yolo Bypass floods and thus gives access to and provides habitat for rearing juvenile salmonids. This process began in 2007 and restoration actions are scheduled to be completed by 2023.

Off-channel and floodplain restoration actions are needed not just in the Delta and lower river but throughout the system. In the upper and mid reaches of the Sacramento River a thorough assessment of all potential floodplain reconnection opportunities within the federal levee system needs to be carried out. There is no one perfect project type, but instead a portfolio of projects, which hydrologically reconnect side channels, oxbow lakes, and varied small floodplains (such as the Willow Bend floodplain restoration north of Colusa) must be aggressively pursued along the entire length of CV river systems. In the lower River and Delta this approach is being used to implement a suite of projects to address the aquatic, sub-tidal, tidal, riparian, floodplain, and upland ecosystem needs are being pursued by a multitude of entities and funding sources including the state and federal public water agencies required to mitigate the ecological impacts of the CVP and SWP, the Central Valley Project Improvement Act, Propositions 1 and 1E, the AB 32 Greenhouse Gas Reduction Fund, and local, federal, and private investment. Yet there remains no central venue to bring the many players together to guide planning, set specific regionally explicit salmon habitat targets, and implement priority projects. The launch of the Central Valley Salmon Habitat Project in spring 2017 is envisioned to fill this critical gap.

SOUTHERN OREGON/NORTHERN CALIFORNIA COASTAL CHINOOK SALMON
Oncorhynchus tshawytscha

Moderate Concern. Status Score = 3.1 out of 5.0. Southern Oregon-Northern California Coastal (SONCC) Chinook salmon in California are limited to a few watersheds but populations remain stable. They may be vulnerable to stochastic events due to small population size and limited range within California.

Description: Chinook salmon can be distinguished from other salmon species by the many black spots on their back, dorsal fins and both lobes of the caudal fin, as well as by the dark pigment along gums in the lower jaw. Morphological characteristics of SONCC Chinook salmon are as follows: fin ray counts are 10-14 (dorsal fin), 14-19 (pectoral fin), 10-11 (pelvic fin), and 13-16 (anal fin) (Snyder 1931, Schreck et al. 1986). Scales along the lateral line number 131-147. They are also characterized by 93-193 pyloric caeca, 13-18 branchiostegal rays and rough, widely-spaced gill rakers, 12-13 of which are on the lower half of the first gill arch.

Adult lengths can be greater than 140 cm SL, but usually fall between 75 and 80 cm SL. Smith River Chinook salmon regularly attain larger sizes, with a majority of fish retained by anglers ranging from 88 and 100cm SL (CDFW 2002, 2006, 2007). SONCC Chinook salmon range widely in size, but when compared to Sacramento River Chinook of the same length, are more rounded and heavier (Snyder 1931). Adult Chinook salmon in California can reach weights of 38.6 kg, but average between 9-10 kg. Sexually mature adults are uniformly colored in dark burgundy or olive brown. Males develop humped backs and hooked jaws and are usually darker than females. Chinook juveniles have 6-12 parr marks equal in width or wider than the spaces between them and an adipose fin with dark coloration along the upper edge only. Although some parr develop spots on the dorsal fin as they grow, most have clear dorsal fins.

Taxonomic Relationships: The SONCC Chinook salmon ESU is distinguished from other ESUs based on genetic analyses. Analysis of microsatellite loci and older allozyme datasets designated Chinook from the Klamath River and Blue Creek (lower Klamath River) into two clusters within the Klamath Basin (Myers et al. 1998). The SONCC Chinook salmon ESU contained genotypes from Blue Creek, which clustered with those from streams north of the Klamath River, including southern Oregon, based on microsatellite DNA. Southern Oregon Northern California Coastal Chinook salmon from the Smith River and Blue Creek also share morphological traits and age (3 years) of maturity (Snyder 1931). In Blue Creek, there is also a late fall-run which seems to be segregated from other fish (Gale et al. 1998). Although spring-run Chinook return to the Smith River, the relationship between these and fall-run SONCC Chinook is not well understood. Myers et al. (1998) regard the few spring-run Chinook in SONCC Chinook streams to be part of the ESU.

Life History: Most SONCC Chinook spawning adults migrate into rivers in the late fall, when increases in stream flow facilitate access into streams. Adults enter tributaries of the lower Klamath River from September through December and spawning occurs in the latter part of this period and into January (Leidy and Leidy 1984). In the Smith River, migration may start as early as late August and continue through early January. Run timing peaks are generally observed from late November to early December, depending on location (Larson 2013, Walkley and Garwood 2017) with spawning typically occurring between October and February. Chinook salmon enter Blue Creek in September and spawning peaks after fall rains, usually in November, but may continue through December, reflecting differences in reproductive maturity between earlier and later arrivals (Gale et al. 1998). Differences in reproductive behavior were observed for females in Smith River tributaries. The amount of time that a female spent on a redd decreased from 10-21 days to 5-10 days as the spawning season progressed and river conditions changed (Waldvogel 2006). SONCC Chinook age at spawning is variable by habitat and can change over time. Spawners in Blue Creek are primarily age-3 with a few age-4 and age-5 fish. A few grilse, reproductively mature age-2 fish, also return to spawn (Gale et al. 1998). In Mill Creek (Smith River), from 1993-2002, most spawners were age-3 (62%), but, from 1981-1992, 4 year-old females comprised the majority of spawners (66%, Waldvogel 2006).

Chinook salmon fry emerge in lower Klamath tributaries from February (Parish and Garwood 2016) through mid-April, some with yolk sacs still intact, and most migrate to the ocean in the same year (Leidy and Leidy 1984). In 1995-96, fry outmigration from Blue Creek began before mid-March, peaked in late April and late May, and continued into August (Gale et al. 1998). Fry grew to 103 mm FL throughout the period of outmigration (Gale et al. 1998). Early outmigrants generally traveled quickly into the estuary, though there is considerable variability in timing and residence times among CC Chinook watersheds. For example, Wallace (2003) found peak juvenile migration through the Klamath estuary occurred between June and July from 1997-1999. During this time, estuary residence was found to vary significantly from one year to the next (8 to 16 days in length) and was positively correlated with prevailing streamflow conditions (Wallace 2000). More recently, Wallace (2010) compiled juvenile outmigration data and found annual median travel times of captured coded-wire tagged Chinook from hatchery to estuary ranged from 30-34 days for Iron Gate Hatchery (IGH) fall Chinook, 10-32 days for Trinity River Hatchery (TRH) spring Chinook, and 23-75 days for TRH fall Chinook (Wallace 2010). Larger juveniles can spend months rearing in freshwater before outmigration (Sullivan 1989).

Zajanc (2000) found that estuary residence time for outmigrating individuals was variable, with late migrating individuals (August) rearing for longer periods than early outmigrants (June-July). In the Smith River, juvenile Chinook were observed in both low salinity zones (<5%) in the upper estuary with abundant cover from overhanging riparian vegetation (Quiñones and Mulligan 2005), while Parish and Garwood (2015, 2016) found Chinook YOY rearing in open, freshwater portions of the lower estuary lacking cover in spring. Estuarine residence times in the Smith River estuary were relatively short (eight to 40 days) compared with other Pacific coast estuaries (e.g., Nicholas and Hankin 1988), and individuals generally showed rapid growth during the early and late summer rearing periods, but not when temperatures were highest during mid-summer (Zajanc 2000).

Ocean survival of Chinook is likely enhanced by larger body size associated with longer periods of rearing in fresh water (Williams et al. 2016). Reedy (1995), Garwood and Larson (2014), and Parish and Garwood (2015) found that juvenile Chinook exhibit a strong stream-type life history on the Smith River to allow them to grow before undertaking taxing ocean migrations, with several months of freshwater rearing observed. Outmigration generally occurs from spring through the summer, but relatively high juvenile stream occupancy rates were observed as late as September. For example, 28% of juvenile outmigrants from Blue Creek in 1996 reared extensively in freshwater (McCain 1994) first, while only about 5% of juveniles rearing in Hurdygurdy Creek (Smith River) in 1987 and 1988 remained in the stream to rear after spring flows receded (McCain 1994). It is possible that high flows in the spring of 1988 shortened freshwater residency in that year. Juvenile Chinook salmon in tributaries of the Sixes River, Oregon, (northern range of SONCC Chinook) also displayed varying degrees of freshwater residency; some moved into the ocean within weeks of emergence, while others reared in freshwater from two months to more than one year (Reimers 1971). Scale aging revealed that most adults returning to spawn had reared in freshwater for two to six months as juveniles (Reimers 1971).

Once in the ocean, Chinook seem to follow defined migration routes to a cool pool of water offshore of the Klamath-Trinidad region (Harding 2015) and the food-rich waters of the North Pacific, but are capable of altering migration patterns to use regions with temperatures of 8°-12°C (Hinke et al. 2005), presumably to follow favored or abundant prey.

Habitat Requirements: SONCC Chinook generally use large cobbles and require sufficient flows to facilitate oxygen delivery to developing embryos. Most SONCC Chinook salmon spawn in the middle reaches of coastal tributaries, but in the Smith River, small tributaries are commonly used for spawning (Walkley and Garwood 2017). In Blue Creek, holding spawners favored deep pools and areas with runs and pocket water with fast flows (Gale et al. 1998). Adults have been observed spawning at depths ranging from a few centimeters to several meters, with water velocities of 15-190 cm/sec. However, preferred spawning habitat depths range from 25 to 100 cm, with water velocities from 30 to 80 cm/sec (Moyle 2002). Embryo survival is enhanced when water temperatures stay between 5°-13°C and oxygen levels are close to saturation (Healey 1991). Water temperature requirements of Chinook salmon are discussed in Moyle et al. (2008). Embryos incubating in optimal conditions generally hatch within 40-60 days (temperature dependent), but remain in the gravel as alevins for an additional 4-6 weeks, usually until the yolk sac is absorbed. Juveniles will continue to rear in streams throughout the summer if water temperatures remain < 20°C (Gale et al. 1998), though Parish and Garwood (2015) observed rearing Chinook in the Smith River at temperatures up to 23°C. Rearing habitats are generally characterized by shallow water in areas with overhanging riparian vegetation that provide cover, food and habitat complexity in stream reaches, though they will move to deep water to shoal and feed in streams and estuaries (J. Garwood, CDFW, pers. comm. 2017).

Distribution: Southern Oregon-Northern California Coastal Chinook salmon range from Cape Blanco, OR (near the Elk River) south to the Klamath River, including Klamath River tributaries from the mouth to the Trinity River confluence. In California, SONCC Chinook salmon were historically found in most small tributaries of the lower Klamath River that are within the ocean-influenced fog belt such as Hunter, Terwer, McGarvey, Tarup, Omagar, Blue, Surpur, Tectah, Johnson, Mettah, and Pine creeks (USFWS 1979). In 1999-2000, Gale and Randolph (2000) found Chinook in Hoppaw, Saugep, Waukell, Bear, Pecwan, and Roaches creeks, but not in Omagar and Surpur creeks. Southern Oregon-Northern California Coastal Chinook in California are currently found in only a few small lower Klamath tributaries (e.g., Blue Creek), Wilson Creek (Del Norte Co.) and the Smith River (J. Garwood, CDFW, pers. comm. 2017).

Chinook salmon from the Rogue River, OR, and Smith River, CA have different ocean migration patterns than Chinook salmon in ESUs to the south (Gale et al. 1998), with a greater tendency for adults in the ocean to stay north of Cape Blanco (Brodeur et al. 2004), while upper Klamath-Trinity rivers Chinook salmon stocks tend to associate with the California Current further south (Harding 2015).

Trends in Abundance: The majority of SONCC Chinook salmon originate from the Rogue River in Oregon, while individuals from the lower Klamath River tributaries and Smith River contribute to the population but to a lesser extent. Abundance of the fall-run appears stable, although populations in the Klamath basin have been adversely affected by land use practices, particularly logging. Spring-run Chinook salmon appear to have largely disappeared from this ESU (Moyle 2002). The numbers of spring-run Chinook adult in the Smith River were probably always low, with 0-38 fish counted on the South Fork from 1982-2016, and 0-17 fish counted on the Middle Fork from 1991-2016 in snorkel surveys (54 to 85 km surveyed, Parish 2016).

There is considerable natural variability in the number of fall-run Chinook observed from year to year in the ESU. Historically, some 2,000-3,000 adult Chinook salmon spawned in the lower Klamath River each year (Moyle 2002). In 1960, an estimated 4,000 Chinook salmon spawned in lower Klamath tributaries (USFWS 1979) while, in 1978-79, the number of spawners dropped to around 500 (USFWS 1979), and estimates of returning adults have fluctuated but generally been relatively low since. In 1995 and 1996, respectively, 236 and 807 fall Chinook salmon were observed in Blue Creek (Gale et al. 1998). A study of spawner survey methodology and effectiveness of Chinook in Blue Creek indicated that surveys generally only count about half the actual number of spawners, with spawner estimates in survey years (1995- 2009) ranging from 100-2,400 fish (Antonetti 2009).

More recently, the numbers of late fall-run Chinook spawning in Blue Creek from 1988 to 2009 followed an increasing trend (Quiñones at al. 2014). The time series is significantly correlated to hatchery returns, suggesting that numbers are supplemented by hatchery strays or that hatchery fish encounter similar ocean conditions as naturally produced fish of the same cohort. In 2011, spawning surveys indicated a record peak count of 1,561 late-fall Chinook (adults and jacks) in Blue Creek, with a total population estimate of 6,100 (± 876). This was the highest peak count of spawners recorded since surveys commenced in 1988 (Antonetti and Partee 2013, Figure 1).

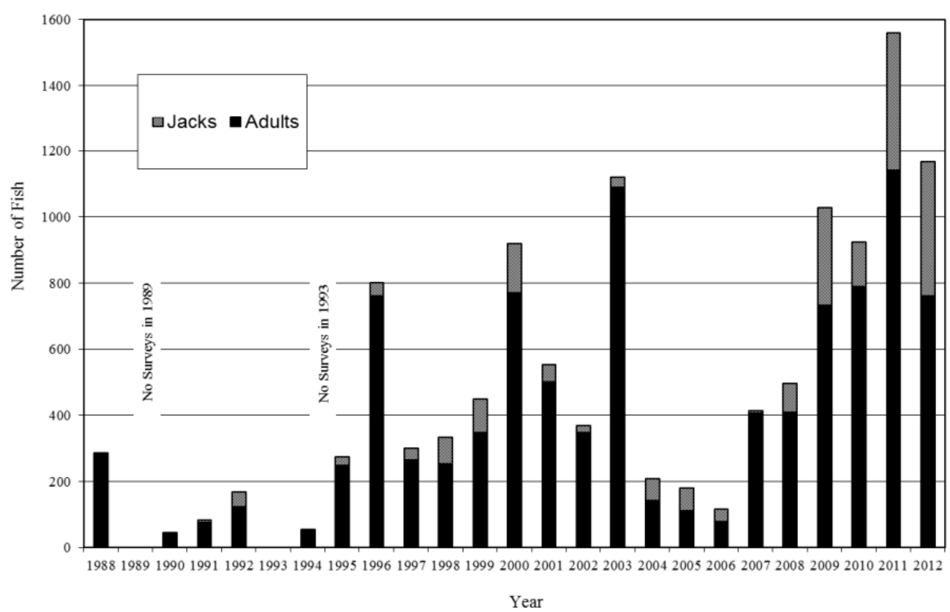


Figure 1. Annual peak count of late-fall Chinook salmon in Blue Creek, 1989-2012. From: Antonetti and Partee 2013. Fig. 5, page 18.

Annual abundance of adult Chinook salmon in the Smith River is estimated to range from 15,000-30,000 fish (Moyle 2002), but robust population estimates have historically not been established. However, using sonar technology, cross-referenced with Rowdy Creek weir fish counts, Larson (2013) estimated adult escapement in the Smith River Basin to range between 22,500 and 20,000 individuals during 2010-2011 and 2011-2012, respectively. Documented spawning redds (a proxy for adult returns) have varied over the last five years in the Smith River, with a peak during 2011-2012 (3,819, 95% CI: 2,777-4,860), a trough during 2013-2014 (516, 95% CI: 284-770) (Garwood and Larson 2014), and a rebound in 2014-2015 (1,715, 95% CI: 1,092-2,337) (Walkley and Garwood 2015). These estimates do not cover all potential Chinook spawning habitats because they were designed to estimate spawning coho (Garwood and Larson 2014). More recent redd estimates for the 2015-16 spawning season (Walkley and Garwood 2017) are similar to 2014-2015, with an estimated 2,000 redds, plus or minus about 1,000.

Forty-six percent of all Chinook redds were observed in Rowdy Creek, while 31percent were observed in Mill Creek during the 2015-16 spawning season (Walkley and Garwood 2017). Stream spatial structure of the population (e.g., proportion of area occupied by juvenile Chinook) in the Smith River watershed appeared relatively stable over the last several years, though summer occupancy dropped during 2014 (coincident with drought effects), and rebounded in 2015 (Walkley and Garwood 2015). There is no evidence of a declining trend in fall-run spawner abundance in the Smith River, and so populations are assumed to be stable.

Factors Affecting Status: SONCC Chinook salmon abundance in California appears to be mostly limited by habitat alteration, hatcheries and harvest. The ESU may also be vulnerable to stochastic events due to small population size and limited range.

Dams. The Smith River is undammed but, in the Klamath River, flow regulation by mainstem dams may affect migration timing and health of adults in the main-stem river prior to entering smaller tributaries. These dams may also negatively affect juveniles outmigrating from the system by reducing peaks of freshets or pulse flows after storm events. In addition, flows regulated by dams in the Klamath River mainstem can also adversely affect migrating Chinook salmon through exposure to high water temperatures that increase the incidence of disease (Belchik et al. 2004).

Agriculture. In the Smith River estuary, construction of dikes and reclamation of lands for agriculture and grazing have reduced the amount of juvenile rearing habitat by more than 40% (Quiñones and Mulligan 2005). Diversions of water for flower bulb cultivation, alfalfa production, pasture irrigation, and other purposes may affect salmon outmigration, depending on seasonal timing and volume of water diversions. Concern over long-term pesticide use, which has known toxicity to aquatic invertebrates, at lily bulb fields adjacent to the estuary has initiated limited water quality testing by the State Water Resources Control Board (Parish and Garwood 2015, Cal EPA 2017).

Grazing. Grazing of riparian areas by feral cattle has been identified as significant cause of habitat degradation in the lower Klamath in the vicinity of the Blue Creek drainage, causing stream bank sloughing and reduced riparian vegetation (Fiori and Beesley 2013). Cattle grazing along the Smith River estuary has also degraded stream banks and reduced or eliminated riparian vegetation (Quinones and Mulligan 2005). Numerous road and cattle crossings have been identified for fish passage restoration, particularly those in the lower watershed on Tyron and Morrison creeks near the Smith River estuary (Parish and Garwood 2015).

Transportation. Roads, including highways, have been identified as a major source of habitat loss in SONCC Chinook streams. However, road building is intimately associated with logging in the Klamath Mountains; see below.

Logging. The coastal watersheds of northern California have been heavily logged, beginning in the mid-19th century (USFWS 1979). Logging has altered most coastal streams by increasing solar input and water temperatures through reduced tree canopy cover, introduction of heavy loads of fine sediments that bury spawning gravels and fill pools, and increased surface runoff of precipitation, leading to increased frequency of flash flooding in streams. Removal of large trees has removed important sources of large woody debris, which provides cover for all life stages of salmonids. In many streams, extensive networks of logging roads (mostly unimproved) in north coastal drainages have blocked salmon spawning migrations. Improperly built stream crossings (culverts, bridges, cattle crossings, and other structures) have created fish passage barriers, impeding fish passage although, in recent decades, many passage impediments have been rectified. Road construction in lower Blue Creek has also altered stream morphology and reduced recruitment of large woody debris into the stream channel (Beesley and Fiori 2008). Roads have increased fine sediment delivery to streams in the Smith River basin (Six Rivers National Forest 2013).

Fire. Most lower Klamath and Smith River tributaries are within the marine fog belt, with cooler temperatures and higher fuel moisture that inhibit wildfires; however, in recent years, inland portions of the Smith River watershed have suffered catastrophic wildfires (e.g., Biscuit Fire in 2002 and Coon, Bear, Peak and Buckskin fires during the summer of 2015) that can potentially degrade tributary and main-stem habitats.

Estuary alteration. The capacity of the Smith River estuary to support juvenile salmon rearing has been greatly reduced due to prevailing land uses and associated habitat degradation. Specifically, levees, dikes, tide gates, and rip-rap have been used extensively in the estuary to control flooding and improve bank stability, and much of the emergent tidal wetlands have been eliminated as a result (Parish and Garwood 2015). Such modifications have disconnected the estuary from historically productive lateral habitats, such as sloughs, with negative consequences for juvenile salmonids that use these areas for rearing (Parish and Garwood 2015).

Harvest. Commercial, sport, and tribal fisheries have likely reduced SONCC Chinook salmon abundance in the past. However, recent regulations to protect Upper Klamath-Trinity rivers Chinook from overharvest (e.g., closure of fishery in 2006, 2008 by Pacific Fisheries Management Council) may have reduced harvest rates of SONCC Chinook salmon from the lower Klamath and Smith rivers in recent years. Mixed-stock commercial fisheries at sea do not currently have the ability to differentiate between Chinook of different ESUs or runs, and so inadvertent harvest of SONCC Chinook likely still occurs, though at unknown levels. Freshwater harvest rates of SONCC Chinook are also largely unknown and require further study.

Hatcheries. Although hatcheries are not operated in tributaries to the lower Klamath River, SONCC Chinook in the basin are likely interacting with salmon produced by hatcheries on the main-stem Klamath (Iron Gate Hatchery) and Trinity (Trinity River Hatchery) rivers. Hatchery-produced juvenile Chinook salmon migrate through the middle Klamath River in late summer (USFWS 2001), around the same time that wild SONCC Chinook are also outmigrating. Hatchery-produced adults may stray into lower Klamath tributaries, perhaps interbreeding with and altering the genetic makeup of wild SONCC Chinook salmon. Returns of adult Chinook salmon to Blue Creek was found to be significantly correlated with returns of adult Chinook salmon to Trinity River Hatchery, suggesting that hatchery strays are contributing to the population (Quiñones 2013). In the Smith River basin, about 50 female Chinook salmon are spawned each year by Rowdy Creek Hatchery which began operation in 1973. CDFW has a production goal of 100,000 smolts annually and requires that all individuals are marked with a fin clip (Garwood and Larson 2014). Smolts are released during spring and have been observed displacing other salmonids (e.g., steelhead trout) from estuarine habitats (Quiñones, pers. obs, 1997-2001). During the winter of 2011-2012, approximately 23% of all recovered Chinook salmon carcasses in the Smith River basin were of hatchery origin (Garwood and Larson 2014). Some Smith River tributaries also have > 5% straying rates from Chinook from the Rowdy Creek Fish Hatchery (Walkley and Garwood 2017). Numerous studies have shown that interactions between wild and hatchery origin salmonids can produce significant negative effects on wild fish (Reisenbichler and Rubin 1999, Araki et al. 2008, Chilcote et al. 2011).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of SONCC Chinook salmon in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	Low	No dams on the Smith River, but Klamath River dams affect migration patterns and reduce habitat suitability.
Agriculture	Low	Agriculture is primary land use in Smith River estuary; wetland reclamation, diking, diversions, and pollutant and pesticide inputs; however, potential effects have not been studied.
Grazing	Medium	Cattle grazing in the Smith River estuary has led to habitat degradation; grazing in Blue Creek drainage has substantially impacted riparian and aquatic habitats.
Rural/ residential development	Medium	Rural development is increasing in north coastal California watersheds, contributing to habitat degradation, water diversion, and pollutant inputs into streams.
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Primary sources of sediment inputs in SONCC watersheds.
Logging	Medium	Most watersheds have been heavily logged in the past; legacy effects remain in many watersheds.
Fire	Medium	Predicted increases in severe wildfires may lead to increased habitat degradation, especially outside fog belt.
Estuary alteration	Medium	Land reclamation, dikes, tide gates for agriculture in the Smith River estuary has significantly reduced juvenile rearing habitats.
Recreation	Low	Most small tributaries not heavily used by swimmers and boaters.
Harvest	Medium	Harvest at sea has presumably reduced Chinook numbers to a fraction of historical numbers; freshwater harvest likely less but requires more study.

Hatcheries	Medium	Hatchery fish probably negatively affect Klamath River populations; impacts to the main population in the Smith River may be minimal, though more research is required.
Alien species	Low	Few alien species reported for Klamath and Smith rivers.

Effects of Climate Change: Moyle et al. (2013) rated SONCC fall-run Chinook as "highly vulnerable" to extinction in the next 100 years as the result of the added impacts from climate change, although uncertainty in this regard is high. Predicted climate change impacts to north coastal streams are expected to be less than those to inland waters in California, since the maritime climate and associated fog belt will likely offset air temperature increases. However, coastal areas have already experienced a 33% reduction in fog frequency since the early 20th century and further reduction is predicted to increase summer drought frequency and duration along the west coast (Johnstone and Dawson 2010). Predicted increases in air temperatures (up to 10°C by 2100; Dettinger 2005), in combination with reduced fog frequency and associated increases in evapotranspiration, may negatively impact juvenile rearing habitat (e.g., warmer water temperatures, lower stream flow). While prospects for long-term survival of Chinook salmon in many watersheds in California are poor, the Smith River watershed (Wild and Scenic River) presents perhaps one of the few basins with potential resilience to climate change based on its current high water quality, no dams or major diversions, lack of development, high summer base flow, and watershed protections (J. Garwood, CDFW, pers. comm. 2017).

Poor ocean conditions (e.g., reduced upwelling, higher temperatures), may also reduce ocean survival and limit gene flow between more northern populations and are expected to become more frequent off the coast of California in the future, to the detriment of all salmonids (Williams et al. 2016). Based on historical analyses of changing sea surface temperatures and survival, Sharma et al. (2013) estimated that a 1°C increase in average sea surface temperatures in the spring and early summer, while juvenile salmon transition from freshwater to salt, may result in 1-4% reductions in survival across their range. Such increases are well within the expected bounds of sea surface temperature increases due to climate change by 2100 (Mantua 2015). In addition, sea level rise will likely reduce rearing habitats in estuaries, unless similar habitats become available in upstream areas as estuaries 'back up' in response.

Status Summary Score = 3.1 out of 5.0. Moderate Concern. The SONCC Chinook salmon ESU in California is limited to a few watersheds that are impaired, to varying degrees, by habitat degradation associated with land and water use practices. This ESU was determined by NMFS on September 16, 1999 to not warrant listing under the Federal Endangered Species Act, although SONCC Chinook salmon are considered a Sensitive Species by the U.S. Forest Service. While there are no discernable declining trends in abundance of SONCC Chinook at this time, this ESU should be monitored closely and better information and surveys (e.g. Walkley and Garwood 2017) must be continued and expanded to help determine changes in spawner success, species distribution, and abundance over time for all salmonids.

Table 2. Metrics for determining the status of SONCC Chinook salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	4	Blue Creek and Smith River are the principal populations, along with smaller populations in tributaries.
Estimated adult abundance	3	Between 5,000 and 50,000 spawners in the Smith is probable in most years; in 2011-2012, adult spawners were estimated at 20,000. <1,500 spawners in Klamath tributaries in most years.
Intervention dependence	4	California populations are largely self-sustaining but some supplementation by hatcheries is likely.
Tolerance	3	Multiple juvenile life histories and spawner age diversity demonstrate physiological tolerances.
Genetic risk	3	Limited hatchery operations in California, but some concern for hybridization with hatchery 'strays' from Rowdy Creek Fish Hatchery and other ESUs.
Climate change	2	Fall-run is least vulnerable to climate change as they spawn later in and scouring of redds is less likely to influence juveniles; sea level rise may negatively affect rearing in estuaries; Smith River likely to retain runs under worst-case scenarios through end of the century. Vulnerable to increasing temperatures, changes in flow regimes in tributaries, and variable ocean conditions.
Anthropogenic threats	3	Multiple threats rated as "Medium."
Average	3.1	22/7.
Certainty (1-4)	2	Least studied of Klamath River Chinook runs.

Management Recommendations: The persistence of the two largest populations of SONCC Chinook salmon (in Smith River and Blue Creek in the lower Klamath River) suggests that conservation of this ESU within California is largely reliant upon protection of spawning and rearing habitats in these two watersheds. Increased protection of these populations would also facilitate recolonization of other degraded streams in the ESU, as habitats recover and are restored, potentially expanding the distribution and increasing the abundance of SONCC Chinook salmon. The Smith River, in particular, may be one of only a few rivers in California that has built in resiliency to a changing climate due to its uninterrupted flow, relatively intact watershed, little development, and cool temperatures. More monitoring and restoration funding should be allocated to its lower tributaries (such as Rowdy and Mill creeks) and especially the estuary, which is relied on by SONCC Chinook, SONCC coho, chum, and pink salmon as well as Northern California steelhead, coastal cutthroat trout, and myriad other native fishes.

It has been shown that interactions of wild Pacific salmon with hatchery-produced salmon can reduce both the overall fitness of a population (Araki et al. 2008), its local adaptability (Reisenbichler and Rubin 1999), reduce long-term resilience of the population, and increase threat of extinction (Chilcote et al. 2011, Johnson et al. 2012). The introduction of hatchery salmon from Rowdy Creek Hatchery on the Smith River may therefore be in conflict with the status of the Smith River as a 'stronghold' for wild salmon and should be re-evaluated.

To determine the status of Chinook salmon within this ESU, both population monitoring and genetic studies are needed to determine levels of introgression between wild and hatchery stocks and to determine the status of spring-run Chinook salmon within this ESU. Such studies may be of particular value in the Smith River drainage (the largest free-flowing river system in the state) which has been designated a National Recreation Area, and is included in the National Wild and Scenic River program. In 2016, a proposal to allow mining in the North Fork Smith River watershed in Oregon was defeated, and the state of Oregon voted to enact a twenty-year ban on mining in this critical watershed. Such protections that ban harmful mining that can endanger the entire watershed should be banned permanently. These designations imply that priority should be given to maintaining self-sustaining, wild populations of native salmonids and other organisms.

UPPER KLAMATH-TRINITY RIVERS FALL-RUN CHINOOK SALMON

Oncorhynchus tshawytscha (Walbaum)

Moderate Concern. Status Score = 3.1 out of 5.0. Abundance of natural spawners in most tributaries has declined from historical levels. The proportion of natural-origin spawners returning to the Upper Klamath-Trinity rivers basin has varied widely over the past decade, though a recent decreasing trend in the proportion of hatchery-origin adults returning to the Trinity River to spawn may signify a stabilization of the population.

Description: See the upper Klamath-Trinity rivers (UKTR) spring-run Chinook and Central Valley fall-run Chinook accounts in this report for detailed species descriptions. UKTR fall-run Chinook enter rivers as reproductively mature fish (ocean-maturing ecotype) exhibiting spawning colors (Kinziger et al. 2008, Prince et al. *In press*). Klamath River Chinook spawning adults are considered to be smaller, more rounded, and heavier in proportion to their length compared to Sacramento River Chinook (Snyder 1931)

Taxonomic Relationships: Fall- and spring-run Chinook salmon trace their evolutionary origins to geologic events during the Pleistocene, when changes on the landscape (mountain range formation, glaciations, volcanic activity) created spatial segregation among populations (Waples et al. 2008). Within California, spring- and fall-run Chinook life histories likely evolved independently and repeatedly in different basins over time, as evidenced by the fact that fish within basins are more closely related than those with shared life histories in other basins (Prince et al., *In press*). Further, fall- run Chinook salmon populations from Klamath and Trinity subbasins appear more similar genetically to the respective spring-run Chinook populations within a given subbasin than they are to fall-run Chinook in Lower Klamath River tributaries (M. Miller, UC Davis, unpubl. data 2016). Recent research indicates genetic variation on the GREB-1L and Omy-5 loci of the genome, which are associated with fat storage, sexual maturation, and run timing in *O. mykiss and O. tshawytscha*, largely differentiate fall-run and spring-run Chinook (M. Miller unpubl. data 2016, Prince et al. *In press*).

The UKTR Chinook salmon Evolutionarily Significant Unit (ESU) includes all naturally spawned populations of Chinook salmon in the Klamath River basin, upstream from the confluence of the Klamath and Trinity rivers. The UKTR Chinook salmon ESU is genetically distinguishable from other California Chinook ESUs (Waples et al. 2004, Kinziger et al. 2008). Although fall-run and spring-run Chinook salmon are both part of this ESU, the two runs are treated here as separate taxa due to their distinctive ecotypes, adaptive life histories, differing migration and foraging strategies at sea (Tucker et al. 2011), and discrete genetics (Kinziger et al. 2008, Prince et al. *In press*). See the UKTR spring-run Chinook salmon account for further details on taxonomy within this ESU.

Life History: Upper Klamath-Trinity rivers fall-run Chinook salmon show considerable variability in adult and juvenile life history strategies. This variability is characteristic of "ocean-type" Chinook salmon juveniles, which spend less than a year in fresh water before migrating to the ocean (see the Central Valley spring-run Chinook account for a more detailed discussion of ocean-type vs. stream-type life histories). Adult UKTR fall-run Chinook salmon enter the Klamath estuary from early July through September (Moyle 2002). They often hold in the estuary for a few weeks and initiate upstream migration as early as mid-July and as late as

October. Migration and spawning both occur under decreasing temperature regimes. Fall-run UKTR Chinook seem to hold extensively in, and travel slowly through, the lower Klamath River during their migration upstream (Strange 2005).

Between 1925 and the early 1960s, the Klamathon Racks provided a counting facility and egg collection station close to the current location of Iron Gate Dam. Chinook salmon historically passed this location in August between 1939 and 1958, with daily fish counts occurring during mid- and late-September and tapering off by late October (Shaw et al. 1997). More recently, peak migration appears to occur one to four weeks later than historical run timing (Shaw et al. 1997). In 2006, Chinook entered the Shasta River between mid-September and mid-December (Walsh and Hampton 2007) and Bogus Creek, adjacent to Iron Gate Hatchery, between mid-September and late-November (Hampton 2006). They reached spawning grounds in the Shasta and Scott rivers as early as September. Spawning in these tributaries tapers off in December, although snorkel surveys at the mouth of the Scott River found Chinook holding through mid-December (Shaw et al. 1997). Fall-run Chinook salmon migration occurs in the Trinity River between September and December, with early migrating fish entering larger tributaries first; spawning on smaller streams occurs later in the season. Spawning in the Trinity River begins earliest in suitable mainstem habitats immediately downstream of Lewiston Dam and extends into late November further downstream. Spawning in the South Fork Trinity River has been documented to begin in mid-October (LaFaunce 1967). In most other UKTR tributaries, spawning peaks during November before tapering off in December (Leidy and Leidy 1984).

Klamath River Chinook salmon have a lower fecundity and larger egg size than Chinook from the Sacramento River (McGregor 1922, 1923). The average fecundity of Lewiston Hatchery fish is 3,732 eggs for 4-kg fish (Bartholomew and Hendrikson 2006). Fry emerge from gravel in late winter or spring. The timing of fry emergence is dictated by water temperature, so the beginning of emergence may differ by over four weeks between years in the mainstem Trinity River (Shaw et al. 1997). The timing of juvenile emigration is highly variable and dependent on river rearing conditions, which are controlled largely by flows, water temperature and food availability. High winter flows, level of snowpack and subsequent spring runoff can reduce water temperatures (Minshall et al. 1989) and may contribute to annual variability in timing and duration of Chinook emigration. Once emigration begins, movement is fairly continuous, although high temperatures may cause emigrants to seek thermal refuges during the day and delay migration. Mean downstream movement rates for hatchery UKTR Chinook juveniles in the Klamath and Trinity rivers are 1.4 to 11.8 km per day (USFWS 2001).

While there is variability in age composition of fall-Chinook spawners returning to the Klamath basin, most fish are age-3, with a slightly smaller proportion of age-4 fish (CDFW 2016). Some age-5 individuals are also observed annually. Recently, CDFW samplers at Willow Creek Weir and in the Yurok/Hoopa Tribal fisheries have also confirmed a few rare age-6 fish (CDFW 2016). Age-2 fish (grilse) in some years can be a large proportion of the run; they are mostly male spawners that are much smaller than other spawners. From 1978 and 2006, grilse constituted 2-51% of all returning salmon to the Klamath Basin (CDFG 2006). In 1986, Sullivan et al. (1989) observed that a larger proportion of age-4 adults returned to the Salmon River (24%) than to other subbasins. During that year, the age structure of Chinook entering the estuary was composed of: two (23%), three (64%), four (12%), and five (1%) year old returns (Sullivan 1987). In 2004, the age structure of the Trinity River Hatchery (TRH) fall Chinook run was composed of: two (8%), three (78%), four (13%), and five (1%) year old fish (CDFG 2006a). In 2006, the Klamath River fall Chinook run was composed of: two (31%), three (21%), four (47%), and five (1%) year old individuals (KRTAT 2007) and these year class contributions have not changed much since (CDFW 2016).

Sullivan (1989) examined scales from returning fall-run adults to determine fry emigration patterns and identified three distinct juvenile freshwater life history strategies: (1) rapid emigration following emergence, (2) tributary or cool-water area rearing through the summer and fall emigration, and (3) longer freshwater rearing and overwintering before emigration. The first is the predominant strategy, where fry leave spawning areas quickly and forage in tributary and mainstem habitat for a short period before emigrating during summer months. In the spring-fed Shasta River, where water temperatures and flow are largely constant, peak fry outmigration occurs in March or early April, while in the snowmelt-fed Scott and Salmon Rivers, outmigration peaks from mid-April to mid-May. Historically, Chinook juvenile emigration initiated in mid-March in the mainstem Klamath River before peaking in mid-June (Shaw et al. 1997). More recently (1997-2000), wild juveniles were not observed in the lower river earlier than the beginning of June, with a peak in mid-July (USFWS 2001).

The second juvenile rearing strategy involves extended freshwater rearing with emigration to the ocean during fall to mid-winter (Sullivan 1989). Juveniles emigrate to the mainstem during spring and summer to rear or may remain in tributaries until fall rains and ocean entry. Multiple juvenile fish kills in July and August (1997, 2000) highlight the extensive use of the middle and lower Klamath River during summer months by juveniles (USFWS 2001). On the lower Trinity River (0.4 Rkm upstream of Weitchpec), naturally produced Chinook salmon emigration peak around mid-April. The first hatchery-produced Chinook salmon are not observed until six weeks later, and emigration of these fish peaks in mid-October on the lower Trinity River (Naman et al. 2004). The first two juvenile rearing strategies are likely influenced by mainstem flows. Wallace and Collins (1997) found Chinook salmon (probably from multiple ESUs) were more abundant in the Klamath River estuary in low flow years than during high flow years, suggesting that this strategy may involve moving into cooler and more productive estuarine water sooner than under high flow conditions.

Although the vast of majority of UKTR Chinook salmon use one of these strategies, a small portion of juveniles spend an entire year mainly in larger tributaries, entering the ocean the following spring as yearlings (Sullivan 1989). From 1997-2000, these hatchery-reared yearlings emigrated as smolts through the middle Klamath River between early May and June, before the peak of 0+ wild juveniles emigrated in mid-June (USFWS 2001). A fourth life history strategy has also been observed, where parr mature in the spring-fed Shasta River (C. Jeffres, UC Davis, pers. comm. 2011). Mature parr are reproductively mature males that have never left fresh water and are known from the Sacramento and other major spawning rivers (Johnson et al. 2012).

In the ocean, Klamath River Chinook salmon (all runs) are found in the California Current system off the California and Oregon coasts. Salmon seem to follow predictable ocean migration routes to feed (Harding 2015).

Habitat Requirements: Sexually mature UKTR fall-run Chinook salmon enter the Klamath estuary for only a short period prior to spawning. Unfavorable temperatures may exist in the Klamath estuary and lower river during summer, and chronic exposure of migrating adults to temperatures of 17°-20°C is detrimental to their survival and spawning success (McCullough 1999). However, if water temperatures are decreasing, UKTR fall-run Chinook will migrate upstream in water temperatures as high as 23.5°C; water temperatures above 21°C generally seem to inhibit migration when temperatures are rising (Strange 2005). The thermal threshold for migration inhibition seems to be higher for UKTR fall-run Chinook than for Columbia River fall-run Chinook (> 21°C), due to adaptations to higher temperatures (McCollough 1999).

Optimal spawning temperatures for Chinook salmon are reported as less than 13°C (McCollough 1999). Water temperatures in the fall are usually within this range in the Trinity River (Quilhillalt 1999), though Magneson (2006) reported water temperatures up to 14.5°C during spawner surveys in 2005. With respect to water temperature, the spring-fed Shasta River was historically the most reliable spawning tributary in the Klamath River system (Snyder 1923), but agricultural diversions and warm irrigation return water have greatly diminished its capacity to support salmon. In addition, Ricker (1997) found that levels of fine sediment in 6 of 7 potential Shasta River and Park Creek spawning locations were high enough to significantly reduce fry emergence rates and embryo survival.

Most UKTR fall-run Chinook spawning habitat is found in the mainstem Klamath and Trinity Rivers and their larger tributaries. Spawning occurs primarily over large cobbles, loosely embedded in gravel, with sufficient subsurface infiltration of water to provide oxygen for developing embryos. In a survey of Trinity River redds, Evenson (2001) found embryo burial depths averaged 22.5-30 cm, suggesting minimum depths needed for spawning gravels. Regardless of depth, the keys to successful spawning are adequate flow and cold temperatures. For maximum embryo survival, water temperatures must be between 6-12°C, with oxygen levels close to saturation (Myrick and Cech Jr. 2004). Fry emergence in the Scott and Shasta rivers begin at water temperatures near 8°C (Bartholomew and Hendrikson 2006). With optimal conditions, embryos hatch after 40-60 days, and remain in gravel as alevins for another 4-6 weeks until the yolk sac is fully absorbed.

On the mainstem Klamath, McCollough (1999) suggests water temperatures above 15°C stimulate juvenile emigration, although temperatures above 15.6°C can increase risk of disease. For example, warmer temperatures favor *Ichthyophthirius multifiliis*, or "ich" disease, and transmission of the bacteria *Columnaris*, which is associated with higher mortality of pre-spawn salmonids that are exposed to above-optimal water temperatures in Northern California (Strange 2007, Power et al. 2015). Daily average temperatures above 17°C increase predation risks and impair smoltification, while temperatures over 19.6°C decrease growth rates and may impact behavior (Marine and Cech Jr. 2004). Temperatures up to 25°C are common in the middle Klamath River during the spring/summer juvenile emigration period, so cool water inputs at tributary confluences, such as Blue Creek, serve as important refuge habitats (Belchik 1997). Stratified pools, springs, and subsurface flows at the base of old landslides and gravel bars are also important thermal refuges (R. Quinones, USFS, unpubl. obs.). Elevated river temperatures (>16°C) increase mortality from *Ceratomyxa shasta* infection in Chinook salmon released from Iron Gate Hatchery, as well as lead to lethargic behavior, reduced body mass, and co-occurring bacterial infections from *Parvicapsula minibicornis*. Belchik (1997) identified 32 cool water refuge areas in the middle Klamath River mainstem; twenty-eight of these locations were tributary confluences, including the Scott River. These areas, although small in size, have temperatures of 10°-21.5°C and provide refuges from temperatures lethal to emigrating juveniles (Belchik 1997).

In the ocean, Chinook from California are caught in trawls in predictable currents off the Klamath-Trinidad region of California and Southern Oregon in water temperatures between 8° and 12°C (Hinke et al. 2005), which is suitable for their euphasiid and copepod prey (Harding 2015). In surveys, Chinook salmon and are most frequently encountered closer to shore than steelhead, and follow a latitudinal gradient: juveniles are mostly captured in the Gulf of the

Farallones, while adults are more commonly captured off of Northern California and Oregon (Harding 2015). Chinook salmon from the Klamath and Trinity hatcheries have been observed in August south of Cape Blanco, suggesting some regional differences in salmon distribution at sea (Brodeur et al. 2004). Recent research using coded wire tags on hatchery fish, has discovered that spring-run Chinook salmon smolts migrate faster and further north than fall-run Chinook salmon at sea, and older fish often range farther from their natal streams than younger fish (Tucker et al. 2011).

Distribution: UKTR Chinook salmon are found in all major tributaries above the confluence of the Klamath and Trinity rivers and are raised in Iron Gate Hatchery (Klamath) and Trinity River Hatchery. UKTR fall-run Chinook salmon historically spawned in middle Klamath tributaries (Jenny, Shovel, and Fall creeks) and in wetter years possibly into rivers in the upper Klamath Basin (Hamilton et al. 2005). Access to these tributaries was blocked in 1917 by construction of Copco 1 Dam, then Copco 2 Dam, J.C. Boyle Dam, and Iron Gate Dam in 1964. As a result, salmon and other anadromous fishes were denied access to approximately 563 km of spawning, passage, and rearing habitat in the upper Klamath River basin (Huntington 2006). In the lower Klamath River, numerous tributaries provide suitable spawning habitat including: Bogus, Beaver, Grider, Thompson, Indian, Elk, Clear, Dillon, Wooley, Camp, Red Cap, and Bluff creeks. The Salmon, Shasta and Scott Rivers historically supported large numbers of spawning Chinook salmon, and remain the most important spawning areas when sufficient flows are present. In the mainstem Klamath River, spawning consistently occurs between Iron Gate Dam and Indian Creek, with the two areas of greatest spawning density typically occurring near Bogus Creek and Indian Creek (Magneson 2006).

UKTR fall-run Chinook salmon once ascended the Trinity River above the site of Lewiston Dam to spawn as far upstream as Ramshorn Creek. Lewiston Dam was completed in 1963, eliminating 56 km of spawning habitat in the mainstem (Moffett and Smith 1950). The Stuart Fork was an important historical spawning tributary upstream of Lewiston Dam, as were Browns and Rush creeks below the dam (Moffett and Smith 1950). Historically, the majority of UKTR fall-run Chinook spawning in the Trinity River occurred between the North Fork and Ramshorn Creek; spawning now primarily occurs above Cedar Flat and, to a lesser extent, in downstream tributaries and the mainstem Trinity River (W. Sinnen, CDFW, pers. comm. 2011). The distribution of redds in the Trinity River is highly variable. While the reaches closest to the Trinity Hatchery support substantial spawning, there is a high degree of variability in spawning habitat utilization in reaches between the North Fork and Cedar Flat (Quihiullalt 1999). The North Fork, New River, Canyon Creek, and Mill Creek also continue to support spawning in the basin. In the South Fork Trinity River, fall-run Chinook historically spawned in the lower 48 km up to Hyampom, and in the lower 4 km of Hayfork Creek (LaFaunce 1967).

Trends in Abundance: It is likely that the spring-run Chinook run was historically the most abundant in the UKTR Basin (Snyder 1931, LaFaunce 1967). However, by the time records were kept, spring run had been reduced to a minor component of Klamath salmon populations. Recent estimates of Chinook salmon in the Klamath-Trinity system are primarily from fall-run fish. There are several historical estimates of Chinook salmon abundance in the UKTR Basin: Snyder (1931) estimated 141,000 in 1912, Moffett and Smith (1950) estimated 200,000 at that time, and USFWS (1979) estimated approximately 300,000 to 400,000 fish from 1915-1928. At the Klamathon Racks, a fish counting station near Iron Gate Dam, an estimated annual average of

12,086 Chinook spawned in the upper basin from 1925-1949, and declined to an average of 3,000 from 1956-1969 (USFWS 1979). At this time, the Klamath River basin was believed to contribute 66% (168,000) of the total number of Chinook salmon spawning in California's coastal basins (CDFG 1965). This production was nearly equally distributed between the Klamath (88,000 fish) and Trinity (80,000 fish) basins, with approximately 30% of the Klamath basin's fish originating in the Shasta (20,000), Scott (8,000), and Salmon (10,000) rivers.

Hatchery operations have supplemented UKTR Chinook runs since the completion of Iron Gate Hatchery on the Klamath and Trinity Hatchery on the Trinity rivers in the 1960s. In the 1980s, Klamath Basin Chinook accounted for up to 30% of the commercial Chinook salmon landings in northern California and southern Oregon, which averaged about 450,000 salmon per year (PFMC 1988). From 1978 through 2006, the average in-river escapement of UKTR Chinook was 112,317 fish. By 2015, the average number of returning adults basin-wide each year increased to approximately 129,000 (Figure 1).

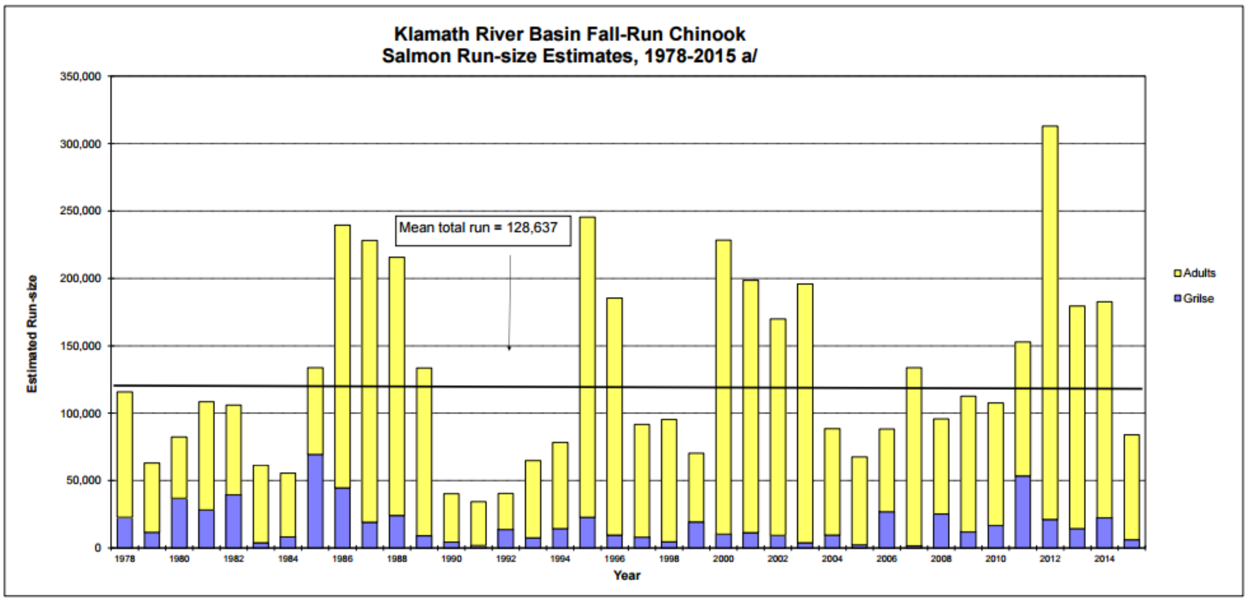


Figure 1. UKTR fall-run Chinook escapement, 1978-2015. From CDFW 2016, Fig. 1, pg. 14.

However, these totals have a high proportion of returning fish of hatchery origin. Since 1978, natural escapement has only surpassed 100,000 adults in four of 38 years, though there is an upward trend in natural escapement from 2004 to 2014 (Figure 2).

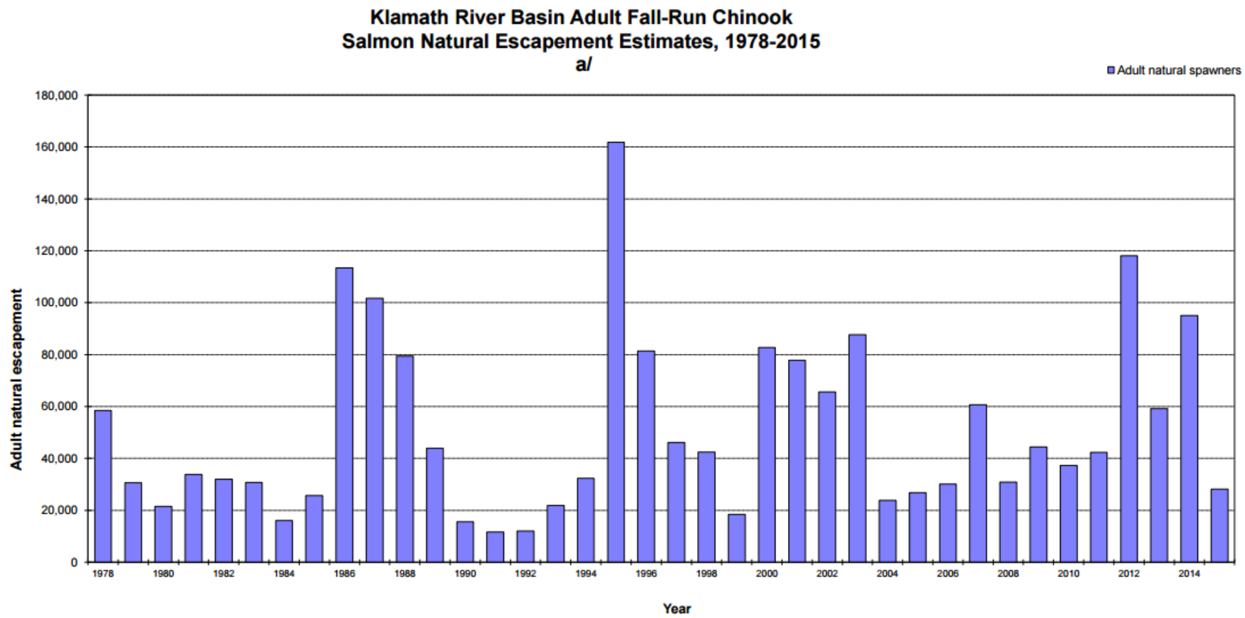


Figure 2. Natural UKTR fall-run Chinook escapement, 1978-2015. From CDFW 2016, Fig. 2, pg. 14.

Between 7 and 12 million juvenile Chinook are released annually from both hatcheries combined (NRC 2004). Between 1997 and 2000, about 60% of juveniles captured at Big Bar were hatchery-origin fish (USFWS 2001). Hatchery-origin adults also spawn in all major Klamath tributaries (e.g., Shasta, Scott, Salmon rivers), although straying of hatchery fish is pronounced closest to the hatcheries (e.g., Bogus Creek and Shasta River, Upper Mainstem Trinity River). On the mainstem Klamath from Iron Gate Hatchery to the Shasta River confluence, the contribution of hatchery fish to the total run in each year class has fluctuated widely since 2001 (Figure 3).

Year	Estimated hatchery-origin proportion	Escapement estimate	
		Total	Hatchery only
2001	11.8%	7,828	925
2002	14.2%	14,394	2,043
2003	3.8%	12,958	489
2004	1.2%	4,715	58
2005	26.6%	4,585	1,222
2006	22.7%	3,587	815
2007	39.8%	5,523	2,201
2008	37.0%	4,894	1,810
2009	25.1%	4,427	1,112
2010	48.1%	2,572	1,238
2011	40.9%	4,880	1,995
2012	45.3%	12,626	5,726

Figure 3. Mainstem Klamath spawner escapement estimates, Iron Gate Dam to Shasta River confluence. From Gough and Som 2015, pg. 22, Table 8.

Snyder (1931) noted that the Shasta River was historically the best spawning tributary in the basin. Adult escapement into the Shasta River was estimated to be 6,032 fish from 1978-1995, and 4,889 fish from 1995-2006 (CDFG 2006b). The Shasta River (~23%), mainstem Klamath River from Iron Gate Dam to Shasta River confluence (~21%), Bogus Creek (16%), and Scott River (~15%) accounted for the most fish, respectively, in the 2014 year-class in the Klamath Basin, with age-4 adults making up the majority of the run (Figure 4).

	AGE				Total	Total
Escapement & Harvest	2	3	4	5	Adults	Run
Hatchery Spawners						
Iron Gate Hatchery (IGH)	1,039	12,864	11,276	160	24,300	25,339
Trinity River Hatchery (TRH)	221	3,653	3,271	51	6,975	7,196
Hatchery Spawner subtotal	**1,260**	**16,517**	**14,547**	**211**	**31,275**	**32,535**
Natural Spawners						
Salmon River Basin	527	865	1,674	167	2,706	3,233
Scott River Basin	2,051	2,977	7,159	283	10,419	12,470
Shasta River Basin	3,945	4,064	10,265	83	14,412	18,357
Bogus Creek Basin	323	6,119	6,448	40	12,607	12,930
Klamath River mainstem (IGH to Shasta R)	1269	6491	8847	114	15,451	16,720
Klamath River mainstem (Shasta R to Indian Cr)	575	2932	4010	50	6,992	7,567
Klamath Tributaries (above Trinity River)	1,498	1,649	4,987	241	6,877	8,375
Blue Creek	332	105	1,108	32	1,245	1,577
Klamath Basin subtotal	10,520	25,202	44,498	1,010	70,709	81,229
Trinity River (mainstem above WCW)	6,576	10,261	12,011	1,004	23,276	29,852
Trinity River (mainstem below WCW)	74	115	135	11	262	336
Trinity Tributaries (above Reservation; below WCW)	47	123	361	31	515	562
Hoopa Reservation tributaries	52	135	398	34	568	620
Trinity Basin subtotal	6,749	10,634	12,905	1,080	24,621	31,370
Natural Spawners subtotal	**17,269**	**35,836**	**57,403**	**2,091**	**95,330**	**112,599**
Total Spawner Escapement	**18,529**	**52,353**	**71,950**	**2,302**	**126,605**	**145,134**

Figure 4. UKTR fall-run Chinook escapement, 2014. From KRRT 2015, Table 5, pg. 10.

Hallock et al. (1970) estimated that 40,000 fall-run Chinook salmon historically entered the Trinity River above the South Fork. Burton et al. (1977 in USFWS 1979) estimated 30,500 Chinook returned below Lewiston Dam on the Trinity River between 1968 and 1972. The average fall Chinook run for the Trinity River between 1978 and 1995 was 34,512; this average declined between 1996 and 2006 to 23,463 fish (CDFG 2007). Recently, escapement goals for the Trinity River Hatchery for fall-run Chinook were set at 62,000 total fish annually (Kier and Hileman 2016). In 2016, an estimated 10,365 adult fall-run Chinook made it past Willow Creek weir, representing less than a quarter of the annual average run size from 1977-2015. Jacks represented about 27% and 7% of all fall-run Chinook at Willow Creek weir and Trinity River Hatchery, respectively (Kier and Hileman 2016). With the exception of 2014 (Figure 5), the percentage of hatchery fish contributions to the total run upstream of the Willow Creek weir have decreased below the 25-year average of about 50% since 2007.

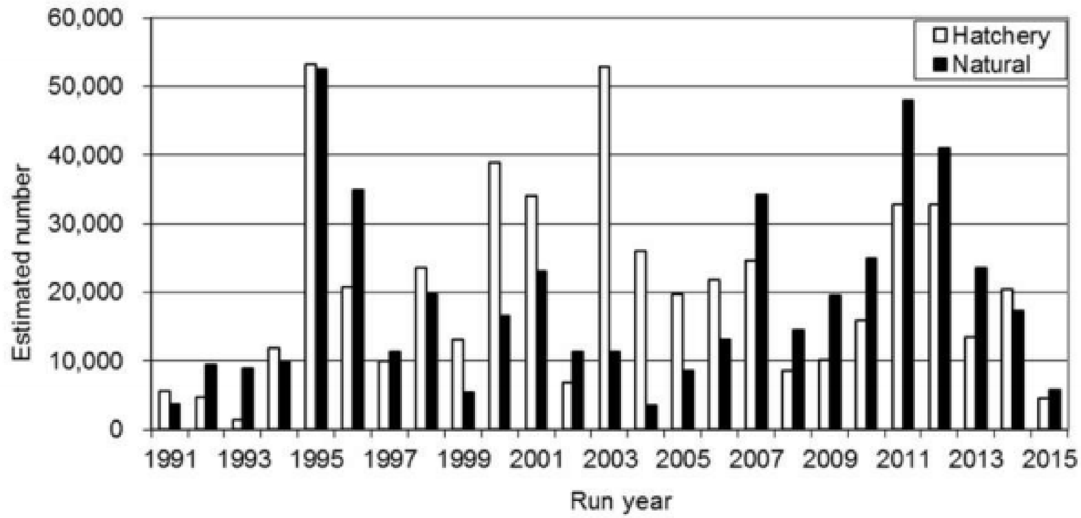


Figure 5. Natural and hatchery-origin fall-run Chinook salmon upstream of Willow Creek Weir on the Trinity River, CA, 1991-2015. From Kier and Hileman 2016 Fig. 13, pg. 33.

Each year, the Pacific Fisheries Management Council works with management partners to make annual run size predictions to determine harvest limits in the ocean, recreational, and tribal fishery sectors. These forecasts are based on hatchery releases from Iron Gate and Trinity, prior escapement estimates, scale sampling, length-at-age data, past harvest, and anticipated survival in the ocean. However, ocean conditions are extremely variable, and tend to shift on the order of decades in relation to Pacific Decadal Oscillation (PDO), which can significantly impact marine survival of salmonids (Mantua 2015).

In fact, poor marine conditions off the coast of California in 2005 and 2006 led to high mortality of juvenile Chinook salmon entering the ocean, and drove the collapse and subsequent closure of the commercial salmon fishery in California and southern Oregon in 2008-2009 (Lindley et al. 2009). In 2015, nearly 120,000 fall-run Chinook were forecast to return to the Klamath Basin, but only an estimated 78,000 (65%) returned, highlighting the significant challenges inherent in predicting return runs during ongoing drought and poor ocean conditions. The majority of these fish returned to Iron Gate Hatchery (19%), mainstem Trinity River above Willow Creek weir (16%), and the Shasta River basin (16%), respectively (CDFW 2016).

The proportion of natural-origin fall-run Chinook returning to the Trinity River has fluctuated while hatchery releases have mostly been constant since 2001. This shift has presumably come in response to Record of Decision environmental flow block adjustments and significant rearing habitat restoration in the mainstem Trinity River, which began in 2005 (Kier 2016). The proportion of natural-origin fall-run Chinook has surpassed 50% in 6 of 8 years since 2007, which had previously only happened once (2002) in the previous 6 years (Kier 2016). More monitoring over time will help elucidate the impacts of habitat and natural flow improvements over the last decade on numbers of returning adults to the Trinity Basin.

Factors Affecting Status: Numerous threats continue to stress UKTR fall-run Chinook salmon. Primary stressors include dams, logging and other land uses, fisheries, hatcheries, and disease.

Dams. UKTR fall-run Chinook are primarily mainstem spawners, so Lewiston and Iron Gate dams negatively affect populations by prohibiting access to upstream habitats and altering seasonal flows and temperature regimes in remaining downstream habitat. Iron Gate, Copco 1 and 2, and J.C. Boyle dams are used mainly for hydropower production and minimally impact total flows downstream in themselves, although agricultural water diversions in the upper Klamath basin do measurably reduce the amount of instream flow. However, dams have eliminated spawning gravel recruitment from upstream areas, reduced hydrologic variability, and significantly shifted peak flows in the basin (Hamilton et al. 2011). On the Shasta River, Dwinnell Dam blocks migrating salmon. The lack of adequate flow releases from Iron Gate and Lewiston dams, and resulting poor water quality and high water temperatures, are principal factors that caused a major kill of adult salmon in the lower Klamath River in September 2002 (CDFG 2004). Poor water quality in the lower Klamath is a major limiting factor on juvenile survival to outmigration, especially during drought years, as high water temperatures and low flows exacerbate stressors on fish and cause increased susceptibility to disease and associated mortality (USFWS 2015).

Lewiston Dam on the Trinity River has substantially modified river flows and generally reduced the size and habitat complexity of the river channel (A. Hill, CDFW, pers. comm. 2016). Starting in 1964, 75-90% of Trinity River flow was diverted to the Central Valley. Declines of naturally-spawning fall-run Chinook populations were exacerbated by diversion of most of the river's water and corresponding reduction in spawning gravels and degradation of habitat. In 1984, Congress ordered restoration of the river to support salmon at historical levels (see http://www.trrp.net/). In 2000, a Record of Decision (ROD) was signed that called for numerous restoration actions, as well as a rough doubling of flows of the river mimicking the natural flow regime. Implementation is now underway: block water releases have been mandated and implemented, and large-scale restoration of spawning and rearing habitat has occurred on the mainstem Trinity River (Kier 2016).

Agriculture. Much of the water diverted from the Trinity River is used for agriculture in the Central Valley. Diversion of water for agriculture from the headwaters of the Klamath River in Oregon, as well as from the Shasta and Scott rivers in California, reduces streamflows and increases water temperatures, reducing suitable habitat for spawning or rearing. Many farms use flood irrigation, whereby return water flows back into the streams at high temperatures, carrying nutrients and fertilizers, further warming streams and degrading water quality. These impacts are particularly acute during summer and early fall months, when ambient temperatures are highest and natural flow inputs are lowest. Pumping from wells adjacent to waterways also reduces groundwater tables and associated coldwater inputs into rivers, which are especially critical for Chinook salmon during the summer and fall months. The Shasta River, for example, has been converted by agricultural diversions from a cold river that supported year-round salmon production to one with degraded water quality, including temperatures too high to support salmon in summer. With the passage of Proposition 64 in California in November 2016, marijuana cultivation for private use has become legalized, which may put increasing pressure on limited water resources within the UKTR Basin. It remains to be seen if this legalization will impact the geography of grow operations in the future, which are currently centered in the UKTR basin and surrounding counties. More effort must be placed on understanding, quantifying, and reducing the extent and magnitude of these impacts on fall-run Chinook. For a full discussion of impacts of agriculture and marijuana cultivation on fishes in the watershed, see the Klamath Mountains Province winter steelhead account.

Logging. The majority of spawning and rearing habitat for UKTR Chinook salmon is surrounded by public lands in the Klamath and Shasta-Trinity National Forests, which have been heavily logged and covered in associated networks of roads. As a result, the Klamath River is regarded as impaired under the Clean Water Act because of high nutrient and sediment loads, high temperatures, and low levels of dissolved oxygen. See the UKTR spring-run Chinook account for further discussion on impacts from logging and other land uses in the UKTR Basin.

Grazing. Livestock grazing is widespread on public and private lands throughout the Klamath-Trinity system. Grazing impacts occur mainly on tributary streams, where livestock can cause severe bank damage and reduce riparian vegetation through trampling, compaction, and grazing, resulting in stream incision and silting of spawning gravels. After decades of ranching, feral cattle continue to degrade riparian vegetation and trample streambanks in the Trinity and Klamath basins (Beesley and Fiori 2008).

Rural/residential development. The long history of mining and logging in the Klamath and Trinity basins has left an extensive network of roads, which continue to provide access to many remote areas, facilitating rural development. Rural development results in increased sediment delivery to streams in the steep, mountainous terrain of this region, effluent from septic tanks and other pollutants, water diversion, deforestation and habitat fragmentation. These impacts are often most acute on small headwater streams; while not critical spawning habitats for fall-run Chinook, these are important sources of cold water and flows for all salmonids.

Mining. Mining has dramatically altered river habitats in the UKTR Basin, with lasting legacy impacts in many inland areas. Intensive hydraulic and dredge mining occurred in the 19th century and, depending on location, these activities caused severe stream degradation and alteration to channel morphology. Mining was a principal cause of decline of UKTR Chinook in the Scott River and large areas in the Trinity River, followed by some level of recovery after large-scale mining ceased. The Scott River Valley remains heavily altered, with immense piles of dredge tailings marking its history. Historical mining impacts still affect the Salmon River Chinook population, as the estimated 16 million cubic yards of sediment disturbed between 1870 and 1950 are slowly transported through the basin. Mining and its legacy effects have disconnected and constricted juvenile salmon habitat, filled in adult holding habitat, degraded spawning grounds, and altered the annual hydrograph of numerous streams. Pool in filling is a particular problem because high stream temperatures have been demonstrated to reduce survival of both holding adults and rearing juveniles (West 1991, Elder et al. 2002). In general, the productive capacity of the Scott River has been significantly reduced as a result of extensive mining in the watershed (Cramer Fish Sciences 2010).

While banned since 2009, suction dredging for gold has likely negatively affected fall-run UKTR populations through disturbance of spawning gravel, siltation, and sedimentation. See UKTR spring-run Chinook account in this report for more details.

Transportation. Roads are widespread along many UKTR streams, and increase sediment and pollutant inputs into waterways due to erosion. Many timber and mining roads were built at a time when little attention was paid to environmental impacts. Many roads have been improved and/or closed to public access, but impacts to stream habitats and water quality remain. Culverts and other passage structures often create migration barriers, although restoration projects have mitigated many of these impediments.

Fire. Wildfires are predicted to become more frequent and severe under climate change scenarios, so may pose increasing threats to spawning and holding habitats, as well as contribute to increasing water temperatures and sediment input that can fill pools, smother spawning habitat, and degrade water quality during rainstorms following fires. The UKTR area is prone to large wildfires as a result of decades of past fire exclusion practices.

Recreation. Water sports have a presumably minimal impact on UKTR juveniles and adults; however, widespread use of motorized boats in the lower Klamath River may affect adult spawner behavior and movement patterns. See the UKTR spring-run Chinook account in this report for more detail on potential recreational impacts.

Harvest. Unlike most other salmon runs in California, the UKTR fall-run Chinook run supports substantial commercial, tribal, and recreational fisheries. Harvest may be selecting for smaller and faster-maturing fish over time, as larger and older fish are removed from the population year after year. Snyder (1931) noted a decline in the proportion of age 4 and 5 Chinook in the estuary, which was most likely the result of harvest focused on larger fish. The Pacific Fisheries Management Council (PFMC), which regulates commercial fishing on the West Coast, has paid particular attention to UKTR Chinook returns because these fish account for a considerable proportion of the harvest of salmon in California and southern Oregon each year. The breakdown of in-river harvest and total escapement, those fish that were not caught in commercial fisheries at sea for the 2014 year-class, is shown below (Figure 6).

	Age 2	Age 3	Age 4	Age 5	Total Adults	Total Run
Recreational Harvest						
Klamath River (below Hwy 101 bridge)	268	249	775	69	1,093	1,361
Klamath River (Hwy 101 to Weitchpec)	2,847	365	1,438	71	1,875	4,722
Klamath River (Weitchpec to IGH)	75	728	759	9	1,496	1,571
Trinity River Basin (above WCW)	168	358	355	45	758	926
Trinity River Basin (below WCW)	3	26	26	3	55	58
Subtotals	**3,361**	**1,726**	**3,353**	**198**	**5,277**	**8,638**
Tribal Harvest						
Klamath River (below Hwy 101)	153	2,262	16,668	1,108	20,039	20,192
Klamath River (Hwy 101 to Trinity mouth)	130	593	2,785	56	3,434	3,564
Trinity River (Hoopa Reservation)	65	524	1,804	111	2,439	2,504
Subtotals	**348**	**3,379**	**21,257**	**1,277**	**25,913**	**26,260**
Total Harvest	**3,709**	**5,105**	**24,610**	**1,475**	**31,190**	**34,898**
Totals						
Harvest and Escapement	22238	57458	96560	3777	157,794	180032
Recreational Angling Dropoff Mortality 2.04%	69	35	68	5	108	177
Tribal Net Dropoff Mortality 8.7%	30	294	1,848	111	2,253	2,283
Klamath River disease testing	11	50	234	4	288	299
Total River Run	**22,348**	**57,837**	**98,710**	**3,897**	**160,444**	**182,792**

Figure 6. Recreational and Tribal harvest estimates of UKTR fall-run Chinook, 2014. From KRTT 2015, Table 5, pg. 10.

Poor ocean conditions can severely impact adult escapement, especially when combined with high rates of harvest that this run has historically sustained. Harvest goals are often difficult to set because poor ocean conditions can devastate several year classes of fish at once. The difficulties are compounded by consistently poor conditions in fresh water and reliance on hatchery fish to support the fishery. This leads to extreme population fluctuations, resulting in complete commercial salmon fishery closures in California for the 2008 and 2009 fishing seasons (Lindley et al. 2009, CDFW 2010). In 2016, the Council cited a substantially low forecast of fall-run Chinook returns to the UKTR and Central Valley, primarily due to ongoing drought and poor ocean conditions associated with El Nino, as the driver for fishery constraints in Oregon and California (PFMC 2016). Fisheries for UKTR fall-run Chinook salmon are likely to be periodically restricted to prevent overharvest of wild fish, unless a mark-selective fishery is instituted [e.g., all hatchery fish are marked and all non-marked (wild) fish are released].

Hatcheries. Although most tributary spawning stocks are apparently comprised mainly of wild fish, the mainstem Klamath River is increasingly supported by hatchery returns. Hatchery operations have likely influenced the age of maturation and spawning distribution of UKTR Chinook salmon and reduced life history diversity in the Klamath-Trinity basin. Hatcheries first began operating on the Klamath River for fall-run Chinook in 1914. A significant proportion of mainstem spawning now occurs between Shasta River and Iron Gate Dam. In 1999, 73% of redds were located between Iron Gate Hatchery and the Shasta River and this proportion has increased over time (Bartholomew and Hendrikson 2006).

Historically, most fall-run Chinook in the Trinity River spawned between the North Fork and Ramshorn Creek (Moyle et al. 2008). More than 50% of out-migrating smolts observed between 1999 and 2000 at the Willow Creek weir were of hatchery origin; this proportion increased to more than two-thirds during the fall monitoring period (USFWS 2001). While the contribution of hatchery-origin fish to the total annual run is trending downward over the last decade (Kier 2016), they still have impacts on natural-origin fish. Most naturally produced Chinook in the basin are ocean-type and emigrate in the spring and summer, while Trinity River Hatchery releases yearlings in October. Large numbers of hatchery fish in the Klamath-Trinity system may impact naturally produced Chinook through competition, hybridization, predation, redd superimposition, and/or disease transmission. Wild populations face reduced fitness through interbreeding with hatchery fish (Araki et al. 2007, 2009, Kinziger et al. 2008).

Competition and predation may be enhanced when releases of large (compared to wild fish) hatchery juveniles occupy shallow water refuge habitats used by naturally spawned juveniles (NRC 2004). Naman and Sharpe (2012) conducted a review of predation impacts of hatchery-origin steelhead on wild juvenile salmonids in the Trinity Basin, and found predation rates were orders of magnitude higher than found in other watersheds throughout the Pacific Northwest. Over 6% of natural-origin Chinook and coho subyearlings in the year class were consumed by hatchery-origin juvenile steelhead in the study reach below Trinity River Hatchery. Hatchery steelhead are released on March 15 every year near thousands of redds, from which fry have recently emerged and before they have emigrated, creating conditions that favor high predation rates.

Hatchery returns are likely replacing natural escapement of at least some wild populations of UKTR fall-run Chinook. The proportion of fall-run Chinook natural escapement has significantly decreased over time, concurrent with significant increases in hatchery returns to IGH and TRH since the 1980s (Quiñones et al. 2013). These patterns suggest increasing dependence on hatchery propagation and perhaps signal similar responses of natural and hatchery spawners to environmental conditions. More research is needed to understand the full extent of hatchery influence on natural production and genetics of fall-run Chinook in the Klamath-Trinity system, especially with removal of the four lowermost dams on the Klamath River and significant restoration and reintroduction efforts looming in 2020. See the Central Valley fall-run Chinook account for further discussion on hatchery effects on wild stocks.

The large-scale die-off of over 60,000 UKTR salmon and other fish in the Klamath River in 2002, and near-repeat again in 2014, provide examples of how multiple factors can create synergistic impacts that are capable of decimating salmon runs (Lynch and Riley 2003). Chinook salmon in the Klamath and Trinity basins migrate when water temperatures and minimum flows begin to approach their limits of tolerance, increasing their susceptibility to stress and disease. In

September 2002, between 30,000 and 70,000 predominantly UKTR fall-run Chinook adult salmon perished in the lower Klamath River due to infection by "ich" disease (caused by *Ichthyopthirus multifilis*) and columnaris disease (*Flavobacter columnare*) (Lynch and Riley 2003). Factors that led to this massive die-off are still not fully understood, but were likely a combination of: (1) high water temperatures, (2) crowded conditions, and (3) low flows. These conditions allowed for a disease epidemic to sweep through the population of highly stressed fish concentrated in pools. Increased base flows likely reduce pathogen transmission risk during Chinook salmon migration (Strange 2007). High water temperatures and low flows can also increase salmonid susceptibility to a number of other diseases, such as *Ceratomyxa shasta* and *Parvicapsula minibicornis*. It is likely that UKTR fall-run Chinook were historically infected by these diseases at low levels, but widespread epidemics rarely occurred because contributing factors such as high temperatures, low flows, and poor water quality throughout the lower Klamath River during summer months did not exist to the extent they do now. When high densities of infected fish and warm temperatures exist in combination, *C. shasta* infection is accelerated (Foott et al. 2003). Large releases of hatchery fish may therefore be particularly susceptible to infection and spread disease to wild fish. It is also likely that most juvenile Chinook from the Scott and Shasta rivers do not survive their exposure during emigration through the lower Klamath, and these diseases may exert selection pressures by killing juvenile UKTR Chinook that emigrate at times when temperatures in the main river are too warm.

While many threats act in concert to reduce abundance and resiliency in salmon populations in California, perhaps none are more important or less understood than the forces that dictate early marine survival of salmon. Population productivity and persistence are often determined during this critical early marine life stage, and low abundance increases susceptibility of salmon to environmental forcing, such as Pacific Decadal Oscillation and El Nino events (Kilduff et al. 2014).

Alien species. Alien species are at best a minor problem to UKTR Chinook salmon. Predation by brown trout (*Salmo trutta*) on juvenile Chinook, coho, or steelhead juveniles in the system appears to have negligible effects compared to other predation. Brown trout may also compete with other native salmonids at all life stages for food, rearing and spawning habitat (NMFS 2014), but effects are likely to be small and localized.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of UKTR fall-run Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Much former habitat is above dams; dams have significantly reduced remaining habitat quality downstream.
Agriculture	Medium	Habitats have been degraded through diversions, warm return water, and associated pollutant inputs.
Grazing	Medium	Livestock are pervasive on public and private lands; impacts concentrated in smaller tributary streams.
Rural/ residential development	Low	Cumulative effects of numerous roads and rural development can negatively affect salmon.
Urbanization	Low	Urban areas are few, small, and restricted to main rivers.
Instream mining	Medium	Legacy effects are still severe in some areas and have altered stream productivity, such as on the Scott and Salmon Rivers.
Mining	Low	Legacy effects of hard-rock mining are potentially severe in localized areas, such as the Salmon and Trinity rivers.
Transportation	Medium	Roads present along many streams and contribute sediment and pollutants along with habitat fragmentation.
Logging	Medium	Both legacy effects and ongoing impacts degrade aquatic habitats, but impacts are much lesser than historical impact.
Fire	Medium	Fires predicted to become more frequent and severe across the Basin, potentially degrading important headwater habitats.
Estuary alteration	Low	The Klamath River estuary is less altered than most North Coast estuaries.
Recreation	Low	Human uses such as boating may impact behavior of spawning fish and juveniles, especially in the lower Klamath River.
Harvest	Medium	Legal and illegal harvest may be negatively affecting abundance.
Hatcheries	Medium	Iron Gate and Trinity hatcheries contribute significantly to returning adults and may impact genetic integrity of wild fish.
Alien species	Low	Brown trout are present in the Trinity River, are known competitors and predators of juvenile salmon.

Effects of Climate Change: Moyle et al. (2013) rated the UKTR fall-run Chinook as "highly vulnerable" to extinction in the next 100 years as the result of the added impacts from climate change. The 'ocean' life history strategy of UKTR fall-run Chinook makes them least vulnerable of all runs to climate change. However, warm temperatures in the lower Klamath River during summer and fall months are already a substantial threat to migrating fish, and are likely to exacerbate problems with disease outbreaks and die-offs, such as those associated with "ich" disease and *C. shasta*. Elevated water temperatures have been identified as a factor limiting anadromous salmonid abundance in the Klamath River basin, especially UKTR spring-run Chinook salmon, as the result of the impacts of multiple land and water use impacts being compounded by climate change. Water temperatures in UKTR Basin rivers have increased approximately 0.5°C/decade, and have resulted in the loss of about 8.2 km of cool summer water in the mainstem each decade (Bartholomew and Hendrikson 2006). Bartholomew and Hendrikson (2006) documented that the timing of high temperatures potentially stressful to Chinook has moved forward seasonally by about one month in spring, and therefore extended the amount of time that stressful conditions exist in the lower Klamath. These temperature changes are consistent with measured basin-wide air temperature increases. The resulting spatial and temporal loss of rearing habitat may also reduce the survival of UKTR fall-run Chinook. See the UKTR spring-run Chinook account for further information on potential climate change impact to salmon populations in this region.

Marine survival plays an important role in abundance trends over time in Pacific salmonids, but the factors that contribute to changes in survival are only recently becoming clear. Researchers at NMFS' Southwest and Northwest Fisheries Science Centers have found that a complex interaction of physical and biological processes, including temperature, upwelling, lower-trophic level production, competition, and predation impact survival of juvenile salmonids at sea (Sharma et al. 2013). Increases in sea surface temperatures, such as those associated with climate change and El Nino Southern Oscillation (ENSO) events, reduce salmonid survival across populations, and play a significant role in predicting salmon abundance from the Klamath Basin to southeast Alaska. Generally, as sea surface temperatures increase, upwelling decreases, and the composition of prey offshore, such as euphasiid species that Chinook prey on, also shift. Based on historical analyses of changing sea surface temperatures and survival, Sharma and colleagues (2013) estimate that a 1°C increase in average sea surface temperatures in the spring and early summer, while juvenile salmon transition from freshwater to salt, could result in 1-4% reductions in survival across their range. Such increases are well within the expected bounds of sea surface temperature increases due to climate change by 2100 (Mantua 2015).

In addition to these predictable outcomes, greater forces such as the North Pacific Gyre Oscillation, which shifts on timeframes of decades, has recently been shown to be highly correlated with survival of salmonids from Alaska to California (Mantua 2015). This oscillation of cool and warm periods in the North Pacific, where most salmon migrate to feed, interact in complex ways with climate change that are not currently well understood. However, there is evidence that as hatchery-influenced runs from the southern edge of the salmon range in North America lose genetic diversity and life history variability, their survival in changing ocean environments has also become synchronized, resulting in greater population fluctuations (Hayes et al. 2016). This is in contrast to the more diverse, wild-origin runs at the northern edge of the range, which have generally enjoyed higher survival through poor growing conditions at sea over time (Mantua 2015). In short, greater diversity in run timing, life history adaptation, and genetic variability can help salmon runs reduce their risk of poor survival in the face of shifting ocean conditions and climate change. This bet-hedging survival tactic employed by anadromous salmonids through expression of various life history strategies so that entire year classes are not devastated by poor conditions at sea, known as the portfolio effect, is akin to how a diversified portfolio of investments can reduce catastrophic losses in the event of a stock market crash (Satterthwaite and Carlson 2011).

Status Score = 3.1 out of 5.0. Moderate Concern. UKTR fall-run Chinook are not in immediate danger of extinction, although their numbers have declined in recent decades. There is increasing reliance on hatcheries to maintain fisheries and returns of hatchery-origin fish may be masking a decline of wild production in the Klamath-Trinity basins. The UKTR Chinook salmon ESU was determined to not warrant listing under the Federal Endangered Species Act on March 9, 1998. However, they are considered a Sensitive Species by the U.S. Forest Service and a Fish Species of Special Concern by CDFW (CDFW 2015). CDFW manages the run for sport in fresh water, while the Pacific Fishery Management Council manages tribal, ocean sport and commercial fisheries.

Table 2. Metrics for determining the status of UKTR fall-run Chinook salmon, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	5	Widely distributed in Klamath and Trinity basins.
Estimated adult abundance	4	Abundant, with several large populations, but hatchery contributions a problem.
Intervention dependence	3	Major intervention is required to maintain fisheries, primarily through hatchery propagation and flow regulations.
Tolerance	3	Moderate physiological tolerance.
Genetic risk	3	Genetically diverse population but heavily influenced by hatcheries.
Climate change	2	Vulnerable to increasing temperatures in mainstem rivers, changes in flow regimes in tributaries, and variable ocean conditions.
Anthropogenic threats	2	1 "High" and 8 "Medium." A highly managed population.
Average	3.1	22/7. No immediate threat of extinction but declines likely.
Certainty (1-4)	4	Most studied of Klamath River Chinook runs.

Management Recommendations: While UKTR fall-run Chinook salmon seem to be decreasing in abundance over time, significant actions have been undertaken to aid their populations, perhaps slowing decline. The Trinity River Restoration Program (TRRP) is focused on maintaining and recovering populations of UKTR Chinook salmon by taking a holistic approach to restoration in the form of flow manipulations, focused restoration activities, and implementation of conservative fisheries management actions. In the Trinity Basin, annual monitoring and assessment of returning fish by CDFW and Hoopa Tribal Fisheries are working to adaptively manage Trinity River flows, as set out in the Trinity River Restoration Program's stated objectives (Kier and Hileman 2016). Restoration activities, such as those that deploy spawning gravel in known spawning reaches, are presumably decreasing competition for limited usable spawning habitat, reducing redd superimposition and interbreeding of fall-run and spring-run Chinook below Trinity River Hatchery (Kier and Hileman 2016). Where possible, spatial and temporal overlap between predators (e.g., hatchery-origin steelhead) and prey (subyearling Chinook and coho salmon) should be reduced by hatchery operations to reduce high rates of predation observed in the Trinity River. In addition, block water releases are being used for adult attraction flows and spring runoff events to aid in juvenile outmigration, especially with the prevalence of poor water quality conditions in the lower Klamath River. Overall, restoration objectives for the TRRP provide reasonable targets for ameliorating limiting factors and increasing suitable habitat quantity and quality in the Trinity River. They need to continue to be implemented and managed adaptively.

A similar program needs to be implemented as part of the Klamath River Restoration Program. Planned removal of the four lowermost Klamath Dams is scheduled to begin in 2020, marking the beginning of the largest river restoration project in North America. This will return the river to a more natural flow regime in part, although summer flows are likely to be lower due to upstream removals of water (in Oregon). While the Salmon River and some smaller watersheds in the Klamath National Forest remain in relatively good condition, the Shasta and Scott rivers need large-scale restoration efforts and improved flows to protect salmon populations and ensure successful re-colonization of newly accessible historical habitat after dam removal. For example, restoration of the highly productive Shasta River, with its constant spring-fed cold water from Big Springs Creek, may be prioritized in the short-term to boost natural-origin salmon populations that could volitionally re-colonize the upper Klamath River.

Protecting and restoring cool water habitats throughout the Klamath and Trinity watersheds will be essential to conserving fall- and spring-run Chinook in the Basin. A changing thermal regime in the Klamath River, which is exacerbated by ongoing drought and climate change, could rapidly eliminate UKTR Chinook spawning habitat in the mainstem and create thermal barriers to migration, effectively disconnecting critical spawning tributaries from the lower mainstem on the Klamath River. Both adult immigrants and juvenile emigrants are often exposed to water temperatures that are bioenergetically suboptimal or even lethal, leading to increased incidence of disease outbreaks.

The behavioral and life history plasticity displayed by Chinook salmon indicates strong potential for management strategies that increase juvenile survival through maintenance of multiple life history patterns. In mainstem habitats, Belchik (1997) demonstrated that UKTR Chinook use cool water areas as refuges; use of such habitats increases adult spawner and juvenile outmigrant survival and should be catalogued and monitored, at a minimum, and expanded where possible through adaptive water management and restoration. For example, the Western Rivers Conservancy and the Yurok Tribe have partnered to purchase the majority of the Blue Creek watershed from the Green Diamond Resource Company and to place it under Yurok Tribal stewardship in perpetuity as a salmon sanctuary. Innovative partnerships such as these should be expanded wherever appropriate in the UKTR. Partnerships with Tribes, private landowners, resource companies, municipalities, and others that conserve and restore cold water and flows to UKTR tributaries, such as the Shasta, Salmon, Scott, and South Fork Trinity Rivers will be critical to the recovery and persistence of robust runs of salmonids in the basin. Safe Harbor Agreements, where assurances are given from management agencies on Endangered Species Act and other environmental regulations to private landowners that willingly engage in conservation efforts on their properties, should be replicated and expanded where possible.

Many of the recommendations for conservation of UKTR spring-run Chinook also apply to fall-run Chinook (see UKTR spring-run Chinook account). In particular, hatchery operations at Iron Gate Hatchery and Trinity Hatchery are likely reducing productivity, survival, and persistence of wild salmon in the basin, and should be examined critically to find ways to reduce competition, predation, loss of genetic diversity, and loss of life history diversity (Tucker et al. 2011). In the short-term, managers should consider limiting harvest to a mark-selective fishery for 100% adipose fin-clipped fall-run Chinook to separate hatchery fish from wild fish and better understand the impacts of the fishery on natural-origin stocks. This could be done experimentally for 5-10 generations, as a learning tool. In addition, the benefits and risks associated with employing a new conservation hatchery on the upper mainstem Klamath River should be carefully considered to help re-establish populations of Chinook salmon in the upper watershed.

UPPER KLAMATH-TRINITY RIVERS SPRING-RUN CHINOOK SALMON

Oncorhynchus tshawytscha (Walbaum)

Critical Concern. Status Score = 1.6 out of 5.0. Small, self-sustaining populations occur in the Salmon and South Fork Trinity rivers, where they are highly vulnerable to climate change, introgression with hatchery fish, and other stressors.

Description: Upper Klamath-Trinity river (UKTR) spring-run Chinook salmon are nearly identical in appearance to fall-run Chinook salmon in the same basin. However, because these two run types differ in maturation, migration, and spawning, they have been reproductively isolated over time (Kinziger et al. 2008). As such, they are treated here in two separate accounts. Klamath River Chinook possess significant differences from Sacramento River Chinook in the number of gill rakers and pyloric caeca, with 12-13 rough, widely spaced gill rakers on the lower half of the first gill arch and 93-193 pyloric caeca (Snyder 1931, McGregor 1923). Dorsal fin ray, anal fin ray and branchiostegal counts are significantly different from Columbia River Chinook (Snyder 1931, Schreck et al. 1956). They have 10-14 major dorsal fin rays, 13-16 anal fin rays, 14-19 pectoral fins rays and 10-11 pelvic fin rays. Branchiostegal rays number 13-18 and there are 131-147 scales along the lateral line. For the 2015 run, mean fork length of all captured spring-run Chinook salmon at Junction City weir and Trinity River Hatchery was 66 cm (Kier and Hileman 2016).

Klamath River Chinook spawning adults are considered to be smaller, more rounded, and heavier in proportion to their length compared to Sacramento River Chinook (Snyder 1931). UKTR spring-run Chinook salmon enter natal streams during spring and early summer months as silvery, sexually immature adults that lack the breeding colors or elongated kype seen in fall-run Chinook salmon (Snyder 1931). They were historically likely the most abundant salmon run in the Klamath watershed and most important to native peoples, who esteemed them for their superior fat content and flavor (J. Saxon, Karuk Tribal Council, pers. comm. 2016).

Taxonomic Relationships: The UKTR Chinook salmon ESU includes all naturally spawned populations of Chinook salmon in the Klamath River basin, upstream from the confluence with the Trinity River (Waples et al. 2004). Within the UKTR Chinook ESU, genetic analyses have demonstrated that stock structure mirrors geographic distribution (Banks et al. 2000). Fall- and spring-run Chinook salmon from the same subbasin appear more closely related to one another than each is to fall or spring-run Chinook from adjacent basins (Pearse et al. 2015, Prince et al. 2016). Furthermore, fall- run Chinook salmon populations from the Klamath and Trinity subbasins appear more similar to the respective spring-run Chinook populations within a given subbasin than they are to fall-run Chinook in Lower Klamath River tributaries. Spring-run (stream-maturing ecotype) and fall-run (ocean-maturing ecotype) Chinook salmon have evolved repeatedly and independently in different geographic locations in a timeframe of perhaps less than about 1,000 years (Waples et al. 2004, Pearse et al. 2014). This pattern is distinct from Chinook in the Sacramento and Columbia rivers, where spring-run Chinook from different basins are more similar to one another than they are to fall-run Chinook within the same basin. It is hypothesized that the spring-run life history evolved first in California, and then radiated out and evolved more recently in more Northern populations (M. Miller, UC Davis, pers. comm. 2016).

UKTR spring-run Chinook are treated here as a distinct taxon because they represent a unique life history strategy that is supported by genetic variation on the Omy-5 locus of the genome, are an essential adaptive component of the ESU, and require separate management strategies for conservation than their fall-run counterparts in the same watersheds.

Life History: Similar to summer-run steelhead in the Klamath Mountains Province, UKTR spring-run Chinook salmon enter fresh water as immature fish, before their gonads are fully developed, and hold in cold water streams for 2-4 months before spawning. This life history strategy represents a relatively uncommon stream-maturing ecotype (Prince et al. 2015). Fish enter the Klamath estuary beginning in March and tapering off in July, with a peak between May and early June (Moffett and Smith 1950, Myers et al. 1998). A majority of late-entry fish are apparently of hatchery origin from the Trinity River Hatchery (TRH) (Barnhardt 1994, NRC 2004). Leidy and Leidy (1984) noted that adult Trinity River spring-run Chinook migration continued until October. However, given this late-entry timing, it is unclear if these fish are sexually mature and capable of spawning with spring-run Chinook adults already in the system. Because this late spring-run type is limited to the Trinity River, it is possible these fish represent hybrid spring and fall-run Chinook from hatchery stocks. Chinook salmon entering the Trinity River before October are considered to be spring-run Chinook, because the hatchery does not start processing adult fall-run Chinook until the day after Labor Day in most years (J. Hileman, CDFW, pers. comm. 2017). For harvest management of Chinook in the UKTR, September 1 is the date when regulations change to favor fall run fisheries (CDFW 2017). Moffett and Smith (1950) noted that spring-run Chinook migrated quickly through the watershed; more recent work (Strange 2005) has confirmed this rapid migration pattern. While migration occurs throughout the day and night, it peaks during the two hours following sunset (Moffett and Smith 1950).

Coded wire tags inserted into juvenile Chinook heads before release from Trinity River Hatchery reveal the age and run-timing composition of fall- and spring-run adults returning to the Trinity River each year for management purposes. Generally, the delineation in run-timing between fall- and spring-run fish falls between the last week of August and the first week of September, depending on flow conditions in a given year (J. Hileman, CDFW, pers. comm. 2017). This somewhat contradicts the management date of September 1 each year; returning adults are managed not necessarily on their run timing but out of convenience. Fish are counted and a portion are tagged as they pass the Junction City (Rkm 132.7) and Willow Creek weirs (Rkm 22.7) to give staff at the Trinity River Hatchery a benchmark to extrapolate run sizes and other important biological information (J. Hileman, CDFW, pers. comm. 2017).

The timing of spawning differs among watersheds throughout the UKTR basin. Spring-run Chinook generally enter the mouth of the Klamath River from late March to July in most years, and begin their migrations inland (W. Sinnen, CDFW, pers. comm. 2016). They reach the Trinity River in May-August and begin spawning in early September. In the mainstem Trinity River, spawning typically peaks 4-6 weeks earlier than that of fall-run UKTR Chinook in the same basin (Moffett and Smith 1950. However, in the upper reaches of the South Fork Trinity River, spawning peaks in mid-October, while in the Salmon River spawning begins in late September (LaFaunce 1967).

Overlap between fall- and spring-run Chinook spawning areas was historically minimal due to differences in temporal and spatial preferences and access to a wide variety of potential spawning habitat. In the South Fork Trinity River, the majority of spring-run Chinook salmon spawning occurred upstream of Hitchcock Creek, above Hyampom Valley, while fall-run

Chinook spawned below this point (LaFaunce 1967, Dean 1996). However, Moffett and Smith (1950) noted that spawning of fall and spring-runs overlapped in October on suitable spawning riffles between the East Fork Trinity River (now behind Lewiston Dam) and the North Fork Trinity River, which enters the mainstem about 64 Rkm downstream of Lewiston Dam. Dam creation and habitat degradation have decreased the amount of potential spawning habitat in the watershed over the last several decades.

Upper Klamath-Trinity rivers spring-run Chinook fry emerge from gravel from early winter (Leidy and Leidy 1984) until late-May (Olson 1996). With optimal conditions, embryos hatch after 40-60 days and remain in the gravel as alevins for another 4-6 weeks, usually until the yolk sac is fully absorbed. Before Lewiston and Trinity dams were completed in 1963 and became the upper limit of spawning in the Trinity River, emergence upstream of Lewiston began in early January; Moffett and Smith (1950) speculated that these early fish were offspring of UKTR spring-run Chinook. More recent reports (Leidy and Leidy 1984) suggest emergence begins as early as November in the Trinity River and December in the Klamath River and lasts until February.

Unlike most spring-run Chinook north of the Klamath River (e.g., Columbia River), UKTR spring-run Chinook do not consistently display "stream type" juvenile life histories, where juveniles spend at least one year in streams before migrating to the ocean (Olson 1996). Juvenile emigration occurs primarily from February through mid-June (Leidy and Leidy 1984). Natural-spawned juvenile Chinook salmon were not observed emigrating past Big Bar (Rkm 91) earlier than the beginning of June, with a peak in mid-July from 1997-2000 (USFWS 2001). In the Salmon River, two peaks of juvenile emigration have been observed: spring/early summer and fall. Snyder (1931) examined scales from 35 adult spring-run Chinook and 83% displayed juvenile "ocean type" growth patterns, in which juveniles entered the ocean just a few months after emerging from the gravel. Other scale studies have found that over two-thirds of sampled spring-run Chinook salmon from the South Fork Trinity River expressed an ocean-type juvenile life history (Dean 1994). In the Salmon River, an otolith study (Sartori 2005) identified 31% of fall-emigrating juvenile Chinook salmon as having similar growth patterns to Salmon River spring-run Chinook, suggesting these were also 'ocean-type' juveniles.

Habitat Requirements: UKTR spring-run Chinook enter the Klamath estuary when river water temperatures are at optimal holding temperatures, typically around March or April (10-16°C; McCullough 1999). Spring-run Chinook use thermal refuges in the estuarine salt wedge and associated nearshore ocean habitats prior to entering fresh water (Strange 2003). Temperatures in the Lower Klamath River typically rise above 20°C in June and can reach 25°C during August, leading to a small migratory window for most fish. Strange (2005) found adult migration changed with different temperature trajectories; however volume of flow was the most important driver of timing of spring-run Chinook salmon migration (Strange et al. 2010). Under favorable flow and temperature conditions, spring Chinook in the UKTR may migrate up to 3.7 km/day (J. Hileman, CDFW, pers. comm. 2017).

When daily water temperatures are increasing, Chinook will migrate upstream until temperatures reached 22°C, but when temperatures are decreasing, fish will continue to migrate upstream at water temperatures of up to 23.5°C (Strange 2005). A cool water refuge at the confluence of Blue Creek (Rkm 64), the largest tributary on the lower Klamath, was used by 38% of spring- run Chinook for more than 24 hours in 2005 (Strange 2005). Optimal adult holding habitat is characterized by pools or runs >1 m deep with cool summer temperatures

(<20°C), all-day riparian shade, little human disturbance, and underwater cover such as bedrock ledges, boulders or large woody debris (West 1991). These habitats are similar to those preferred by holding summer steelhead in the basin (Nakamoto 1994); the two species are often found together in similar habitats (L. Cyr, USFS, pers. comm. 2016). Because the Salmon River and its forks regularly warm to summer daytime peaks of 21-22°C, the best holding habitats are deep pools that have cold water sources, such as those at the mouths of tributaries, areas with hyporheic flow from springs, or those deep enough to thermally stratify.

Spawning habitat is mainly comprised of low-gradient, gravelly riffles or pool tail-outs, and is typically found at higher elevations than areas utilized by fall-run Chinook. Spawning and redd construction appear to be triggered by a change in water temperature rather than an increase in flows. Therefore, redd superimposition may occur when flows are low, limiting suitable habitat to areas near holding pools. Redd superimposition and even hybridization among spring- and fall-run Chinook has been noted in the mainstem Trinity River (Kinziger et al. 2008), South Fork Trinity River (Dean 1995), and Salmon River, though spatial segregation is still obvious upstream of Matthews Creek (Olson et al. 1992). More recent studies hypothesize an important role of dams and habitat degradation constraining suitable spawning habitat for Chinook, but could not quantify the role of Lewiston Dam or the Trinity River Hatchery on observed rates of redd superimposition or hybridization (Kinziger et al. 2008). Juvenile habitat requirements for spring-run UKTR Chinook salmon are presumably similar to those of fall-run UKTR Chinook salmon.

Distribution: UKTR spring-run Chinook were once found throughout the Klamath and Trinity basins in suitable reaches of larger tributaries (e.g., Salmon River) or smaller tributaries with suitable flows for holding and spawning (Wooley Creek). Historically, they were the most abundant Chinook in the basin, and were found in major tributary basins such as the Salmon, Scott, Shasta, South Fork and North Fork Trinity rivers (Moffett and Smith 1950, Campbell and Moyle 1991). Their distribution is now restricted by dams on the Shasta (Dwinnell), Trinity (Lewiston), and Klamath (Iron Gate, Copco 1 and 2, and J.C. Boyle) rivers. Passage of spring-run Chinook through Upper Klamath Lake to access holding and spawning grounds in the Sprague, Williamson and Wood rivers, was blocked in 1918 by completion of Copco 1 Dam (Hamilton et al. 2005). Currently, only the Salmon and the South Fork Trinity rivers maintain self-sustaining populations with little hatchery influence (SRF 2016). Approximately 177 km of habitat is accessible to spring-run Chinook in the Salmon River (West 1991), but most of it is underutilized or unsuitable. The South Fork Salmon River supports the majority of the remaining spawning population, although redds have been found in some smaller tributaries of the Salmon River basin including Nordheimer, Knownothing, Methodist, and Wooley creeks. In addition, there are small populations of spring-run Chinook in Elk, Indian, Clear and creeks.

In the Trinity River basin, spring-run Chinook salmon historically spawned in the East Fork, Stuart Fork, Coffee Creek, Hayfork Creek and the snowmelt-fed mainstem upper Trinity River (Gibbs 1956, Campbell and Moyle 1991). The completion of Trinity Dam in 1962 and Lewiston Dam in 1963 blocked access to 56 km of what was considered to be prime spawning and nursery habitat (Moffett and Smith 1950). Currently, Trinity River spring-run Chinook are present in small numbers in the New River (mainstem Trinity River), Hayfork and Canyon creeks (South Fork Trinity River), and the North Fork Trinity River, but only the South Fork population appears to maintain itself through naturally spawned fish (W. Sinnen, CDFW, pers. comm. 2013). LaFaunce (1967) found spring-run Chinook spawning in the South Fork Trinity

River from about 3 km upstream of Hyampom and in Hayfork Creek up to 11 km above its mouth. However, decreases in water quality and increases in water temperature have been cited as leading to extirpation of spring-run Chinook in Hayfork Creek (A. Hill, CDFW, pers. comm. 2017).

Trends in Abundance: The UKTR spring-run Chinook population once likely totaled greater than 100,000 fish (Snyder 1931, Moyle 2002). The spring run was thought to be the main run of Chinook salmon in the Klamath River, but the stocks had been depleted by the early 20th century as the result of irrigation, overfishing, mining, and other causes (Snyder 1931). Historical run sizes were estimated by CDFW to be at least 5,000 fish annually in each of the following Klamath tributaries: Sprague and Williamson rivers (Oregon), Shasta River and Scott River (CDFG 1990). The runs in the Sprague, Wood, and Williamson rivers were extirpated after the construction of Copco 1 Dam in 1918. Healy (1963) estimated 7,000-10,000 spring-run Chinook spawned in the South Fork Trinity River, while LaFaunce (1964) estimated about 11,600 spring-run Chinook adults returned to spawn in 1964. Very low numbers (approximately 500 adults) returned to Iron Gate Hatchery (IGH) on the mainstem Klamath during the 1970s, but could not persist without cold water during summer; the last spring-run Chinook returned to IGH in 1978 (Hiser 1979). The run in the Shasta River, probably the largest in the middle Klamath drainage, disappeared in the early 1930s as the result of habitat degradation and blockage of upstream spawning areas by the construction of Dwinnell Dam in 1926. The smaller Scott River run was extirpated in the early 1970s, possibly earlier, from a variety of anthropogenic causes that depleted flows and altered habitats (Moyle 2002). In the middle reaches of the Klamath, spring-run Chinook have been extirpated from their historical habitats except the Salmon River and one of its tributaries, Wooley Creek (NRC 2004). Less than 10 spring-run Chinook have been annually observed in Elk, Indian, and Clear creeks (Campbell and Moyle 1991).

In the UKTR, spring-run Chinook abundance is highly variable over time. Large swings in abundance can be partially attributed to changes in habitat and survival of young: West (1991) noted that spring-run Chinook egg survival to emergence in the Salmon River ranged from 2-30% in 1990. The number of spring-run Chinook salmon adults appears to be decreasing in the Klamath River while increasing on the Salmon River, but continue to be a fraction of historical runs (Hamilton et al. 2011). Quiñones et al. (2014) found a correlation between spring-run Chinook returning to the Salmon River and TRH returns in the same year, but these trends may reflect similar responses of both wild and hatchery-reared fish to changing environmental conditions rather than hatchery supplementation. The US Forest Service has been collecting and compiling summer snorkel survey data on returning fish in the basin for many years, and have found small numbers of spring-run Chinook jacks (yearling male fish) that have either matured in fresh water or returned to spawning tributaries after spending only a few months in the ocean (NMFS 2016, Figure 1).

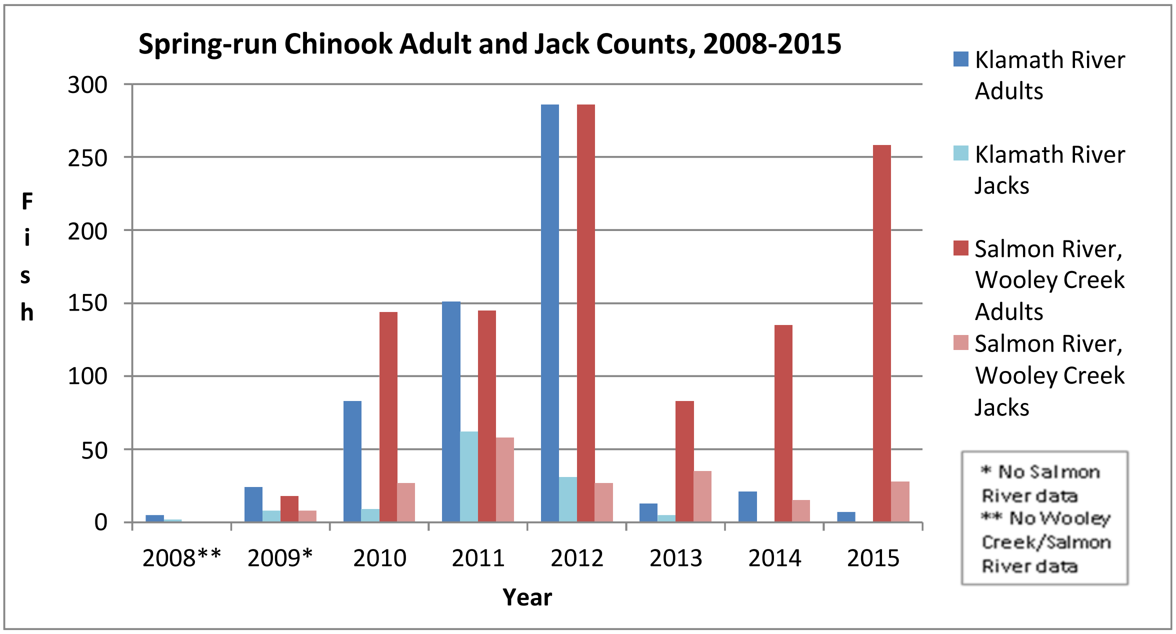


Figure 1. Snorkel survey counts of spring-run Chinook salmon adults and jacks in lower Klamath River tributaries, Salmon River, and Wooley Creek, 2008-2015. From L. Cyr, USFS, 2016.

Spring Chinook have been extirpated above Lewiston Dam on the Trinity River, but historically included more than 5,000 adults in the upper Trinity River and 1,000-5,000 fish in each of the Stuart Fork Trinity River, East Fork Trinity River, and Coffee Creek (CDFG 1990). For the 2015-2016, sampling year, 4,400 spring Chinook were estimated to migrate upstream of the Junction City weir, with only about a quarter (1,090 adults) estimated to be of natural origin (18% of the Trinity River Restoration Program target of 6,000 adults, Kier and Hileman 2016.) Over the last 30 years, an average of 263 fish were counted annually in the South Fork Trinity River (Hill et al. 2015).

While spring-run Chinook salmon are scattered throughout the lower Klamath and Trinity basins, the only viable wild populations appears to be limited to the Salmon, New, and South Fork Trinity rivers. The South Fork and New River remain the largest producers of spring Chinook abundances in the Trinity River tributaries (Hill et al. 2015). Mainstem Trinity River and nearby tributary numbers are presumably influenced by fish from the TRH, though this influence is likely small and decreases with distance from the hatchery (A. Hill, CDFW, pers. comm. 2017). Even if Trinity River tributary spawners are considered to be all wild fish, the total number of spring-run Chinook in the Trinity River rarely exceeds 1,000 fish, and may drop to < 300 in many years (Figure 2).

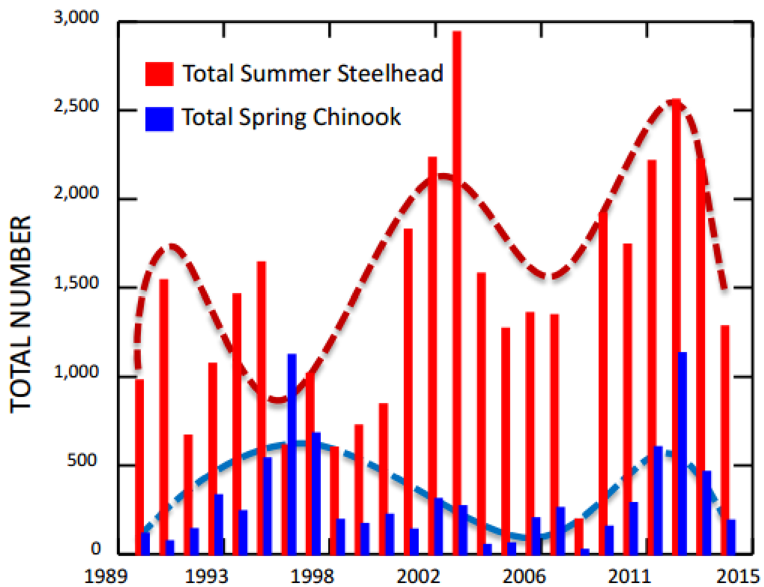


Figure 2. Spring-run Chinook salmon counted in summer snorkel surveys in Trinity River tributaries, 1990-2014. From Hill et al. 2015, Fig. 3, pg. 6.

In recent years, efforts have been made to compile all spring-run Chinook survey data by CDFW, from the U.S. Forest Service, CDFW, and others (Figure 3). The data include escapement and tribal/angler harvest of wild and hatchery fish in both the Klamath and Trinity basins. Since 1980, returns have fluctuated widely, but in most years, estimates are less than 3,000 individuals for the entire Klamath Basin. However, these numbers represent varying degrees of sampling effort among years, so are very imprecise and are best used to highlight trends rather than being actual numbers.

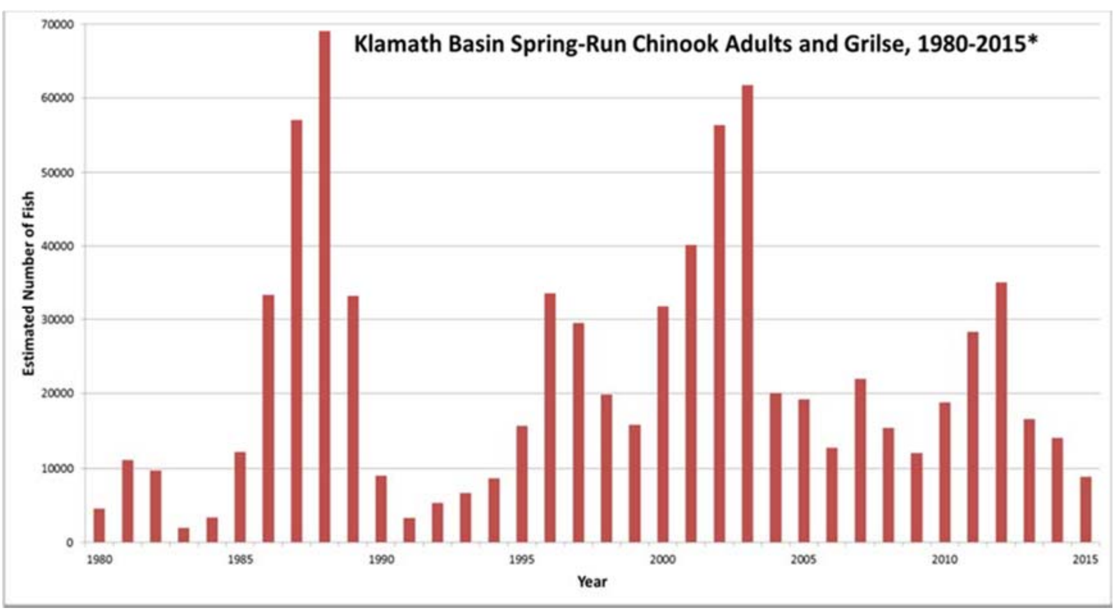


Figure 3. Estimates of Klamath Basin spring-run Chinook salmon adults and grilse, 1980-2015. Data from multiple sources and includes hatchery returns; all streams not surveyed every year. Data from CDFW 2016.

Factors Affecting Status: UKTR spring-run Chinook have been largely extirpated from their historical range because their distinctive life history makes them vulnerable to the combined impacts of climate change, dams, hatchery operations, and habitat degradation as well as other anthropogenic (Table 1) and natural factors (e.g., ocean conditions). UKTR spring-run Chinook have declined from being the most abundant run in the Klamath-Trinity system to one that is in danger of extinction in the near future. They may be the hardest species of all to recover because they migrate to spawning rivers in spring and are increasingly at risk to warm temperatures and low flows (Cannon 2016). In addition, climate change and ongoing (and more frequent) drought represent significant threats to the persistence of spring-run Chinook salmon in the Upper Klamath-Trinity rivers basin.

Dams. A significant portion of the historical UKTR spring-run Chinook range has been lost upstream of Iron Gate, Lewiston, and Dwinnell dams. These dams block access to about 970 km of upstream anadromous habitats of varying quality (Hamilton et al. 2005). They serve as barriers to adult holding and spawning habitats, as well as juvenile nursery areas, and have reduced resiliency of remaining small populations, reduced available habitat, and eliminated spatial segregation between spring and fall-run Chinook. This has likely led to significant interbreeding between fall- and spring-run Chinook in the Trinity River, to the detriment of each (Myers et al. 1998, Kinziger et al. 2008). Dams and diversions have also led to extirpation of spring-run Chinook in the Klamath and Shasta rivers due to alteration of water quality and temperature, channel simplification, and disconnection of mainstem river channels from floodplains (Lewis et al. 2004. Peak flow timing has also shifted a month earlier than prior to dam construction (Hamilton et al. 2011), further reducing viability of this life history strategy. In the Shasta River, spring-run Chinook historically entered during late May through early July; however, diversions of cold (11°C) water from Big Springs for irrigation created a thermal regime that could no longer support year-round juvenile rearing (Stenhouse et al. 2012). This situation has been partially rectified through purchase of Big Springs by The Nature Conservancy and CDFW in 2008 (Jeffres et al. 2009).

There is potential for UKTR spring-run Chinook salmon to be restored to large portions of their former range in the Klamath-Trinity basins through dam removal, especially on the mainstem Klamath. The four lowermost Klamath Dams (Copco 1 and 2, J.C. Boyle, and Iron Gate) are slated for removal beginning in 2020, which will restore access to significant potential holding a rearing habitat of unknown quality. Under the Klamath Basin Hydroelectric Agreement and Klamath Settlements, hundreds of kilometers of potential Chinook, coho, and steelhead habitat will be available in the future. The quality of the habitat upstream of these barriers is unknown, and significant, long-term restoration and monitoring will be required to allow anadromous salmonids to utilize the habitat in the future.

Agriculture. Most spring-run Chinook holding and rearing habitats are upstream of areas heavily influenced by agriculture (e.g., Scott and Shasta valleys); nonetheless, pasture and crops along the Shasta and Scott rivers utilize cold water that would otherwise be available for instream flow. Agricultural return waters are generally warm and often deliver pesticides, fertilizers and other pollutant to streams. These degrade the quantity and quality of habitat for spring-run Chinook, resulting in fewer successful spawning adults and hatching young in the watershed. With the legalization of marijuana cultivation in California under Proposition 64, more effort must be placed on understanding, quantifying, and reducing the extent and magnitude of impacts of this pervasive industry in the UKTR on spring-run Chinook. For a full discussion of impacts of agriculture and marijuana cultivation on fishes in the watershed, see the Klamath Mountains Province winter steelhead account.

Logging. Logging and associated road building have dramatically altered aquatic habitats in the Klamath and Trinity River basins (NRC 2004). Intensive and widespread logging began in the mid-19th century and legacy effects continue to affect rivers and streams in this region. Historical logging and the development of most early access roads occurred with little regard for environmental impacts. The steep and unstable slopes of this region, combined with local geology, make them particularly prone to erosion following road development and timber harvest (NRC 2004). The low numbers of spring-run Chinook salmon currently using the heavily-logged South Fork Trinity River may be due to the catastrophic 1964 flood, which altered channel morphology and hydrology and triggered landslides that filled in holding pools and covered spawning beds. Primary and ongoing impacts from timber operations in the Klamath-Trinity province include: increased erosion rates (delivering large amounts of sediments into streams, which often imbed spawning areas and fill essential holding pools for spring Chinook needed by holding adults in the summer), increased surface run-off of precipitation (and corresponding and decreased aquifer recharge capacity, which decreases cool hyporheic flow during summer months and increases flash flooding, leading to increased frequency of flash flooding in streams), and increased summer stream temperatures due to lack of aquifer recharge and reduced canopy and riparian vegetation (instream shading, KNF 2002). Logging also removes trees that historically provided large wood that increased habitat complexity for all life history stages of all salmonids: its absence reduces the utility of remaining habitats.

Hatcheries. The Trinity River Hatchery below Lewiston Dam is the only hatchery in the Klamath basin that still raises spring-run Chinook salmon. The impacts of hatchery propagation on wild spring-run fish in the Trinity basin may be substantial. Most naturally spawning fish are considered to be of wild origin, though there is a component of hatchery fish that spawn in natural areas, particularly close to the Lewiston Dam (W. Sinnen, CDFW, pers. comm. 2017). Mixed runs of wild and hatchery-reared fish tend to segregate themselves above Junction City (Rkm 127), with a significant portion of hatchery fishes returning to TRH. However, artificial selection in a hatchery has been demonstrated to reduce fitness in fish reproducing in the wild (Araki et al. 2007, 2009). Hatchery spring-run Chinook hybridize with fall-run fish on the Trinity due to reductions in habitat and shifts in run timing (Kinziger et al. 2008). Attempts to ameliorate these impacts to the extent possible are reflected in a TRH hatchery genetics management plan (J. Hileman, CDFW, pers. comm. 2017).

Rural/ residential development. The long history of mining and logging in the Klamath and Trinity basins has left an extensive network of roads, which continue to provide access to many remote areas, facilitating rural development. Rural development, particularly in the steep, mountainous terrain that characterizes this region, may have substantial impacts on streams through increased surface run-off, sedimentation, effluent from septic tanks and other pollutants, water diversion, deforestation and habitat fragmentation. Over time, these increasing human pressures on watersheds exacerbate other problems.

Fire. Altered forests in the region have also become more prone to large-scale catastrophic fires and increased erosion. For example, over 50% of the Salmon River watershed has been severely burned in the past 100 years (NRC 2004). Portions of UKTR watersheds are subject to intense fires (e.g., Forks, Salmon, and Corral complex fires, 2013), and are likely to increase in frequency under predicted climate change scenarios. Fires can increase water temperatures of important holding and rearing habitat, cause landslides, increase sediment loading, and remove shading canopy cover, all to the detriment of salmonids. Large rainfall events in these areas can quickly mobilize debris from the steep, fragile slopes and bury spawning and rearing habitats in headwater reaches.

Mining. Mining has dramatically altered river and stream habitats in the Klamath-Trinity Province, with lasting legacy impacts in many areas form historical activity. Intensive hydraulic and dredge mining for gold occurred in the 19th century, causing severe stream degradation and alteration to channel morphology. In fact, mining was a principal cause of decline of spring-run Chinook in the Scott River and large areas in the Trinity River, followed by some recovery after large-scale mining ceased (Cramer et al. 2010). Spring-run Chinook once swam through Scott Valley; now, only a degraded river winds through immense piles of dredge tailings (SRRC 2009). Historical mining impacts still affect the Salmon River spring-run Chinook population as the estimated 16 million cubic yards of sediment disturbed between 1870 and 1950 are slowly transported through the basin (J. West, USFWS, pers. comm. 1995). Legacy effects include disconnected and constricted juvenile salmon habitats, filled-in adult holding habitats, degraded spawning grounds, and altered annual hydrograph of many streams. Pool in-filling is a particular problem because high stream temperatures have been demonstrated to reduce survival of both holding spring adults and rearing juveniles (Elder 2002).

Although suction dredging has been banned since 2009 (CDFW 2016), historical dredging has been particularly damaging to spring-run Chinook habitats. Of particular concern, in the Klamath, Salmon, and Scott rivers and their tributaries is the creation of piles of suction dredge tailings in the past that may be utilized by spawning salmonids. Although these tailing piles are often comprised of suitable substrates for salmon redd creation and successful spawning, they are likely to be mobilized during high flows, greatly reducing survival of embryos within the gravel.

Harvest. Both legal harvest of spring-run Chinook in the ocean and illegal harvest of adults in-river can limit abundance of spawning populations. Because UKTR spring- and fall-run Chinook are indistinguishable when encountered at sea and belong to the same ESU under federal law, they are taken legally in sport and commercial fisheries in the ocean. Recreational and tribal harvest of both wild and hatchery-origin spring-run adults is also allowed in the lower Klamath and upper Trinity rivers under special regulations. Spring-run Chinook are subject to special harvest restrictions from the South Fork Trinity River and mainstem Klamath reaches upstream of Weitchpec, during their migration season (W. Sinnen CDFW, pers. comm. 2017). Based on management and regulatory mechanisms rather than biological reasons, regulations stipulate that Chinook in the Klamath-Trinity rivers before August 31 are considered spring-run and no more than 2 fish per day may be taken per angler, per day; Chinook in the river after September 1 are considered fall-run and are managed by an annual quota set by the Pacific Fishery Management Council. No harvest of spring-run Chinook is allowed in the South Fork Trinity or Salmon rivers (CDFW 2016).

Holding adults are vulnerable to poaching due to their reliance on deep pools with cold water, although the extent to which poaching affects populations is largely undocumented. Recreational angling may strain small populations through inadvertent hooking mortality.

Recreation. Spring-run Chinook may be absent from many suitable areas because of repeated disturbance by humans. Gold dredgers, swimmers, and boaters may stress and displace fish, particularly holding adults (P. Moyle and R. Quiñones, pers. obs. 2000). Displacement from suitable habitats may make spring-run Chinook less able to survive natural periods of stress (e.g., high temperatures) or survive to spawning. Increased and unnatural movements of fish make them more noticeable, potentially increasing incidence of poaching. Not surprisingly, spring-run Chinook tend to persist mostly in the most remote canyon pools in their watersheds, in some of the same habitats utilized by summer steelhead.

Alien species. Alien species are rare in the UKTR system, although brown trout (*Salmo trutta*) may pose some predation and competition risk to juvenile salmon in the upper and North Fork Trinity River. While not alien species, native California black bear (*Ursus americanus californiensis*) are known predators on spring Chinook in the Trinity River watershed (J. Hileman, CDFW, pers. comm. 2017). River otters (*Lontra canadensis*) will prey on summer-run steelhead in the Klamath-Trinity basins, suggesting they could also impact spring-run Chinook salmon, especially during periods of low flows while fish are concentrated in pools (M. Sparkman, CDFW, pers. comm. 2016). However, natural predation rates are not well documented.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of UKTR spring-run Chinook salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Large portions of historical range are blocked by dams.
Agriculture	Medium	Agriculture reduces habitat quality and quantity through diversions, warm return waters, and pollutants.
Grazing	Medium	Grazing and irrigated pasture pervasive on public and private lands.
Rural /residential development	Low	Cumulative effects of roads and widespread rural development pose ongoing and chronic threats.
Urbanization	Low	Urban areas are few and restricted to main rivers.
Instream mining	Medium	Dredge mining currently banned but legacy effects remain in many areas; gravel mining may cause localized impacts.
Mining	Medium	Legacy effects of intensive and widespread gold mining remain severe in some areas.
Transportation	Medium	Roads present along many streams; impacts from increased run-off, sedimentation and habitat fragmentation.
Logging	High	Both legacy and ongoing impacts have dramatically altered and degraded salmon habitats.
Fire	Low	Climate change may contribute to increased fire frequency and intensity, affecting headwater holding areas.
Estuarine alteration	Low	Klamath River estuary is less altered than most on the North Coast.
Recreation	Medium	May be a chronic source of disturbance for some populations.
Harvest	Medium	Legal and illegal harvest takes many fish; evidence of poaching is annually found in the Salmon River basin.
Hatcheries	High	Spring Chinook stocks are supplemented by TRH production. Potential reduction in fitness through spring-/fall-run interbreeding.
Alien species	Low	Few alien fishes co-occur with the salmon.

Effects of Climate Change: Climate change is likely the greatest threat to the long-term persistence of stream-maturing life history expression in UKTR spring-run Chinook salmon. Moyle et al. (2013) rated the UKTR spring-run Chinook salmon as "critically vulnerable" to extinction because of the added effects of climate change on top of diminished populations. Climate change is already reducing stream volume, increasing stream temperatures, and altering seasonal flow patterns of water, which will likely lead to further reductions in suitable upper watersheds that spring-run Chinook use to oversummer. Increased protection and restoration efforts to improve stream flows, allowing accessibility to prime holding and spawning habitat, and maintenance of cool temperatures in headwater tributaries for both spring Chinook salmon and summer steelhead, are essential for recovery. Recent (2012-2016) drought in California may have contributed to higher pre-spawn mortality of spring-run Chinook in the basin, because lower flows and higher water temperatures are associated with increased pre-spawn mortality rates, which are typically less than 30% across Chinook salmon in California (Bowerman et al. 2016). Since 2012, the Bureau of Reclamation has augmented flow releases downstream of Trinity and Lewiston dams in August and September to reduce die-off of fall-run Chinook in the lower Klamath associated with high water temperatures and disease outbreaks (USBOR 2016). The impact of these releases on spring-run Chinook is unknown at this time.

Climate change is predicted to decrease snowpack (reduce instream flows), increase water temperatures, and alter flow timing. Lower flows are of particular concern in the spring and summer, because remaining water can reach daytime temperatures of 24-26°C in the Klamath, rendering them unusable to spring-run fish. The Salmon River already reaches summer temperatures of 21-23°C, approaching lethal thresholds for salmonids. 1-2°C increases in stream temperatures by 2100, as forecasted in moderate climate model projections, will greatly reduce the amount of suitable habitat available for spring-run Chinook. Reduced reservoir recharge may limit thermal stratification and the amount of cold water pool available for environmental flows via dam releases, which may be particularly acute in the Klamath River. The frequency and intensity of both drought and flash floods are likely to increase as well. Studies similar to Nakamoto's (2004) study of habitat use by summer steelhead in the New River should be undertaken to determine how emerging temperature/hydrologic regimes in the UKTR impact salmonid habitat use, survival, and productivity.

Climate change may also increase the incidence of disease outbreaks due to warmer water temperatures and further stress adult salmon. For example, warmer temperatures favor epizootic outbreaks of *Ichthyophthirius multifiliis* and transmission of the bacteria *Columnaris*, which is associated with higher mortality of pre-spawn salmonids that are exposed to above-optimal water temperatures (Strange 2007, Power et al. 2015).

Status Score = 1.6 out of 5.0. Critical Concern. The principal self-sustaining wild populations of UKTR spring-run Chinook exist in the Salmon and South Fork Trinity rivers. The largest component of the run, the hatchery-influenced upper Trinity River portion, was derived from original stocks, which migrated past Lewiston Dam historically (W. Sinnen, CDFW, pers. comm. 2017). UKTR spring-run Chinook are considered a Sensitive Species by the USDA Forest Service and a Species of Special Concern by CDFW (2015). The spring-run Chinook salmon of the Central Valley was listed as threatened under the California Endangered Species Act in 1999, but this designation has not yet been made for fish in the Upper Klamath-Trinity rivers basin (CDFW 2016). Genetic risk from low populations and interaction with TRH fish, climate change impacts, and anthropogenic threats plague UKTR spring-run Chinook salmon and make them vulnerable. Without significant human intervention in the form of changing logging practices, updating hatchery operations, and especially removal and restoration of the four lowermost Klamath dam sites, this run of fish is likely to become extirpated within 50 years.

Table 2. Metrics for determining the status of UKTR spring-run Chinook salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Only Salmon River and South Fork Trinity River support wild, self-sustaining populations.
Estimated adult abundance	2	A few hundred natural spawners support the population, with attendant impacts of small populations and hatchery influence.
Intervention dependence	2	Hatchery stocks maintain the mainstem Trinity run; Klamath dam removal in Klamath needed to restore access to historical range.
Tolerance	2	Narrow temperature tolerance (<20ºC) during migrations; temperatures and other factors limiting in summer holding areas.
Genetic risk	1	Hybridization with fall-run and/or hatchery spring-run is occurring in Trinity River; fitness reduction may result from hybridization.
Climate change	1	Increased temperatures, density of adults and juveniles and disease outbreaks, and reduction in suitable habitats will limit populations.
Anthropogenic threats	1	3 High, 7 Medium factors.
Average	1.6.	11/7.
Certainty (1-4)	3	Fairly well-studied.

Management Recommendations: UKTR spring-run Chinook are an indicator of ecosystem health due to their narrow tolerance to water temperature and quality and presence in upstream reaches of river during warm summer months when flows are reduced. The rarity of cool water refuges throughout the UKTR during their migration period is already a significant threat to long-term persistence.

Monitoring of spring-run Chinook by several agencies occurs annually throughout the Klamath-Trinity system. However, data need to be compiled and standardized to be most useful for managers. Existing surveys demonstrate that suitable habitat exists for adult holding and spawning in the UKTR, yet spring-run Chinook abundance continues to fluctuate at low numbers. Spring-run Chinook may be particularly susceptible to warming trends, especially in the face of predicted climate change impacts and ongoing drought, which has reduced flows during summer and fall months since 2012. To combat reduced flows and increased temperatures in the mainstem Trinity and Klamath rivers, the U.S. Bureau of Reclamation increased flows up to 1,300 cfs out of Lewiston Reservoir in 2012-2016 (USBOR 2016). The goal of these augmentation flows was to keep density- and flow-dependent disease of fall-run Chinook at bay in the lower Klamath River. Disease was implicated in the 2002 die-off and may have had unintended consequences on spring-run Chinook in the Trinity River. The effects of these flows are unknown at this time. Restoration activities in the Salmon and South Fork Trinity rivers, two spring-run Chinook strongholds, should be prioritized (SRF 2016). For example, plans to place hundreds of whole trees with intact root wads via helicopter into the South Fork Trinity River in the snowmelt-fed Hayfork Valley should be replicated and expanded to other key spring-run habitats (J. Smith *in* SRF 2016).

Of all salmonids in this drainage, spring-run Chinook would likely benefit the most from increased access to cold-water habitats. Reconnecting historical habitats in the upper watersheds of the Klamath and Trinity rivers, with their tributaries fed by spring sources and snowmelt, are critical after dam removal for long-term persistence of this run. Habitat for spawning and rearing should be restored as quickly as possible to avoid potential negative impacts on salmonids immediately following dam removal (Quiñones et al. 2014). Literature suggests that these activities will increase diversity of life histories and increase resiliency of anadromous fish populations by offering the ability to differentiate spatially and temporally over the habitat (Hodge et al. 2015). However, restoration and volitional re-colonization of historical habitats may not be enough to recover the run. The tradeoffs of employing hatcheries to raise ocean-maturing and stream-maturing Chinook ecotypes should also be carefully weighed and considered, as hatchery selection pressures may negatively impact genetic integrity and variation in colonizing salmon (Quiñones et al. 2014). However, these potential impacts may be outweighed by bolstering wild runs with greater numbers of fish to increase genetic diversity in the short-term and avoid inbreeding depression or founder's effect (Fraser 2008). Other specific recommendations to restore UKTR spring-run Chinook salmon include:

- Restore the Shasta River as a cold-water refuge for all salmonids in the Klamath basin by recapturing spring flows in the river, reducing ground water extraction and exploration of potential for developing fish passage over Dwinnell Dam to spawning habitat.
- Protect and restore Blue Creek watershed to maintain critical coldwater flows in summer and fall months for migrating spring-run Chinook, as the Yurok Tribe and Western Rivers Conservancy have committed to do.
- Manage the Salmon River as a spring-run Chinook and summer steelhead refuge by restricting extractive use of the river in summer by promoting alternative sources of water for agriculture and human consumption and amending regulations to protect holding fish.
- Develop restoration actions and priorities for reducing the impacts of sediment inputs from roads, logging and other activities into rivers of the Klamath-Trinity system, especially on public lands. In addition, determination of effects of diversions, stream alterations, and pesticides/herbicides associated with illegal marijuana cultivation should be undertaken and increased under funding provisions associated with recent passage of Proposition 64.
- Develop a program to investigate impact(s) of the Trinity River Hatchery on spring-run Chinook populations (e.g., number of hatchery-reared fishes spawning in the wild, genetic shifts in population) and manage hatchery production accordingly. Rates of hybridization between spring-run and fall-run Chinook, and relative fitness of the offspring, should be paid particular attention.
- Investigate whether spring Chinook from a conservation hatchery built expressly for this purpose can play a role in facilitating re-colonization of Klamath River tributaries after dam removal occurs (after Kinziger et al. 2008). If such an approach is explored, efforts

must be made to reduce genetic impacts of founder's effects and inbreeding/outbreeding depression.

- Manage downstream reaches below dams to favor conditions for out-migrating smolts.
- Limit recreational in-river harvest to a mark-selected, fishery for 100% adipose fin-clipped TRH-produced spring-run Chinook to keep them separate from wild fish.
- Increase enforcement focus of fishing and land use regulations in over-summering areas, especially related to groundwater pumping, illegal diversions, marijuana cultivation, etc. Proceeds from taxation of legalized marijuana could help fund expansion of staff for targeted enforcement and administrative management of such illicit activities.

CENTRAL CALIFORNIA COAST COHO SALMON
Oncorhynchus kisutch (Walbaum)

Critical Concern. Status Score = 1.3 out of 5.0. Most or all populations in small coastal streams will be extirpated from California in the next 50 years without increased intervention and protection of watersheds.

Description: Central California Coast coho salmon are morphologically similar to coho salmon in the Southern Oregon-Northern California Coast (SONCC) Evolutionary Significant Unit (ESU). Coho in the two ESUs are so similar that they can only be distinguished by genetic differences (Gilbert-Horvath et al. 2016).

Taxonomic Relationships: California Central Coast (CCC) coho salmon are highly adapted to local environments at the southern end of coho distribution. Bucklin et al. (2007) showed that each population in every stream sampled was distinctive and most closely related to populations in nearby streams. Populations demonstrate concordance between their geographic and genetic differences, with adjacent populations from the Mendocino coast to the Golden Gate generally appearing more closely related to each other (Gilbert-Horvath et al. 2016). Populations further south did not fit this pattern (Good et al. 2005), presumably because movement among these basins is more pervasive than among ones further north; this problem has been enhanced by extirpation of populations from streams between Santa Cruz and Marin counties. There has also been some dispersal by humans through stocking, such as the movement of coho from Scott Creek to Waddell and Gazos creeks (Santa Cruz Co.) (Smith 2015). However, Bucklin et al. (2007) confirmed that planting of coho from outside stocks in the past has had minimal influence on the genetics of local populations within this ESU. A similar genetic and geographic pattern was also observed at the southern end of steelhead range (see south-central coastal and southern steelhead accounts).

Life History: The first comprehensive life history study of coho salmon was done on fish of the CCC ESU, namely the classic studies in Waddell Creek by Shapovalov and Taft (1954). Their life history throughout their range is summarized in Sandercock (1991) while Baker and Reynolds (1986), Moyle (2002), CDFG (2002), and NMFS (2012) review their biology in California. In most respects, the life history of CCC coho is the same as that of SONCC coho, including the presence of small numbers of juveniles that spend two years in the creeks and overwinter and rear in non-natal streams and estuaries during every month of the year (see SONCC coho salmon account). Dr. J. Smith of San Jose State University has in recent years continued the life history and monitoring studies of Shapovalov and Taft (Smith 2015).

Habitat Requirements: Habitat requirements of CCC coho are basically the same as those of SONCC coho, the summary of which is based on Moyle (2002) and DFG (2002, 2004). Smith (2015) notes that success of coho in Scott and Waddell creeks depends on the timing and severity of floods from winter rainstorms and the timing and severity of droughts. Severe high flow events that occur early in the winter (December, January) can scour pools, move large wood, open the mouth of the lagoon for access, and generally improve coho habitat, while similar flood events later in the season (February, March) can scour out redds or flush juvenile coho out of over-wintering habitat (pools, side channels, under large wood). Likewise, severe drought, especially if aggravated by diversions, can dry up stream reaches and confine juvenile coho to pools, which may become too shallow or warm to support them. Smith (2015) notes other CCC coho populations face similar problems.

Distribution: For broad aspects of coho salmon distribution, see the SONCC coho account. CCC coho were historically native to California coastal streams from Punta Gorda (southern Humboldt Co.) down to the San Lorenzo River and (probably) Soquel and Aptos creeks (Santa Cruz Co., Spence et al. 2005, Adams et al. 2007, NMFS 2012), as well as at least four streams tributary to San Francisco Bay (Leidy et al. 2005). It is also possible that a small run once existed in the Sacramento River (Brown et al. 1994). They are currently extirpated from tributaries to San Francisco Bay (NMFS 2012).

The distribution and abundance analysis for California coho salmon of Brown and Moyle (1991) and Brown et al. (1994) was updated by Spence et al. (2005), Bjorkstedt et al. (2005), Spence and Thomas (2011), NMFS (2012), and DFG/W (2002, 2004, 2015), from which this information comes. Bjorkstedt et al (2005) identified 76 streams in the CCC region that likely historically supported populations of coho salmon, based on analysis of historical records, genetic structure, and hydrology. Twelve of these were large enough to support self-sustaining populations through severe environmental conditions ("functionally independent populations"), while the rest presumably required periodic recruitment from permanent populations to be sustained ("functionally dependent"). The 12 main populations were found in the following rivers: Ten Mile, Noyo, Big, Albion, Navarro, Garcia, Gualala, Russian, and San Lorenzo, along with Lagunitas and Pescadero creeks. With a few exceptions, CCC coho distribution is linked to the cool fog belt, close to the ocean; they are also associated with coastal redwood forests in the EPA's West Coast Forest Ecoregion.

CCC coho salmon thus were once more or less continuously distributed in coastal streams from Mendocino County south to Santa Cruz County, with extensive inland distributions in the larger river systems such as the Eel. However, the general trend has been downward in the number of wild populations, with individual populations becoming more isolated, the overall distribution becoming fragmented, and fish being extremely rare in the southern two-thirds of the historical range of this ESU (Swales *in prep.*).

Trends in Abundance: Overall historical abundance of coho salmon in California is discussed in the SONCC coho account. There seems to be little doubt that CCC coho are closer to extinction than SONCC coho:

CCC Coho salmon... "are listed as an endangered species... due to a precipitous and ongoing decline in their population. Since their initial listing in 1996...their population has continued to decline and the species is now close to extinction (NMFS 2012, p. v.)."

"Overall, all CCC coho salmon populations remain, at best, a slight fraction of their recovery target levels, and, aside from the Santa Cruz Mountains strata, the continued extirpation of dependent populations continues to threaten the ESU's future survival and recovery." (NMFS 2016, pg. 11).

"...the overall long-term trend in coho salmon populations in most monitored streams in the State remains downward and many populations have either already been extirpated or may be approaching extirpation (Swales *in Prep.*, p. 17)."

Very rough estimates indicate that the number of coho salmon returning to streams in the CCC region 50-60 years ago was somewhere between 50,000 to 100,000 spawners (or more) per year, with 350 or more streams used for spawning and rearing. This suggests a long-term decline in excess of 95% in population size and a decline in number of streams used annually on the order of 50%, although most of the streams with recent records do not have fish every year, and those that do have very small numbers.

In the first statewide assessment of coho status, Brown et al. (1994) considered 5,000-7,000 fish to be a realistic estimate of the total number of naturally spawned adults returning to all California streams each year in 1987-1991 (20% CCC coho, or ca. 1,000 to 1,400 spawners). NMFS (2012) put the estimate at 2,000-3,000 spawners in 2011, with an estimate of less than 500 CCC coho in 2009. The actual number is hard to estimate, and varies with cohort and annual survival in both stream and ocean. Presumably, the actual number varies between 500 and 3,000 per year, depending on conditions in both fresh water and the ocean. Regardless, the number is low and represents a long-term decline from thousands of spawners in the ESU as late as the 1940s.

Mendocino County. Swales (2016) noted that CCC coho were once found in over 200 streams in the county, including most tributaries of Ten Mile, Noyo, Big, and Navarro Rivers. However, a 2002 assessment found that only 62% had signs of still supporting coho salmon. Since that assessment, all streams monitored "have shown a continued downward trend" (Swales 2016, p 16). For example, in the Noyo River, where reasonably good records have been kept since the 1960s (Grass 2008), early counts ranged between 1,200 and 5,000 spawners. Since 1990, most counts have been < 500 fish, with 79 fish in 2005-2006 and 59 in 2006-2007 (Grass 2008). Numbers have continued to remain low but fluctuating since then (Swales 2016).

Sonoma County. Coho salmon are still found in just 5 of 70 streams in Sonoma County, in the Russian River and Gualala River watersheds. In the Russian River, coho are being reared in Russian River Coho Salmon Captive Broodstock Program (RRCSCBP) at Warm Springs Hatchery, on Dry Creek. Outplants of juveniles have been successful, especially in the Austin Creek watershed, resulting in about 200 returning adults in 2010-11, 400 in 2011-12, 500 in 2012-13, 300 in 2013-14, and 400 in 2014-15, up from 2-7 fish 2000-2009 (Swales *in prep.*, Higgins 2016) (Figure 1). The increasing trends reflect the success of stocking juveniles in tributary streams to the Russian River. Recent hatchery broodstock introduction efforts from fish from the Russian River and Olema Creek into Salmon Creek have allowed successful spawning, with juveniles captured there for the first time in many years (NOAA Fisheries 2016).

Marin County. CCC Coho are present today mainly in two watersheds, Redwood Creek and Lagunitas Creek. They seem to have largely disappeared from Redwood Creek during the 2011-2015 drought years. A significant proportion of the remaining fish are found in Lagunitas Creek and its tributaries. From 1997-98 through 2014-15, Ettlinger et al. (2015) recorded between 26 and 634 coho redds; the average number of redds was about 250, representing about 500 fish assuming one male and one female per redd. The assumption that adult numbers are twice the redd count i.e. two fish per redd, is consistent with the numbers of adults estimated through spawner surveys (Ettlinger et al. 2015).

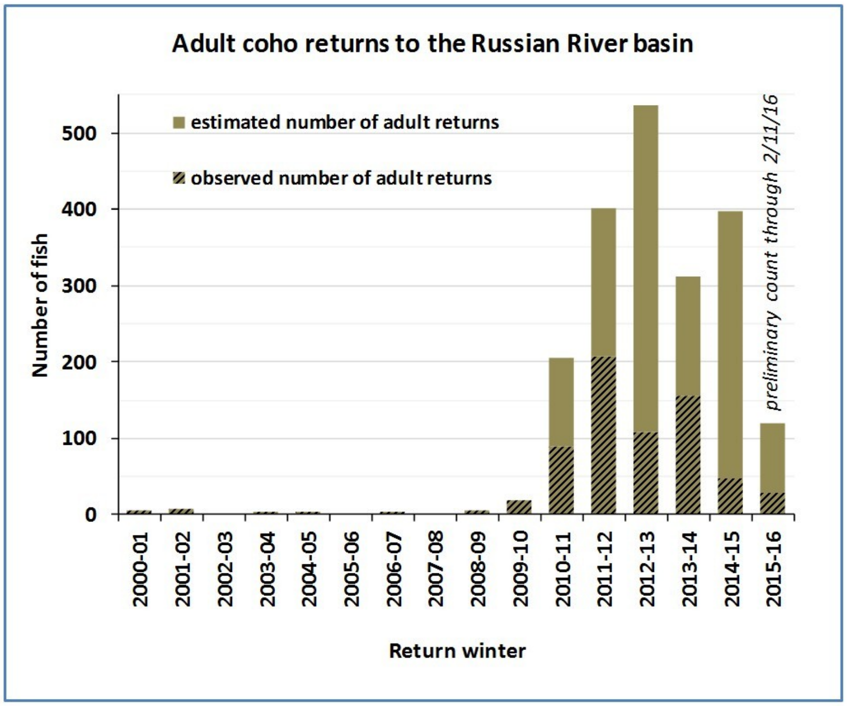


Figure 1. Adult coho salmon returns to the Russian River basin, 2000-2016. Data from UC Davis Cooperative Extension, 2016 (Higgins 2016).

San Francisco Bay tributaries. Coho were historically found in many tributaries to San Francisco Bay, including: San Mateo Creek (San Mateo Co.), Walnut Creek (Contra Costa Co.), San Leandro Creek (Alameda and Contra Costa Co.), San Pablo Creek (Contra Costa Co.), Strawberry Creek (Alameda Co.), Temescal Creek (Alameda Co.), San Lorenzo Creek (Alameda Co.), Alameda Creek (Alameda Co.), Coyote Creek (Santa Clara Co.), Guadalupe River (Santa Clara Co.), San Francisquito Creek (Santa Clara and San Mateo Co.), Corte Madera Creek (Marin Co.), Mill Valley Creek (Marin Co.), and Napa River (Napa Co.), and likely occurred in other tributaries as well, but data are inconclusive (Leidy 1983, Leidy et al. 2005).

San Mateo and Santa Cruz counties. The southernmost populations of coho salmon were originally found in at least 17 streams south of San Francisco Bay (Adams et al. 2007). At present, juveniles survive mainly in Scott, San Vincente, Soquel, and Waddell creeks, although they were present in Gazos Creek until 2005 (Smith 2015). The Scott drainage recorded the highest returns of coho in over a decade during 2014/2015 (NOAA Fisheries 2016). Returns are now entirely dependent on the Kingfisher Flat Restoration Hatchery (Smith 2015). The most recent samplings in Pescadero Creek and the San Lorenzo River indicate that coho have been extirpated in those two independent populations (NOAA Fisheries 2016).

In summary, almost all of the remaining CCC streams with coho have populations of fewer than 100 spawning adults, unless enhanced through hatcheries. These small populations are probably below the minimum population size required to preserve the genetic diversity of the stock and to buffer them from natural environmental disasters. There is every reason, therefore, to think that most CCC coho populations are facing extirpation in the near future, with the exception of populations in Lagunitas Creek, Russian River, and Santa Cruz County drainages, due largely to the heroic efforts by stream managers and conservation hatcheries. In most streams, reliable data on numbers of spawners, especially in recent years, is difficult to come by. The available information indicates that CCC coho salmon live in a small fraction of their historical habitat, with low population sizes. To make matters worse, these fish are mostly in isolated populations that show evidence of genetic and demographic factors that increase the likelihood of extirpation (Bucklin et al. 2007).

In the past 10 years, all CCC coho salmon populations have remained low, with numbers in 2007-08 being exceptionally low. The findings of Bucklin et al. (2007) suggest that most CCC coho populations are in a state of collapse from which recovery will be difficult. The drought has likely also depressed populations somewhat, even if temporarily.

Factors Affecting Status: The same factors that affect SONCC coho populations affect CCC coho populations, only more so. NMFS (2012) notes "Logging, agriculture, mining, urbanization, stream channelization, dams, wetland loss, water withdrawals, and unscreened diversions for irrigation contributed to the decline of the CCC coho salmon ESU" (p 89). As indicated in the SONCC account, the effects of anthropogenic change on coho are particularly severe because of the rigid life-history age-structure. Such stochastic events as floods or severe droughts, when acting on a severely depleted population, can eliminate one or more entire year classes from a stream, although sometimes precocious male parr, one year old jacks, and strays from nearby streams can help to bring back a missing year class. There is good evidence that this has already happened repeatedly in Waddell Creek and other coastal drainages, where the decline of coho is linked to poor stream and watershed management.

Agriculture. Unlike the situation for SONCC coho, many of the heavily logged watersheds in the CCC have not been returned to forest, but have been converted to agricultural lands, especially vineyards. Most of the Navarro River basin, for example, has been converted from dense redwood forest to open farmland, with much of the water diverted for agricultural use. As a result, due to low flows, much of the watershed is largely incapable of supporting *any* salmonids, much less coho salmon (Viers 2008). In the Russian River, the water in the tributaries is all over-allocated and diversion (for frost protection of vineyards) takes place even in winter, leaving little water for fish. A developing and critical issue in coho streams in California is the diversion of water for legal and illegal marijuana growing operations (NMFS 2012), which during summer can desiccate critical rearing habitat.

Dams. The Russian River has two major dams (Warm Springs and Coyote Valley) that have drastically altered its flow regime. In Lagunitas Creek, however, summer flow releases from Peters Dam help to create cold water habitat needed for persistence of juvenile coho. The creek has seven dams on it and is an important source of water for the Marin County Water District.

Urbanization. Another growing problem is urbanization, which has eliminated populations in the San Francisco Bay region and is increasingly contributing to the loss of CCC coho habitat in streams elsewhere. For example, a key tributary to Lagunitas Creek, San Geronimo Creek, is now lined with houses and controversy rages around protecting the creek from encroaching lawns and other issues.

Logging. In CCC coho streams, the most severe damage from logging is a legacy of historical logging practices, as described graphically by NMFS (2012). Starting in the 19th century, unrestricted logging caused massive erosion, removed riparian vegetation and woody debris from channels, caused stream temperatures to increase, filled pools with silt and gravel, altered stream channels, and degraded water quality. The redwood forests were logged off almost completely before 1900. On the Mendocino Coast, the first wave of redwood logging occurred in the late 1800s, and the practices employed severely modified coho habitats. Splash dams were commonly used to get logs from the harvest site down to ports at the mouths of rivers, and crib dams were common on the larger streams. Crib dams impounded water upstream so that logs could be floated downstream, or so water could be released to flush logs that had been dragged into the channel below the dams. Often streams had multiple crib or splash dams left in place on them for many years, preventing upstream migration by salmon. In the Santa Cruz Mountains, virtually all of the redwood forests, with the exception of the headwaters of the San Lorenzo (Big Basin State Park), a small grove near Felton, and some groves in the headwaters of Pescadero Creek were gone before 1900 (B. Spence, pers. comm. 2016). Although splash damming was apparently not used on the San Lorenzo River, mill pond dams were built on most of the major tributaries that would likely have been coho habitat, resulting in early extirpation from the river.

It is hard to overestimate the importance of loss of large wood to coho as the result of historical logging practices. The trunks and large branches of trees provide cover for fish and interact with high flows to create pools and other habitat features. The streams in the Santa Cruz Mountains and Mendocino Coast contain little of the low-gradient, wide-valley streams that tend to be the most productive habitat for coho salmon. Thus, the role of large wood in these steeper streams was, in all likelihood, essential for providing refuge, particularly from high flows, because there were fewer off-channel habitat refuges. Lack of habitat structure is clearly a major problem facing CCC coho, especially in the winter months when refuges from high flows are needed (e.g., Stillwater Sciences 2008). Even in state parks in the region, which often have 100-year old riparian forests, large in-channel wood remains extremely scarce and is present largely as the result of stream enhancement projects (e.g., Ferguson 2005).

The early logging in most CCC coho watersheds was followed by permanent clearing of much of the land for urban and agricultural use, which continued to degrade water quality, quantity, and habitat for coho salmon and other salmonids (NMFS 2012). Thus, Opperman et al. (2005) found that in the Russian River watershed, the pervasive large-scale changes in land use had resulted in high sedimentation that precludes successful salmon spawning. In this way, many CCC coho streams never had the opportunity to recover from earlier damage because of conversion of watersheds to vineyards, farms, suburbs, and towns. As a result, CCC coho are disappearing rapidly.

Estuarine alteration. All CCC coho streams have either estuaries or lagoons at their mouths, which provide important seasonal rearing habitat. The lagoons can close naturally as the result of the combination of low flows and movement of sand by wave and tidal action, keeping adults out and juveniles in at times when both need to migrate. Historically, high stream flows from winter storms would keep the lagoons open to the sea, at least intermittently. Today, lagoons are often opened artificially to prevent flooding of agricultural fields or houses, and they may drain before juveniles are ready to be carried out to sea. The lagoons are also often polluted with agricultural and suburban waste, and can become too warm and anoxic for coho in summer. In addition, marshes that once lined the lagoons on their upper ends have been converted to agriculture or other uses, reducing cover and food supplies.

Harvest. The ocean commercial fishery for coho salmon in California was halted in 1993 and the ocean sport fishery in 1994 and 1995, despite the fact they are mixed stock fisheries, with many of the fish coming from Oregon hatcheries and streams. Sport fishing is now not allowed in streams as a result of listing of coho under the Endangered Species Act. Small numbers are undoubtedly caught and released in both commercial and sport fisheries targeting other species. However, overall, fisheries seem to be having only a minor impact on coho populations today, and the closure of fisheries has presumably helped to protect the dwindling California populations.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Central California Coast coho salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years, whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	Medium	Most streams without dams, but dams affect Russian River, Lagunitas Creek, Mad River, and a few other streams.
Agriculture	Critical	Irrigation diversions in many streams reduce flows; vineyard expansion is a problem. Illegal and unregulated marijuana cultivation is a major and growing issue, especially during drought.
Grazing	Medium	Chronic stream bank alteration.
Rural /residential development	Medium	Many homes along streams.
Urbanization	High	Many important streams flow though urban areas.
Instream mining	Medium	Gravel mining in Russian River reduces habitat.
Mining	Low	Hardrock mining a minor problem, although mercury-laden water affects some streams.
Transportation	Medium	Roads & railroads create sediment and erosion.
Logging	High	A chronic problem related to roads and other impacts; legacy effects a major issue.
Fire	Low	Can cause siltation of coho streams, loss of shade to cool water.
Estuary alteration	High	Most estuaries and lagoons highly altered with reduced rearing habitat.
Recreation	Low	Boating, rafting probably have low impacts.
Harvest	Medium	Mostly protected but still some harvest, including poaching.
Hatcheries	Medium	Conservation hatcheries important for some populations.
Alien species	Low	Few aliens in coho watersheds.

Effects of Climate Change. Moyle et al. (2013) rated CCC coho salmon as "critically vulnerable" to climate change, indicating it could be the final blow driving it to extinction. This is a result of low populations, stream flow, and highly damaged watersheds. Predicted effects on coho habitat include increases in stream temperatures, variability in flows (including greatly reduced summer flows), and changed ocean conditions that dictate coho growth and survival. Increased frequency of wildfires may increase erosion and reduce shading of already warming streams. These ongoing changes are being superimposed on the other threats to coho, increasing the likelihood of rapid extirpation as time passes. Without dramatic action to protect and enhance habitats, especially at the southern end of their range, CCC coho are likely to continue to decline.

Status Score = 1.3 out of 5.0. Critical Concern. Highly vulnerable to extinction within next 50 years. This score is the result of the precarious state of all populations and the 95% plus decline in abundance from 50-60 years ago. Present trends suggest that most or all populations in small coastal streams will disappear in the next 50 years without increased intervention and protection of watersheds. NMFS (Good et al. 2005, NMFS 2012), DFG (2002), and Swales (2016) agree that coho salmon are in danger of extirpation from the southern end of their range in the near future, and that the condition of CCC coho populations continues to deteriorate. CCC coho are listed as endangered by both state and federal governments (1996). The federal status was reaffirmed in 2016.

Table 2. Metrics for determining the status of CCC coho salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Most populations not self-sustaining in long run.
Estimated adult abundance	2	All populations are small, isolated, and may function independently. Most are <100 in most years.
Intervention dependence	1	All populations require intervention to persist and most have intensive management in place or proposed.
Environmental tolerance	1	Coho are among the most sensitive salmonids to environmental conditions.
Genetic risk	1	Populations small and isolated.
Climate change	1	At southern end of range so exceptionally vulnerable.
Anthropogenic effects	1	1 Critical, 3 High factors.
Average	1.3	9/7.
Certainty (1-4)	4	Well documented.

Management Recommendations: Many of the conservation measures discussed for SONCC coho salmon are important for CCC coho as well. However, given the extreme, largely irreversible alteration of many, if not most, CCC coho watersheds, it is clear that keeping the ESU from extinction will require special, high energy/cost efforts, many of which are underway (NOAA Fisheries 2016).

- Protect the few watersheds that have the potential to support coho in the future, such as Scott and Waddell creeks, Lagunitas Creek, and the Garcia, Noyo, and Gualala rivers. They require protection from further degradation and large-scale restoration efforts.
- Develop and maintain restoration hatcheries where they can be used in conjunction with habitat improvement and evaluation measures. The captive broodstock and rearing program on the Russian River seems to be producing successful results, but it is not certain if coho populations, even in restored habitats, can survive without supplementation. Populations in Scott and Wadell creeks appear to be totally reliant on production from the conservation hatchery. However, more monitoring is needed of genetic and demographic effects on both source and receiving populations.
- Resolve all the complex water allocation issues in the watersheds to make sure adequate

water is left to support coho salmon.

- Work with vineyard owners to reduce the impact of vineyards on coho. Viers et al. (2013) indicate that there are a number of strategies than can work well. Similar actions are needed to work with marijuana growers.
- Focus on Lagunitas Creek as a demonstration stream to publicize the plight of the coho and to demonstrate restoration techniques, such as placement of large woody debris (Ferguson 2005). Spawning coho are already a major public attraction in the lower creek (in Samuel P. Taylor State Park) but more could be done to enhance their numbers and to protect habitat. In particular, housing developments along San Geronimo Creek must be constructed in such as way as to do no damage to the creek or to increase its sediment flow into Lagunitas Creek.
- Provide additional special status and protection to the Santa Cruz County CCC coho, as the southernmost populations of the species. The entire watersheds should be managed with coho salmon as the highest priority.

Other management actions put forward by CDFG (2004) and NMFS (2006, 2012, 2016) could go a long ways towards reversing the trends if properly implemented, but they also will require increased funding, increased interagency cooperation, mobilization of public opinion, and development of an extensive monitoring program. Finding a way to work with the numerous private owners of land along coho streams is critical. Monitoring the populations is a necessity; spawning streams should be identified and populations should be sampled annually.

SOUTHERN OREGON/NORTHERN CALIFORNIA COAST COHO SALMON
Oncorhynchus kisutch

Critical Concern. Status Score = 1.7 out of 5.0. Critically vulnerable to extinction as wild fish within next 50-100 years. There has likely been 95% or more decline in numbers since the 1960s in California.

Description: Spawning adult coho salmon are 55-80 cm FL (35-45 cm FL for jacks) and weigh 3-6 kg. Meristic counts are as follows: 9-12 dorsal fin rays, 12-17 anal fin rays, 13-16 pectoral fin rays, 9-11 pelvic fin rays, 121-148 scales in the lateral line and 11-15 branchiostegal rays on either side of the jaw. Gill rakers are rough and widely spaced, with 12-16 on the lower half of the first arch. Spawning adults are dark green on the head and back, maroon on the sides, and grey to black on the belly. Females are paler than males. Spawning males are characterized by a bright red lateral stripe, hooked jaw, and slightly humped back. Both sexes have small black spots on the back, dorsal fin, and upper lobe of the caudal fin. The adipose fin is grey and finely speckled, while the paired fins lack spots. The gums of the lower jaw are grey, except the upper area at the base of the teeth, which is generally white. Parr have 8-12 narrow parr marks centered along the lateral line and are distinguished by the large sickle-shaped anal fin with a white leading edge, bordered on the inside by a black line. Southern Oregon-Northern California Coast coho salmon (SONCC coho) are an Evolutionary Significant Unit (ESU) that can only be distinguished from other coho ESUs by genetic means.

Taxonomic Relationships: Coho salmon are most closely related to Chinook salmon among the six Pacific salmon species (including the cherry salmon, *O. masou,* of Asia) and have hybridized with them in hatcheries (Moyle 2002). Populations in California are the southernmost for the species. As discussed in Moyle (2002), spawning coho salmon demonstrate strong fidelity to natal streams, thus showing some local differentiation, but there is enough movement of fish between streams so that genetically distinct groups occur only over fairly wide areas, separated by natural features that reduce genetic exchange. In California, Punta Gorda (Humboldt County) is the separation point between California's two coho ESUs, the Southern Oregon-Northern California Coast ESU and the Central California Coast ESU. Punta Gorda is not only a prominent feature that affects local ocean currents but it marks the northern end of a long stretch of steep coast line where the streams are too small and precipitous to support coho salmon. The Mattole River at Punta Gorda is home to the southernmost SONCC coho population.

The genetics of coho salmon in California were not well studied until relatively recently (Bucklin et al. 2007, Gilbert-Horvath et al. 2016). The most recent, detailed genetic study of California coho salmon populations, using microsatellite DNA markers, is that of Gilbert-Horvath et al. (2016) who confirmed the validity of the SONCC and CCC coho ESUs. They also discovered that historical widespread planting of coho salmon from non-natal stocks had minimal influence on the genetic integrity of local populations. These results demonstrated that coho from each stream sampled were distinct, yet more closely related to coho from nearby streams than to those in streams further away. Bucklin et al. (2007, p. 40) concluded the following:

> "Our study implicates population fragmentation, genetic drift, and isolation by distance, owing to very low levels of migration, as the major evolutionary forces shaping genetic

diversity within and among extant California coho populations… [Our] resolution of smaller population units suggests that they are experiencing rapid genetic drift, inbreeding, and the associated deleterious effects of inbreeding depression. Accordingly management and rehabilitation of these populations is needed at much smaller scales than current ESU designations."

Life History: The life history of coho salmon in California was first documented in the classic studies in Waddell Creek by Shapovalov and Taft (1954). Coho life history throughout their range is summarized in Sandercock (1991), while Baker and Reynolds (1986), Moyle (2002) and CDFG (2002, 2004, 2015) reviewed their biology in California. Because of the availability of these detailed reviews, our account will be brief and provide references mainly to studies on SONCC populations. A critical element of their biology and conservation is that coho salmon use at least some part of their spawning streams on a year-round basis (Table 1).

Table 1. Timing of use of different life stages of California coho salmon in natal streams. Modified from CDFG (2002) and S. Ricker, CDFW, pers. comm. 2017. X = major use, x = minor use; each 'x' = ca. 2 weeks.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Adult migration	xx								x	xx	XX	XX
Spawning	XX	xx								x	XX	XX
Incubation	XX	XX	xx							x	XX	XX
Alevin/Fry		xx	XX	XX	XX	x						
Juvenile rearing	XX	XX	XX	XX	XX	XX	XX	XX	XX	XX	XX	XX
Out-migration	x	x	xx	XX	XX	XX	xx					x
Estuary rearing			xx	XX	XX	xx						

Coho salmon in California return to their natal streams to spawn after spending 6-18 months in the ocean. Typically, some fraction of males, called "jacks," return after one growing season in the ocean (at age two years), but most males and virtually all females return after two growing seasons in the ocean (typically age three). The fairly strict three-year life cycle is reflected in numbers of spawners in many streams, which have highs and lows at three-year intervals. However, the number of jacks in proportion to the number of hooknose (three year old) males in a spawning population is determined in part by their differential growth and survival as juveniles under different freshwater conditions (Watters et al. 2003, Koseki and Fleming 2007). Recent studies indicate that juveniles can emigrate as young-of-year, one year olds, or two year olds, indicating more flexibility in life history than previously perceived for California populations (Gallagher et al. 2012).

Spawning migrations begin after increased stream flows in fall and early winter allow the fish to move into coastal rivers. Upstream migration usually occurs when stream flows are either rising or falling. The timing of their return varies considerably, but in general coho salmon return earlier in the season in more northern areas and in larger river systems. In the Klamath River, SONCC coho salmon run between September and late-December, peaking in October-November. Spawning occurs in November and December (USFWS 1979). In the Eel River, SONCC coho run 4-6 weeks later than in the Klamath River; arrival in the upper reaches peaks in November-December. In smaller coastal streams coho generally return during mid-November through mid-January. In some years, spawning can occur as late as March, especially if stream flows are low or access is limited because of drought. In smaller coastal streams (such as Redwood Creek and Mattole River), the peak coho runs are in late October-November, or are determined by the first rain event afterward, which increases flow sufficiently to break bars at the mouth of estuaries, permitting access to the stream (M. Sparkman, CDFW, pers. comm. 2017). Coho salmon migrate up and spawn mainly in low-gradient streams that flow directly into the ocean or in tributaries of large rivers.

Females choose redd sites where gravel is mixed in size and sufficiently coarse so that it is easy to move by digging and facilitates subsurface flow around the buried embryos. The best redd sites are often at the head of a riffle, just below a pool, where the water changes from a smooth to a turbulent flow, is deep enough to cover the female when she is digging (ca. 20-75 cm), and typically has high intragravel flow. Each female builds a series of redds, moving upstream as she does so, and deposits a few hundred eggs in each. A dominant male accompanies a female during spawning, but one or more subordinate males and jacks also may engage in spawning. Spawning can take about a week to complete, with females depositing 1,400-7,000 eggs; bigger females produce more eggs. Both males and females die after spawning, although the female may guard her redd for up to two weeks (Hassler 1987).

Embryos hatch after 8-12 weeks of incubation, the time depending on both temperature (colder temperatures increase incubation time) and on inherited adaptations to local conditions. Hatchlings (alevins) remain in the gravel for 4-10 weeks, until their yolk sacs have been absorbed. Under optimum conditions, mortality during this period may be as low as 10 percent; but under high scouring flows or heavy siltation, mortality can reach 100 percent. Upon emerging, fry (30-35 mm TL) seek out shallow water, usually along stream margins. In the Klamath River watershed, emergence of fry starts in mid-February and peaks in March and early April (May in the Shasta River), although apparently fry have been found into July (CDFW, unpubl. data, C. Bean, CDFW, pers. comm. 2017). After moving into shallow water, fry form loose aggregations, but as they grow bigger (50-60 mm TL), most parr set up feeding territories. Behavior of parr, however, shows considerable variation (Nielsen 1992a, b). In smaller streams, as parr continue to grow they move into increasingly deeper water until by mid-summer, they are in the deepest pools available, often swimming in small shoals. If temperatures become high enough to be stressful, individuals will seek cool water refuges, usually where cooler subsurface flows upwell through the gravel. In the Klamath River, SONCC juveniles seek cool water refuges at the mouths of tributary streams in early summer but these areas are usually too warm or crowded with other salmonids to support them by late summer (NRC 2004). At least some of these fish, however, may migrate upstream into coldwater tributaries if access is present. Growth rates slow down at this stage, possibly due to lack of food or because the fish reduce feeding as a result of warmer temperatures (see Box 1).

During December-February, winter rains result in increased stream flows and by March, following peak flows, fish again feed heavily on insects and crustaceans and grow rapidly. During winter, refuges from high, turbid flows are required for survival. Typically, these refuges are side channels, complex masses of large woody debris, and small, clear tributaries. A variable, but substantial, portion of coho parr emigrate to stream-estuary ecotones to exploit the rapid growth potential of these habitats (Wallace et al. 2015). Towards the end of March and the beginning of April, juvenile coho begin to migrate downstream and into the ocean, though a small fraction of juvenile coho smolts may emigrate to the ocean in December-February based on limited occurrences in screw trap data (S. Ricker, CDFW, pers. comm. 2017). Outmigration in California streams typically peaks in April if conditions are favorable (B. Spence, NMFS, pers. comm. 2008) although Shapovalov and Taft (1954) found that coho emigration from Waddell Creek peaked in mid-May. Migratory behavior is stimulated by a variety of factors: rising or falling stream flows, size of fish, day length, water temperature, food densities, and dissolved oxygen levels and available rearing habitat. At this point, outmigrants are typically about one year old and are 10-13 cm FL. Larger fish (ca. 20 cm FL) have usually spent two years rearing in the stream. In Prairie Creek (Humboldt Co.), over 20% of emigrating juvenile SONCC coho are two years old (Bell and Duffy 2007), though this proportion can vary widely from year to year. Ransom (2007) found the portion of age-2+ coho to range from 0-30% in Prairie Creek. According to Brakensiek and Hankin (2007), age-2 fish in Prairie Creek were smaller in length during their first year in freshwater than other fish in the same cohort and larger the next year than fish of the subsequent year class. Large numbers of age 2+ coho were observed in 2015 in tributaries to Humboldt Bay, possibly as a result of poor rearing conditions from drought (M. Wallace, CDFW, pers. comm. 2017). Some juveniles also emigrate from streams as young-of-year (Gallagher et al. 2012), and move in small schools of about 10-50 individuals. Parr marks are still prominent in the early migrants, but the later migrants are silvery, having transformed into smolts.

All juveniles leaving streams have to spend some time in estuaries, a habitat that has been underappreciated for its importance in California. Wallace et al. (2015) found estuaries in Humboldt Bay, including those in which no coho spawned, to be a major rearing habitat for juvenile coho. They identified three life history patterns:

- Juveniles aged 1+ that had reared in streams and largely migrated through estuaries in spring.
- Yearling fish that moved in during the first high-flow event of the fall and reared in non-natal estuaries and off-channel habitat during winter and spring.
- Young of year that moved downstream in spring and reared in the main channels of the larger estuaries in summer and fall.

The importance of these findings is that estuarine habitat was used by SONCC coho year-round, and that different ages and sizes of coho used different parts of the estuarine environment in different ways. Juveniles using the estuarine habitats were generally larger at age than those that remained in streams to rear, and fed largely on amphipods, small crustaceans, and shrimp (R. Taylor, R. Taylor Associates, pers. comm. 2016). Young-of-the-year (YOY) found in stream-estuary ecotones exhibited growth rates as high as .7 mm/day, while yearling fish could grow as much as 1.0 mm/day (Wallace et al. 2015). The increased food availability and temperatures, coupled with lower bioenergetic demands in low-gradient habitat, most likely increase growth rates (Wallace et al. 2015).

After entering the ocean, immature salmon initially remain in inshore waters close to parent streams. They gradually move northward, staying over the continental shelf. Coho salmon can range widely in the north Pacific, but movements of California fish are poorly known. Most coho caught off California in ocean fisheries were reared in coastal Oregon streams (natural and hatcheries). In 1990, for instance, 112,600 coho were caught in commercial and recreational ocean fisheries, which may greatly exceed the present production capability of California populations alone (A. Baracco, CDFW, pers. comm. 1994). Oceanic coho tend to school together, with fish from different regions found mixing in the same general areas. Adult coho salmon are primarily piscivores, but shrimp, crabs, and other pelagic invertebrates can be important food in some areas.

Habitat Requirements: This section is based on Moyle (2002) and CDFG (2002, 2004). For a useful tabular summary of coho habitat requirements see CDFG (2004, p. 222). In general, coho salmon respond to multiple habitat cues at any given time. Bioenergetics is the key to understanding why coho juveniles choose a particular combination of habitat characteristics and how habitat affects growth and survival.

Adult coho salmon move upstream in response changes in stream flows caused by fall storms, especially in small streams when water temperatures < 16°C. However, their presence on occasion in the Lower Klamath River as early as mid-September when flows are low and temperatures are high suggests that other cues are important as well. For example, high turbidity may delay migration even if other conditions are optimal.

Spawning sites are typically at the heads of riffles or tails of pools where there are beds of loose gravel (< 15 cm average diameter) and cover nearby, such as a deep pool or undercut bank or log. Coho salmon redds can be excavated in substrates composed of up to 20 percent fine sediment, but spawning success and fry survival generally are best in very clean gravel (<5 percent fines). Spawning depths are 10-54 cm, with water velocities of 0.2-0.8 m sec-1. Optimal temperatures for development of embryos in the gravel are 4.4-13.3°C, although eggs and alevins can be found in 4.4-21.0°C water. Dissolved oxygen levels should be above 8 mg l^{-1} for eggs and above 4 mg l^{-1} for juveniles.

Juveniles are generally most abundant where there are deep (0.5 to 1+ m), well-shaded pools with plenty of overhead cover; highest densities are typically associated with instream cover such as undercut banks or logs and other woody debris in pools or runs. Optimal summer habitat seems to be pools containing rootwads and boulders in heavily shaded sections of stream, although warmer, more open areas may be used if food is abundant.

Juveniles can move to neighboring streams to rear. Non-natal coho rearing has been documented in the Smith, Klamath, Eel, Russian, Salt, and Elk rivers, as well as Redwood, Freshwater, Jacoby, and Salmon Creeks in California (Wallace et al. 2015). Juveniles have been observed rearing in winter and in spring in watersheds where there is a lack of documented coho spawning, raising the point that spatial habitat usage can vary broadly in a region. For example, juvenile coho were documented in Wood Creek, Martin Slough, and Rocky Gulch (streams without spawning coho), indicating that juveniles either migrated over flooded pasturelands or entered Humboldt Bay to reach suitable rearing habitat in adjacent watersheds (Wallace et al. 2015). It is possible that spawning adults were present but not observed during sampling.

In winter, refuge habitat, especially large wood and complex side channels and off-channel habitat such as alcoves are needed to protect juveniles from being washed away by high flow events (Gallagher et al. 2012, and references therein). Bell (2001) found the habitat with highest site fidelity and survival of juveniles consisted of deep mainstem pools with off-channel refuge, such as alcoves, nearby. Preferred water velocities for juveniles are .09-.46 m/sec, depending on habitat (Gallagher et al. 2012). High turbidity is detrimental to emergence, feeding and growth of young coho.

Juvenile coho require cold water during rearing, generally regarded as less than 18-20°C. In the Mattole River watershed, for example, SONCC coho were absent from streams that had maximum weekly maximum temperatures (MWMT) exceeding 18°C, but were consistently found in streams in which temperatures did not exceed MWMT of 16.3°C (Welsh et al. 2001). Stenhouse et al. (2012), based on a literature review, concluded that:

- Optimal temperatures for growth, swimming, and disease resistance were 10-15.5°C.
- Suboptimal temperatures were 15.5-20.3°C.

- Temperatures greater than 20.3°C were detrimental or lethal.

In their review, Stenhouse et al. (2012) discounted laboratory studies that showed rapid growth of coho at temperatures higher than 15.5°C, although juveniles have been shown to grow under stream temperatures regularly exceeding 24.5°C, up to 29°C, when conditions are appropriate in the wild (Bisson et al. 1988; Moyle 2002). In the temperature gradient of the Shasta River, where food is essentially unlimited, caged juvenile coho grew much faster at warmer temperatures (MWMTs: 19.1-21.8 °C) than at cold temperatures (MWMTs: 15-16°C) (R. Lusardi, unpubl. data). In contrast, Gallagher et al. (2012) found lowest growth rates in coho in fairly typical oligotrophic forest streams in Mendocino County, when summer temperatures were warmest. In general, optimal conditions for coho (and other fishes) are determined by bioenergetic considerations, not just temperature (Box 1) and are context and watershed specific.

Box 1. Bioenergetics: a key to salmon survival

In the laboratory, most fishes have an 'optimal' temperature range for growth, in which the conversion rate of food to fish flesh is most efficient. For juvenile coho, this range appears to be 12-14°C. The problem is, of course, that stream environments are rarely constant and juvenile coho are often found at higher temperatures. In tributaries to the Mattole River, juvenile SONCC coho were absent from streams where mean weekly maximum temperature exceeded 18°C (Welsh et al. 2001). This suggests that Mattole River fish are persisting mainly where temperatures are close to optimal. Similar observations have been made for SONCC coho in Redwood Creek (Madej et al. 2005). In contrast, Bisson et al. (1988) observed juvenile coho rearing in a Washington stream where maximum weekly temperatures regularly exceeded 20°C and daily maxima sometimes reach 29°C for short periods. This was hypothesized to be possible because (1) coho had essentially unlimited food, (2) there were no competitors or predators present, (3) night-time temperature were cool (often around 12°C) and (4) thermal refuges may have been present (springs, etc.), although there was little evidence of refuge use.

The explanation for this becomes clear if survival and growth of coho is put in terms of an energy budget. Basically, juvenile coho will grow if they ingest more energy than they consume, through activities such as searching for food, avoiding predators, or even resting. Individuals eventually die if they ingest less energy than they use during daily activity. Part of that energetic cost can be increased metabolic rates and stress caused by temperatures higher than optimum. As temperatures increase, so do metabolic rates. In the studies by Bisson et al. (1988), bioenergetic conditions (most likely high availability of food) were adequate to sustain coho at nearly lethal temperatures. A similar phenomenon was documented on the Shasta River in northern California. Here, researchers found that juvenile SONCC coho grew at faster rates under warmer water conditions (~15°-16° C vs. ~19°-22° C), primarily because food resources were extremely abundant (R. Lusardi, pers. comm. 2016). In Mattole River tributaries, where food was not abundant and predators and competitors were common, even moderately high temperatures may be lethal if experienced on a regular basis. The energetic costs of living at higher temperatures are simply more than the fish can sustain.

Distribution:

General. Coho salmon are widely distributed in northern temperate latitudes. In North America, they spawn in coastal streams from California to Alaska. In Asia, they range from northern Japan to the Anadyr River in Russia. In California, they live in streams from Del Norte County on the Oregon border to Santa Cruz County. SONCC coho salmon are found from Cape Blanco in Oregon south to the Mattole River, just north of Punta Gorda. Historically, SONCC coho salmon occupied numerous coastal basins where high quality habitat was located in their lower portions and three large basins where high quality habitat was located both in lower tributaries and in headwaters, while the middle portions of the basins provided little habitat (Williams et al. 2006). In Oregon, south of Cape Blanco, the Rogue River is apparently the only river with a persistent run of coho, although a few coho are observed on occasion in the Chetco and Winchuk rivers and other smaller streams. Most SONCC coho are therefore in California.

NMFS (Williams et al 2006) divided the California populations into five diversity strata, each representing environmentally and ecologically similar regions: Klamath River, Trinity River, Eel River, Central Coastal, and Southern Coastal strata. Among these six strata, the SONCC historically had 14 functionally independent populations, 11 potentially independent populations, and 6 dependent populations (Williams et al 2006). The largest remaining SONCC coho populations in California are in the Klamath, Trinity, Mad, Humboldt Bay, Eel and Mattole drainages, with additional populations in some smaller coastal streams.

Garwood (2012) updated the distribution of California coho salmon of Brown and Moyle (1991) and Brown et al. (1994) through 2004. In sum, 540 historical SONCC coho streams were identified in California, which was a 40% increase from the 325 streams identified by Brown and Moyle (1991). Presence/absence observations from surveys for a subset of these streams found 31% to 62% were still used by coho each year (Garwood 2012). The following is a description of their use of major watersheds in within California.

Smith River and Del Norte County streams. In the Smith River and smaller streams in the region, 53 potential coho salmon streams were identified; 28 were sampled with juveniles detected in 61% (Garwood 2012). However, Mill Creek, tributary to the Smith River estuary, appears to be the principal stream supporting spawners, averaging 54/year (CDFW 2015).

Klamath River. Historically, coho were found throughout much of the ~ 4000 km^2 watershed, spawning and rearing primarily in coldwater tributaries. In the mainstem Klamath, they presumably were present roughly up to the mouth of Spencer Creek, about 350 river km upstream and used all permanent tributaries for which there was access. They were found throughout the watersheds of two major tributaries, the Scott and Shasta Rivers. At the present time, coho use the mainstem Klamath up to Iron Gate Dam, where the Iron Gate Fish Hatchery is located. In the Shasta River, Dwinnell Dam blocks upstream access and coho are absent from several major tributaries due to a lack of instream flow during irrigation season (Little Shasta River and Yreka Creek). Below the dam, the principal tributaries suitable for coho rearing during the summer months are Big Springs Creek and Parks Creek. In both tributaries, but Parks Creek in particular, rearing habitat occurs adjacent to cold water spring inflow. The Scott River watershed is a snowmelt driven system, and the mainstem goes dry during the summer months due to irrigation demands. However, tributaries adjacent to the Marble Mountain Wilderness, located on the west side of the watershed, provide summertime refugia (C. Bean, CDFW, pers. comm. 2017).

In the Trinity River and its forks, coho were once distributed well upstream of Lewiston

Dam. Below the dam (about 175 km upstream from the mouth on the Klamath River), they were present in most tributary streams, as well as in the mainstem up to the Trinity Fish Hatchery. Garwood (2102) estimated 184 streams in the Klamath basin below Iron Gate and Trinity dams were historically available to coho salmon spawning and rearing. In 2001-2003, 63 of these streams were sampled (not randomly selected) and coho were found in 65% of them. In the Trinity River, upwards of 90% of the coho are of hatchery origin, so the recent distribution may reflect hatchery production rather than use of the streams by naturally spawned fish (Spence et al. 2005).

Redwood Creek. Redwood Creek and its major tributary Prairie Creek were historically important coho streams, as were their 30 tributaries. Today coho are largely confined to the lower 20 km of the 90 km-long Redwood Creek and tributaries to the lower 20 km, including Prairie Creek, as a result of elevated summer water temperatures higher upstream (Madej et al. 2005). Prairie Creek, however, is an important rearing stream for coho and other salmonids because of its relatively undisturbed nature inside State Park boundaries with intact old-growth forests (Sparkman et al. 2015).

Mad River and Humboldt County streams. The Mad River historically supported coho salmon in its lower reaches, as did the smaller coastal streams in the coastal fog belt, where air and water temperatures were consistently cool. Coho apparently ascended the Mad River to either Bug or Wilson creeks, just below a relatively steep area on the main river ("the roughs"), a distance of about 80 km. They have been reported in recent years in some of the larger tributaries (e.g., Lindsay Creek). There are 62 potential coho streams, of which 29 were sampled in 2001-03; 62% contained coho (Garwood 2012). Freshwater Creek and Elk River and their tributaries seem to be particularly important coho streams in the region.

Eel River. In the 9500 km^2 Eel River system, coho formerly ascended the mainstem, the South, Middle, and North forks, 69 tributaries of the South Fork Eel, and the Van Duzen River. They are currently absent from the Middle and North fork drainages and from over 40% of the tributaries in which they once existed (Garwood 2012).

Mattole River. The Mattole River (watershed area, 787 km^2) and its 21 larger tributaries presumably all once supported coho salmon but Welsh et al. (2001) found them in 9 of 21 tributaries and largely absent from the mainstem. Garwood (2012) identified 44 potential coho streams in the watershed, of which 27 were sampled in 2001-03; coho were detected in 63%.

As the above summary indicates, SONCC coho salmon were and still are widely distributed in coastal streams from the Oregon border to Punta Gorda, and fairly far inland in the Klamath and Eel rivers. However, the long-term trend has been downward in the number of wild populations, with individual populations becoming more isolated and the overall distribution becoming fragmented. The spawning population in the Klamath basin is increasingly dominated by hatchery fish (Quinones et al. 2013). Of 541 coastal and tributary streams that historically held SONCC coho salmon, coho have been lost from at least 38% of them, assuming the study of Garwood (2012) involves unbiased sampling of the streams in 2001-03. Spence et al. (2005) found that the number of California streams containing SONCC coho salmon probably changed little in the period 1987-2001; over the 15 yr period occupancy rate varied from 55 % to 67% with no trends. The effects of the on-going drought have yet to be fully evaluated but the effects are likely to be strongly negative, pushing numbers even lower.

Trends in Abundance:

Overview. Very rough estimates indicate that the number of coho salmon returning to streams in the SONCC region 50-60 years ago was somewhere between 100,000 to 300,000 spawners (or more) per year (Brown et al. 1994), using several hundred streams for spawning and rearing. This suggests a long-term decline in excess of 95% in population size and a decline in number of streams used annually on the order of 40-50%.

Since the statewide assessment of coho status in the 1980s (Brown et al. 1994), SONCC coho salmon have remained in low numbers and have probably declined further. Monitoring is inadequate to say that populations have definitely decreased, but they certainly have not increased significantly. According to CDFG (2004) "...declines appear to have occurred prior to the late 1980s and the data do not support a significant decline in the distribution between the late 1980s and the present (p. 2.2)". Nevertheless, they recognize that the severe declines in habitat quality indicate that "…coho salmon populations …of this ESU will likely become endangered in the foreseeable future in the absence of protection and management required by the CESA (p 2.2)." Similarly, Bucklin et al. (2007) suggested that most SONCC coho populations are in decline from which recovery will be difficult.

Garwood (2012) summarized survey results that indicated there were 542 streams in California with historical use by SONCC coho; non-random sampling of a subset of these streams indicates somewhere between 30 and 60% of the streams supported coho in each year in 2001-03. How this relates to populations of naturally produced coho at present is not certain, especially given the effects of severe drought during 2012-2016. Reductions in streamflow and higher water temperatures in many habitats due to the ongoing drought and other anthropogenic effects have likely decreased juvenile over-summer survival. These drought impacts, coupled with poor ocean conditions, likely depressed SONCC coho populations, but the magnitude of these effects will likely not be known for at least another few years when three-year old adults return to spawn.

Historical abundance. Historical estimates of statewide coho salmon abundance were essentially best guesses made by fisheries managers, based on limited catch statistics, hatchery records, and personal observations of runs in various streams. Maximum estimates for the number of coho spawning in the state in the 1940s range from 200,000-500,000 to close to 1 million (Calif. Advisory Committee on Salmon and Steelhead Trout 1988). Coho numbers held at about 100,000 spawners statewide in the 1960s (California Advisory Committee on Salmon and Steelhead Trout 1988), with 40,000 in the Eel River alone (U.S. Heritage Conservation and Recreation Service 1980), and then dropped to a statewide average of around 33,500 during the 1980s (Brown et al. 1994). The reliability of these estimates is uncertain, and so they must be viewed only as "order-of magnitude" approximations, although Brown et al. (1994) attempted to estimate abundance on a stream-by-stream basis. Coho salmon in this ESU, including hatchery stocks, presently seem to be less than 6 percent of their abundance during the 1940s, with probably at least 70 percent decline in numbers since the 1960s. Brown et al. (1994) estimated that the total number of adult coho salmon entering California streams in 1988-90 averaged about 31,000 fish per year, with SONCC coho making up about 80% of the total. However, fish of suspected hatchery origin made up 57 percent of the state total. The hatchery stocks have in their ancestry fish from other river systems and often from outside California, although extra-basin stocks rarely seem to establish permanent populations or contribute to the wild populations (Bucklin et al. 2007).

Klamath River. Klamath River populations presently are largely maintained by Iron

Gate hatchery production. About 80% of returning fish are of hatchery origin and a small percentage of these originate from the Trinity River Hatchery, as well as from hatcheries in Oregon and Washington (Chesney 2007). Hatchery returns are highly variable among years (Chesney 2007). At Iron Gate Hatchery, for example, only 322 coho returned in 2006-2007, and less 100 in 2015 and 2016, although returns of over 2,500 adults have occurred in the past (1,605 fish average (D. Bean, CDFW, pers. comm. 2017). Historical annual total spawning escapements for the Klamath River system have been estimated at 15,400-20,000 fish, with 8,000 for the Trinity River (USFWS 1979). Numbers are presumably much less today, even with hatchery production. Recent returns from the Upper Klamath population unit have dwindled to a few hundred to less than 55 fish to the Iron Gate Hatchery and Bogus Creek, a major tributary, in 2009 and sometimes fall below the high risk abundance level of 425 individuals (CDFW 2014). The Shasta presumably once supported runs of several thousand fish each year, based on the presence of high-quality coldwater habitat in upstream areas. Dwinnell Dam blocks access to some of this habitat, while other habitat has been made unsuitable by agricultural diversions and warm return flows (Jeffres and Moyle 2012). In 2001, CDFG started counting coho salmon coming through a weir on the lower river. Despite considerable difficulties in operating the weir, especially during high water, the counts suggest that annual runs are now between 40 and 400 fish per year (Walsh and Hampton 2007, Swales *in prep.*). Improvements to the river's coldwater flows are apparently responsible for increases in survival of juveniles, even during drought years, although many of the returning adults are of hatchery origin (Swales 2016). However, very low numbers of returning coho in the Shasta River in 2014-2015 raise a serious cause for concern (NOAA Fisheries 2016) and highlight the need for research into how the ongoing drought may affect spring-dominated systems and ocean survival.

The Scott River is run-off dominated tributary just south of the Shasta River. Adult returns to this river, monitored since 2007, have been highly variable, from 63 fish in 2008 to 2750 fish in 2013 (NOAA Fisheries 2014). Documentation of hatchery origin coho in the Scott River is very rare (2 to 5 fish over all sampling seasons); in general the returning adults are of wild origin (CDFW unpubl. data).

Trinity River. In the Trinity River over 90% of returning coho are of hatchery origin, indicating natural spawning of wild-origin fish is depressed (Spence et al. 2005). Hatchery returns and 'wild' populations fluctuate more or less in synchrony. The Trinity River Hatchery releases over 500,000 smolts each year, with unknown, but presumably detrimental (density-dependent) effects on wild-produced fish. However, the number of natural origin fish returning to the Trinity River seems to have increased somewhat in recent years (2012, 2013) (Swales 2016). Total numbers of adults returning to the Trinity River watershed are estimated be between 5,000 and 39,000 fish, with considerable year to year variability; the number of adults that are not of hatchery origin is presumably between 500 and 3900 each year, usually on the lower end of this range.

Eel River. Probably the largest concentration of wild SONCC coho (with little or no hatchery influence) is in the South Fork of the Eel River, which has been estimated in recent years to have adult runs ranging between 1,000 and 2,000 fish. The latter number is from 2010/11 (CDFW 2015). Counts at Benbow Dam from 1938-1976, showed numbers to be highly variable but in long-term decline: 7,000-25,000/year in 1940s, 2000-11,000/year in 1950s, 1,200-14,000/year in 1960s, and 500-2,000/year in 1970s. They are apparently now in only small numbers (< 200) in historical habitats in other parts of the Eel watershed, including the Van Duzen River, Mainstem Eel and tributary creeks (NOAA Fisheries 2014); they are absent from most of the watershed.

Mattole River. According to NOAA Fisheries (2014, 29-6): "CDFG estimated an average run size of 8,000 coho salmon in the mid-to late 1950s, and in 1960 the United States Fish and Wildlife Service (USFWS) estimated an average run size of 2,000 coho salmon and a potential population abundance of 20,000 coho salmon based on habitat characteristics at the time." In 2000, Juvenile coho salmon were noted by Welsh et al. (2001) to be present in just 9 of 21 tributaries of the Mattole River and scarce in the mainstem river. Surveys of adults and redds by the Mattole Salmon Group since 1995 have shown a general decline in spawners since the surveys were initiated. In the winter of 2012-13, there were an estimated 39 coho redds in the entire watershed (Ricker et al. 2014), presumably representing a run of < 100 fish.

Other populations. Populations in other coastal watersheds (such the Smith River and Redwood Creek) are highly variable from year to year but represent important source populations, especially in Prairie Creek (Redwood Creek, Table 2).

Table 2. Redwood Creek redd estimates and DIDSON camera estimates. From Ricker, S., Lindke, K., and C. Anderson 2014 Fig. 9, pg. 18.

Redwood Creek adults	Chinook salmon		Coho salmon		Steelhead	
Year	Redd estimate	DIDSON estimate	Redd estimate	DIDSON estimate	Redd estimate	DIDSON estimate
2009-10	520	2438	382	373	436	560
2010-11	1566	*768*	1148	*322*	172	695
2011-12	1732	1455	1080	803	100	267
2012-13	1880	3401	810	747	810	1331
2013-14	1926	3487	1410	2175	164	787
2014-15	2126	x	594	x	670	x
2015-16	1480	*1839*	412	*144*	566	*203*
Delisting Target		3450		3000		6000

Overall. Presently, there are likely less than 5,000 wild coho salmon (no hatchery influence) spawning in the SONCC region of California each year, but this number should vary with cohort and with variation in survival in both stream and ocean. In their 2016 status review, NOAA Fisheries found that of the seven major independent populations within the ESU, none supported viable populations according to the recovery plan (NOAA Fisheries 2016). In fact, only the populations of the Little, Klamath, Scott, Upper Trinity, South Fork Eel, and Humboldt Bay watersheds have moderate extinction risks based on spawner density estimates (NOAA Fisheries 2016). Many of these fish are in populations of less than 100 individuals, below the minimum population size required to preserve stock genetic diversity and to buffer populations from natural environmental disasters. The small populations also present major difficulties for conducting a census of fish numbers; a large effort is required to obtain estimates that are still of marginal reliability (Gallagher and Wright 2007, NOAA Fisheries 2014).

There is every reason to think that SONCC coho populations are not secure, even though hard data on numbers, especially in recent years, are hard to come by in most watersheds. Most available data comes from redd counts and some spawner surveys that mostly cover the last fifteen years and are not comprehensive (NOAA Fisheries 2016). Recent efforts to standardize methods among survey crews has provided some continuity and consistency in sampling across agencies, tribes, and other cooperators, which should help to improve reliability of abundance estimates in the future and coordinate recovery efforts (J. Garwood, CDFW, pers. comm. 2016). What evidence there is makes it seems likely that in most years, total SONCC adult coho spawners in California are somewhere between 3,000 and 10,000 wild fish, excluding the hatchery-dominated numbers from the Trinity River. The actual numbers are imprecise, but SONCC coho salmon are certainly at a small fraction of historical numbers, are likely decreasing, and are highly vulnerable to continued environmental change. To make matters worse, these fish are mostly in small isolated populations that show evidence of genetic and demographic problems that are likely to lead to extinction (Bucklin et al. 2007).

Factors Affecting Status: The general reasons for the decline of coho salmon in California are many and well known (Brown et al. 1994, NOAA Fisheries 2014); they include (1) poor land-use practices that degrade streams, especially those related to logging and agriculture, (2) dams and diversions, (3) urbanization, and (4) overharvest in combination with natural cycles of floods and droughts and ocean productivity, and, in addition, climate change. NOAA Fisheries (2014) identified multiple factors limiting SONCC coho populations, covering virtually every means by which humans damage streams and fish populations. CDFG (2002, 2004) provided extensive discussion of these factors and how they affect coho populations. Although all salmon are affected by the above factors, their effects on coho are likely to be particularly severe because virtually all females are three years old. Therefore, a major flood or severe drought, in conjunction with one of the above human-caused factors, can eliminate one or more year classes from a stream. There is good evidence that this has already happened repeatedly in coastal drainages, where the decline of coho is linked to poor stream and watershed management and legacy impacts from logging practices. This problem has been exacerbated by the rapid growth of marijuana cultivation in recent years, which removes water from small coho streams, damages stream channels, and introduces fertilizers and other pollutants. In addition, the long-term impact of dams of denying access to upstream areas and altering downstream habitats is still a problem. Clearly, existing regulatory mechanisms, such as forest practice rules, water agreements, streambed alteration agreements, and state and federal Endangered Species Act take prohibitions have been inadequate to protect SONCC coho. The relationship of people with landscapes containing coho salmon needs to be changed on a large scale to prevent extirpation from California.

Here we briefly discuss: (1) dams, (2) diversions, (3) logging, (4) grazing, (5) agriculture, (6) mining, (7) estuarine alteration, (8) alien species, (9) harvest, and (10) hatcheries.

Dams. Dams have two major general impacts on coho salmon: they deny or reduce access to upstream areas and they alter habitat below the dams. In the SONCC area, there are major dams on the Rogue (Oregon), Klamath, Shasta, Trinity, and Eel Rivers (CDFG 2004). All of the California dams have cut off access to upstream spawning and rearing habitat, which CDFG (2002) estimates to be 311 km of stream, mostly (175 km) above Lewiston Dam on the Trinity River alone. Likewise, Dwinnell Dam on the Shasta River cuts off access to approximately 22% of the historical coldwater habitat upstream and the reservoir prevents cold water from reaching downstream areas where it is critically needed (NMFS 2007). As in the Shasta River, rivers downstream of dams are typically unsuitable for coho spawning and rearing because of reduced flows, altered flow regimes, increased temperatures, embedded gravel, and other problems. The main function of the mainstem rivers is reduced to providing passage for upstream and downstream migrating fish, although some rearing of juveniles may occur where there are 'cool pools' of upwelling or tributary water. Decreased habitat connectivity in the form of dams, levees, diversions, tide gates, and other obstructions can reduce winter growth and survival of coho juveniles and may potentially reduce life history diversity and expression (Wallace et al. 2015).

There are literally hundreds of diversions on SONCC coho streams, which cumulatively reduce flows and increase temperatures. If diverted water is used for flood irrigation of pasture, much of it can return to the river at high temperatures and often polluted with nutrients. The problem with diversions is particularly acute during summer when flows are naturally low and temperatures are stressful to salmonids, especially in dry years. In the Shasta River, the combined effects of diversions are to turn what was once the coldest (in summer) large tributary to Klamath River into one that is largely too warm for most salmonids. However, acquisition of a key parcel that includes a portion of Big Springs Creek by The Nature Conservancy (TNC) in 2009, and consequent reductions of surface water diversion on their property, has resulted in an increase in available coldwater habitat (Jeffres 2009). Other tributaries (e.g., Little Shasta River) dry up in their lower reaches from diversions. Conditions in the Scott River are similar in that much of the water is diverted for agriculture and pasture; when irrigation season begins in the summer, streamflows drop and water quality becomes unsuitable for juvenile coho salmon (NMFS 2007). However, some tributaries upstream of diversions still support coho populations. During recent years, the mainstem goes dry because of diversions and groundwater pumping, as do the lower reaches of most of its tributaries.

Logging. Logging is one of the principal uses of both public and private land in the range of SONCC coho. It is most likely the single largest cause of coho decline overall because it began in the 19th century with the logging of key coho watersheds at lower elevations and then gradually moved upslope and inland. In SONCC coho streams, there were essentially two waves of damaging logging. The first involved logging the original old-growth forests, with complete disregard for watershed and salmon effects. Streams were largely regarded as convenient ways to float or drag logs to accessible locations (often behind a mill dam) so splash dams and log drives down larger rivers were commonplace. These dams were temporary dams constructed to back up water to float logs and then to wash them downstream when a dam was deliberately breached. The damming was usually preceded by channel clearing to allow unobstructed washing of logs to the mills, usually on or near the estuaries. This practice essentially scoured coho habitat and deprived fish of essential cover in the form of fallen trees (large woody debris). The second wave of damage was the result of post-World War II logging practices that reversed partial stream recovery from past damage. Unrestricted logging using trucks and other heavy equipment caused massive erosion and removed riparian vegetation and woody debris from channels. Over time, stream temperatures increased, pools filled with silt and gravel, stream channels became altered, and water quality declined. SONCC coho streams still suffer from this double legacy of harmful logging; streams are still suffering and the coho are disappearing from them as a consequence. For many years, fisheries agencies continued the practice of "debris" removal on the assumption that debris jams prevented upstream migrations of spawning fish. These legacy effects still compromise the ability of many streams to support large numbers of coho salmon.

While logging today is much more regulated than in the past (at least since the 1970s), it is still having multiple, cumulative effects on coho streams. Removal of trees reduces shade, increases water temperatures, and reduces the amount of large woody debris that falls into the streams, which provide critical habitat for rearing salmonids. An even more detrimental effect of logging is the creation of thousands of miles of temporary roads, which create large-scale instability of soils on the steep slopes that characterize coastal northern California. The result has been the erosion of huge quantities of sediment into streams, burying coho habitat. Sediment deposition and channel alteration was particularly severe as the result of the large floods of 1955 and 1964, from which the SONCC salmon basins have still not recovered. Forest practice rules are now much more stringent and restoration projects (eliminating roads etc.) are common, but the continued low populations of SONCC coho indicate that the rules (and enforcement) are still not strong enough to make up for past transgressions, nor are habitat restoration projects on a large enough scale.

Many of the streams containing SONCC coho salmon are impaired under the Clean Water Act because of high sediment loads, although low dissolved oxygen, and high water temperatures and nutrients (e.g., in the Klamath River) may also lead to impaired status. Many of the streams have Total Maximum Daily Load standards that are supposed to be met under section 303(d) of the Clean Water Act, but rarely are. High sediment loads in streams is a common legacy of past logging, road building, and other activities in SONCC coho streams.

Grazing. Grazing practices have had less of an impact on SONCC coho than on more southern populations, but are nevertheless a factor in preventing recovery. Many areas that were historically forested have been turned into pasture or grazing lands, so water flowing into the streams tends to be warmer and flashier in flow and there is less wood available to create cover for the fish. In the Shasta River, especially Big Springs Creek, cattle not only historically trampled banks but also grazed on aquatic plants in the water itself, especially in winter when the spring water was warmer than the air. This problem was largely alleviated after TNC acquired Big Springs Ranch and fenced out cattle. See estuarine alteration for a discussion of conversion of estuarine marshland into pasture for dairy cows.

Agriculture. Historically, agriculture, aside from grazing and pasture, had a minor influence on SONCC coho populations because most streams flowed through forested lands, although diversion of water for agriculture from the Klamath Basin in Oregon may have had indirect effects on coho through decreasing water quantity and quality. In recent years, marijuana cultivation on private and public forest lands has increased dramatically, resulting in the alteration of coho rearing streams, as well as their dewatering for irrigation (Bauer et al. 2015). While much of this activity is illegal or quasi-legal, its rapid increase in recent years has made enforcement of water rights and stream alteration rules overwhelmingly difficult. Impacts on coho populations are not known but can only be harmful to already stressed populations.

Mining. As in the case of logging, historical placer mining in SONCC rivers has had strong legacy effects. Long reaches of the mainstem Scott River, for example, are now lined with piles of rocky spoils from the large dredges that turned over the landscape in the 19th century. These reaches are largely too warm and shallow to support coho during the summer months today. Similar effects can be seen on other SONCC streams such as the Trinity River. Unfortunately, the rise in the price of gold in recent decades saw a resurgence of instream mining, mostly though the use of small gasoline-powered vacuum dredges. This activity disturbs fish, turns over streambeds, and reduces water clarity when juvenile coho are most stressed because of natural conditions (e.g., warmer temperatures). Fortunately, instream dredging was banned by CDFW in 2016 after a seven-year moratorium.

Estuarine alteration. Perhaps the least appreciated crucial habitat for juvenile salmonids, including coho salmon, is the estuary or lagoon at the river mouth (Wallace et al. 2015). Juvenile coho rear in an estuary for varying lengths of time and most are resident for a few weeks to over a year as they adjust to the shift from fresh to salt water. Consequently, estuaries with abundant food and cover can significantly improve survival rates of out-migrating juveniles. Unfortunately, most estuaries in the SONCC coho region are degraded to some degree. The largest, such as those on the Eel and Mad rivers, have large sections that are diked and drained, with comparatively little habitat remaining for coho rearing. Much of the diking and draining of estuarine habitat has been done to create pasture for livestock.

Alien species. Non-native predators are mainly a problem for coho salmon in the Eel River and off-channel habitats in Humboldt Bay tributaries, where the out-migrants have to pass through large stretches of river containing Sacramento pikeminnow (*Ptychocheilus grandis)*, introduced in the 1980s. The effects of pikeminnow predation on coho are not known.

Harvest. Both legal and illegal harvest have had important effects on coho populations in the past, although until 1950s record keeping was poor and in the early cannery records for the Klamath River coho were often not distinguished from Chinook salmon. Between 1952 and 1992, about 40,000 fish were caught per year in the commercial fishery (high =362,000) and about 10,000 per year (high 69,000) in the sport fishery. The ocean commercial fishery for coho salmon was halted in 1993 and the ocean sport fishery in 1994 and 1995, despite the fact they are mixed stock fisheries with many of the fish coming from Oregon hatcheries and streams. Sport fishing is now not allowed in streams as a result of listing of SONCC coho as threatened under the federal Endangered Species Act. Small numbers are undoubtedly caught and released in both commercial and sport fisheries for other species. Overall, fisheries are having only a minor impact on coho populations today and the closure of fisheries has presumably helped to protect the dwindling California populations.

Hatcheries. Coho are/have been produced in a number of California hatcheries in the SONCC coho region: Rowdy Creek (Smith River), Iron Gate (Klamath River), Trinity (Trinity River), Mad River, and a number of small cooperatively-run hatcheries, although the Rowdy Creek and Mad River hatcheries are no longer in operation. There is also a large hatchery on the Rogue River, Oregon. The largest hatchery is on the Trinity River, which began production in 1963. It has a production goal of 500,000 volitionally released smolts per year, which it usually meets. The other hatcheries combined produce or produced about 200,000 smolts per year. It is significant that hatchery production has failed to halt the decline of SONCC coho salmon spawners or the decline in the fishery. Estimated survival of hatchery-produced smolts from Iron Gate Hatchery is 1.5%, with a range of 0.3 to 3.5% (Chesney 2007). Iron Gate Hatchery completed a hatchery genetics management plan in 2014 to help them move from mitigation to recovery operations, and a plan is currently being developed at the Trinity River Hatchery as well (NOAA Fisheries 2016). All SONCC coho released from these hatcheries are coded-wire tagged to allow for identification, monitoring, and management. According to DFG (2002), 80-90% the coho spawning below Trinity Dam are of hatchery origin, and roughly 1000-2000 fish return to the hatcheries each year (CDFG 2002). The fish produced in these hatcheries have origins from mixed stocks of California, Oregon, and Washington. Until there is evidence to the contrary, it must be assumed that hatchery coho salmon are having a negative effect on native wild coho salmon by competing with them for resources at all stages of their life history (Nielsen 1994). In the Trinity River, it appears that wild SONCC coho have been completely replaced by hatchery fish. The hatchery fish are nevertheless considered part of the ESU because non-native strains of coho ceased being used by the 1970s and all fish spawned at the present time are of Trinity River origin (Spence et al. 2005). If present trends continue, the only coho left in the Klamath-Trinity system will be hatchery origin fish in declining numbers (Quinones et al. 2012).

Table 3. Major anthropogenic factors limiting, or potentially limiting, viability of populations of SONCC coho salmon in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years, whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Reduced habitat, inadequate releases.
Agriculture	High	Irrigation diversions in many streams reduce flows, especially from marijuana cultivation.
Grazing	Medium	Chronic stream bank alteration.
Rural /residential development	Low	
Urbanization	Low	Urban areas mostly small in CA.
Instream mining	Medium	Dredging ban reduces risk; past mining has altered habitats.
Mining	Low	Hardrock mining limited; could become problem on Smith River.
Transportation	Medium	Roads create sediment and erosion.
Logging	Medium	A chronic problem related to roads and other impacts; legacy affects a major issue.
Fire	Low	Can cause siltation of coho streams, loss of shade to cool water.
Estuary alteration	High	Most estuaries highly altered with reduced rearing habitat and connectivity among habitats.
Recreation	Low	Boating, rafting.
Harvest	Medium	Mostly protected; some inadvertent ocean mortality and poaching.
Hatcheries	Critical	Hatchery-origin fish increasingly dominate populations.
Alien species	Low	Few aliens in coho watersheds.

Effects of Climate Change: Moyle et al. (2013) rated SONCC coho as "critically vulnerable" to climate change, indicating it could drive them to extinction. This is a result of populations being low, stream flows being greatly reduced, and watersheds being highly damaged. Predicted effects on coho habitat include increases in stream temperatures, increased variability in flows (including greatly reduced summer flows), and changed ocean conditions increasing variability in productivity. Increased frequency of wild fires may increase erosion, sedimentation, and remove riparian habitat and large woody debris input into streams, thus reducing shading of already warming streams. These on-going changes are being superimposed on the other threats to coho, increasing the likelihood of rapid extirpation as time passes without dramatic action to protect and enhance habitats. As the linkages between climate change and more frequent and prolonged drought become clearer, there is evidence that the combined effects will increase temperatures and reduce streamflow and survival of SONCC coho. During the recent drought, record low precipitation coupled with several of the warmest years on record combined for a "hot drought," which likely significantly exacerbated general drought impacts. Unfavorable oceanographic conditions related to El Nino and the overarching background conditions of

Pacific Decadal Oscillation in temperature in the Northeast Pacific likely also reduced ocean survival of coho even further. Despite numerous and ongoing recovery efforts statewide to address instream habitat, poor productivity in the ocean is likely to reduce abundance in the short-term (NOAA Fisheries 2016).

Status Score = 1.7 out of 5.0. Critical Concern. Critically Vulnerable to extinction as wild fish within next 50-100 years (Table 2). This score is conservative, given the apparent rapid declines of most populations and the probable 95% or more decline in numbers from 50-60 years ago. Garwood (2012) lists 542 potential SONCC coho streams, with an "occupancy rate" of 62%, meaning at least one detection over a three year period in a non-random sampling of streams; it is uncertain what this analysis means, given the general indications of continued long-term decline and/or high variability in numbers in the larger streams. SONCC coho are listed as threatened by both state and federal governments (NOAA Fisheries 2016).

Table 4. Metrics for determining the status of SONCC coho salmon, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	4	Populations in Eel, Klamath, Mattole, and other watersheds.
Estimated adult abundance	2	Most populations are isolated and function independently and are < 100 fish. There are < 5,000 returning wild adults each year.
Intervention dependence	2	All populations require continuous intervention to persist.
Tolerance	1	Coho are among the most sensitive of salmonids to environmental conditions.
Genetic risk	1	See Bucklin et al. (2007).
Climate change	1	Rated critically vulnerable in Moyle et al. (2013).
Anthropogenic threats	1	1 Critical, 3 High factors.
Average	1.7	12/7.
Certainty (1-4)	4	Well-studied populations.

Management Recommendations: Literally hundreds of management actions on hundreds of individual streams throughout the range of SONCC coho are needed to prevent its eventual extinction. This is recognized in the recovery plans of CDFW (2015) and NOAA Fisheries (2014). Over the last several years, state, federal, and county governments have come together to undertake habitat improvements to create off-channel pools, woody debris, alcoves, and beaver dams across the SONCC coho range in California. The Yurok, Karuk, and Hoopa Tribes have also partnered to restore off-channel habitat on the Klamath and Trinity Rivers to support coho and other salmonids. These individual actions must be taken, however, in the context of a broader conservation strategy, on which we focus here.

To stop the decline of coho salmon, spawning and rearing streams must be protected and restored, and connectivity to those habitats that allows for expression of life history diversity must be restored at the watershed scale. Improving conditions for coho salmon is a difficult task because it means modifying logging, farming, and road construction activities in dozens of watersheds and implementing habitat restoration plans along hundreds of miles of streams. In many streams it means that major restoration projects must be funded, completed, and monitored. Keeping sport and commercial fisheries closed or greatly restricted is also a necessity. Given the large scale of problems facing coho salmon, innovative approaches to stream restoration must be tried, working with landowners, timber companies, and gravel miners and capitalizing on voluntary forbearance by citizens in some watersheds such as the Mattole, where local and state partnerships are providing water storage tanks to residents to reduce diversions in the summer and fall (R. Taylor, R. Taylor Associates, pers. comm. 2016). CDFG (W) reports (2002, 2004, 2012) provide many recommendations for improving management, but they are probably insufficient without further changes in public attitudes towards conservation and large increases in funding for restoration of streams, changing forest practice rules, and other major actions. The federal recovery plan for SONCC coho salmon lists projects that should be undertaken to prevent further declines and help bolster SONCC coho numbers; some broad-scale recommendations are listed below:

1. Eliminate or greatly reduce all *production* hatchery programs in order to protect the remaining wild stocks. There is growing evidence that genetically based domestication can occur in a single generation of hatchery rearing (e.g., Christie et al. 2016). Hatchery fish, even with low survival rates in the wild, can compete with, dominate, and interbreed with wild fish, reducing fitness of offspring and eventually creating genetically uniform populations (Chilcote 2011). Careful breeding programs using naturally spawned fish as hatchery stock, can reduce these problems but probably only temporarily. Klamath dam removal may eliminate the source water for the Iron Gate Hatchery and render that facility and its operations ineffective (CDFW 2014). While the current hatchery operates as a mitigation hatchery, it will gradually shift operations to an "integrated type" program, which supports conservation and recovery of coho under the new Hatchery Genetics Management Plan by allowing the environment to drive adaptation of a combined hatchery- and wild- population. This program has an annual production target of 75,000 yearling coho, which consists of juveniles that are 20-50% of natural origin. Whether or not this program can sustain coho populations in the long run is problematical. A grand experiment to see if the effects of hatchery production can be reversed would be to shut down the Iron Gate Hatchery for 5 coho generations (15 years) and then carefully monitor the origins of all fish spawning in tributaries to the Klamath River. New genomic techniques allow for much more sophisticated monitoring at lower cost than current tagging and genetic monitoring programs.
2. Maintain/develop intensive, long-term monitoring programs on the least disturbed streams containing SONCC coho such as Blue Creek, Prairie Creek, and South Fork Eel, as well as some highly disturbed streams, in order to be able to distinguish among causes of coho population fluctuations such as ocean conditions and drought. CDFW is currently monitoring degraded tributaries to Humboldt Bay, and this program should be replicated where possible. Monitoring in these watersheds concurrently would allow population trends to be followed and provide focus for restoration efforts. With impending large changes to the Klamath River, monitoring should be implemented to help assess habitat conditions and coho usage of the nearly 112 km of habitat in the Upper Klamath Population Unit before, during, and after dam removal.
3. Improve over-wintering and off-channel habitat in important coho producing streams to increase survival in fresh water, including greatly increasing the amount of large wood in stream channels (Gallagher et al. 2012).
4. Remove dams on the Klamath River to increase coldwater habitat for coho. Dam removal is slated to be completed by 2020. Consider removing dams on other coho rivers as

well (e.g., Shasta, Mad).

5. Work with landowners on the Scott and Shasta Rivers to improve thermal habitat.
6. Use environmental DNA techniques to sample all 542 streams considered by Garwood (2012) to potentially offer coho suitable habitat. Sample a random selection of these streams to confirm DNA results over three years. Use this information to focus restoration on clusters of streams with consistent coho use.
7. Develop and implement strategies for increasing life history diversity, to get away from the strict three year life cycle model. For example, increase numbers of smolts that leave after two years in freshwater or find ways to increase numbers of jack males. Habitat restoration in estuaries, such as Humboldt Bay, that increase connectivity between stream-estuary ecotones and neighboring watersheds can play a role in potentially increasing life history diversity and expression (Wallace et al. 2015).
8. Design and implement emergency rearing facilities to increase juvenile coho survival during periods of drought or during significant restoration activities. Such facilities should focus on creating as 'natural' an environment as possible.
9. Set aside funds to regain control over illegal marijuana grow sites and their water usage. This is currently being addressed in the California State Legislature with several bills introduced.

While these actions are necessary, most will not realize marked improvement in coho abundance for some time. The species is likely to become endangered in the near future (NOAA Fisheries 2016) without concerted and broad scale efforts to restore their habitat and increase their survival dramatically. The large-scale habitat restoration program necessary to avoid this fate for coho will require hugely increased effort involving increased funding, considerable interagency cooperation, and development of an extensive monitoring program. The challenges of managing such a diffuse resource as coho salmon are considerable, but if the population declines are not reversed soon, all coho salmon are likely to disappear from California.

CHUM SALMON
***Oncorhynchus keta* (Walbaum)**

Extirpated or Critical Concern. Status Score = 1.6 out of 5.0. The status of chum salmon is poorly understood. If they have not already been extirpated as a self-sustaining species, their populations are so small as to be hard to detect.

Description: Chum salmon reach up to 1 m TL and 20.8 kg, but in California they are typically <65 cm TL. Unlike other salmon, except sockeye, they lack black spots on the back and fins. They have 10-14 rays in the dorsal fin, 13-17 in the anal fin, 14-16 in each pectoral fin, and 10-11 in each pelvic fin as well as 11-17 short, smooth gill rakers on the lower half of the first gill arch. The scales are tiny (124-153 in the lateral line) and branchiostegal rays are 12-16 on each side. Spawning male chum salmon have a slight hump and a hooked snout with conspicuous canine-like teeth; they are dark olive on the back and dark maroon on the sides, with irregular greenish vertical bars on the sides. Females are similar in color, although they are less maroon on the sides; they also lack a hump and the jaw is less hooked. Parr have 6-14 pale vertical bars (parr marks) that seldom extend below the lateral line, with light areas in between the marks being greater in width than the width of the marks themselves.

Taxonomic Relationships: The chum salmon forms a distinct evolutionary lineage within the genus *Oncorhynchus* with the pink (*O. gorbuscha*) and sockeye (*O. nerka*) salmon (Healey 1991). Chum salmon have strong homing tendencies (Salo 1991) which contributes to genetic isolation of spawners in different streams. No systematic genetic studies on chum salmon are available for California fish, so their relationship to more northern populations is not well understood. However, coastal populations in California, Oregon, and Washington are considered part of the "loosely defined" Pacific Coast ESU (Johnson et al. 1997, p. 105; NMFS 2005). DNA 'bar coding' of a single chum salmon taken from the San Joaquin River in 2013 indicated it was related to fish from British Columbia and Oregon (Root et al. 2015).

Life History: Because of their economic importance, life history, wide distribution, and habitat requirements chum salmon have been well studied in Asia, Alaska, and Canada (Salo 1991, Moyle 2002).

Although chum salmon have been recorded as migrating over 2,500 km up the Yukon River, Alaska, and the Amur River, Russia, they are not particularly strong swimmers for salmon and are easily stopped by low barriers. This results in most chum salmon spawning within 200 km of the ocean. Some populations even spawn in the intertidal reaches of streams. Chum salmon in the northern half of their range in North America tend to spawn in June through September, while more southern populations spawn in August-January. Adults are usually observed in California streams in December and January, but they can occur as early as August. In Mill Creek, a tributary to the Smith River, chums enter during mid-December, but only in years when stream flows are high. During years of low flow, the fish may be spawning instead in the mainstem Smith or in larger tributaries.

Adults home to natal streams where they spawn at 2-7 years of age, but primarily at ages 3-5 (Salo 1991, Moyle 2002). Each female digs a connected series of redds in which the eggs are deposited during spawning; the female guards the last redd until she dies. Males are sexually active for 10-14 days, spawning with multiple females. Large females produce over 4,000 eggs, but the average fecundity is 2,400-3,100 eggs. Fertilized eggs hatch after about 2-6 months of incubation, usually from December to February. Alevins absorb their yolk sac in 30-50 days, growing to approximately 35 mm TL before emerging from the gravel. Like pink salmon, fry spend only a short time in fresh water, moving into estuaries soon after emerging from the gravel. Depending on distance traveled, fry in fresh water are 35-70 mm TL. They may remain in estuaries, however, for several months before moving out into more oceanic waters. Movement of fry is mainly nocturnal, although they may migrate during the day if water clarity is low.

Fry may not feed in fresh water if their downstream migration is short; otherwise they feed on small aquatic invertebrates, primarily as drift. In estuaries, they feed mostly on benthic prey, such as copepods and amphipods. As they move into deeper water and grow larger, chums devour a wide variety of invertebrates as well as fishes. However, for subadults, gelatinous zooplankton, especially pteropods, seem to be especially important in their diet (Salo 1991). As a result of their short stay in fresh water compared to other salmon, their growth and survival is more closely tied to local ocean productivity than food availability in fresh and estuarine waters (NOAA Fisheries 2014).

Habitat Requirements: Chum salmon adults and maturing juveniles live in the open waters of the ocean, but juveniles are bottom oriented in rivers and streams. Optimal temperature ranges for freshwater portions of the life cycle are: adult migration, 7-11°C (range, 0-21°C); spawning, 7-13°C; incubation, 4-12°C; fry rearing/outmigration, 11-15°C, although fish can successfully live through periods of suboptimal temperatures (Moyle 2002, Richter and Kolmes 2005). Spawning takes place in gravel 1-10 cm in diameter but optimal gravel size seems to be 2-4 cm (Salo 1991). Relatively shallow depths (13-50 cm) for spawning are preferred.

Eggs and alevins occur primarily in fresh water, although spawning in intertidal areas occurs. The fry prefer shallow (< 1 m) water during their out-migration. An acclimation period to estuarine (10-15‰ salinity) conditions may be required prior to entering salt water. Juveniles can be killed by high suspended sediment loads (15.8-54.9 g l^{-1}) that abrade gills and prevent feeding (Moyle 2002).

Environmental tolerances have not been studied in California, the southern end of chum salmon range, but limiting factors are likely to be (1) temperatures >13°C for spawning and incubation, (2) intragravel dissolved oxygen in spawning areas at less than 90% saturation, (3) current velocities over spawning areas flowing at less than 30 cm/sec, (4) potential spawning areas dominated by gravel < 2 cm in diameter, (5) areas suitable for spawning that are < 50 cm deep, (6) streams containing high silt loads, (7) limited ocean access during fry outmigration periods, and (8) limited access of adults to spawning streams (Salo 1991).

Distribution: Chum salmon have the widest natural distribution of any Pacific salmon (NOAA Fisheries 2014). In Asia, the native range of spawning chum salmon seems to be streams from Korea north to the Arctic coast of Russia, although they are now raised in large numbers in hatcheries in northern Japan. In North America they range from the Mackenzie River on the Canadian Arctic coast southward into central California. Nineteenth century narratives generally have chum salmon occurring in coastal streams from San Francisco northward, albeit with no documentation of southern fish (Leidy 2007). They have been caught in the ocean as far south as San Diego, but the southernmost freshwater record is the San Lorenzo River, Santa Cruz County (Moyle 2002). At present, they become progressively less common in southern streams within their historical range (Moyle 2002). The southernmost populations for which annual spawning is documented appear to be in streams tributary to Tillamook Bay region in northern Oregon south to the Nestucca River (Johnson et al. 1997; NOAA Fisheries 2014).

Historically, they were considered to have small spawning runs in the Sacramento and Klamath (Trinity) rivers (Mills et al. 1997) and fish were commonly observed in other coastal rivers as well. During a ten-year (1949-1958) survey of the Sacramento River system, 68 chum salmon were recorded, leading Hallock and Fry (1967) to conclude that a very small run was present. A few spawners still are observed in the Sacramento River but not every year. In recent years, small numbers of adults have been recorded from two San Francisco Bay tributaries, and in 2004, 2005 and 2006 juveniles were collected from the Napa River estuary during a fish monitoring program (Stillwater Sciences 2006, Leidy 2007, Martin 2007). A single adult female was captured in an irrigation canal flowing into the San Joaquin River in December 2013 (Root et al. 2015) and small numbers (6 total in 2001-2004) have been observed ascending the Mokelumne River, which flows into the Delta (M. Workman, EBMUD, pers. comm. 2016).

Chum salmon are observed in the Klamath and Trinity rivers on a regular basis. The California Academy of Sciences has a small collection of parr taken from the Klamath River in 1944. Screw traps set in the rivers catch juvenile chum salmon on an annual basis, at least when they are looked for (Moyle 2002), suggesting small runs still exist. Adults are also occasionally seen but most are presumably overlooked. Thus a few adults have been observed annually in the South Fork Trinity River, the apparent remnant of a larger run that existed there prior to the 1964 flood (T. Mills, CALFED, pers. comm. 1995). One adult was seen in the Salmon River ca. 2007 (J. Grunbaum, USFS, pers. comm. 2009). Monitoring of Mill Creek, a tributary to the Smith River estuary, by J. Waldvogel (2006) suggests that chum salmon spawn there on a regular basis, based on the occurrence of adults, juveniles, and smolts (Stillwater Sciences 2002). They occur often enough to suggest that there may be a small annual run in the lower Smith River. Three adults were caught in the Rowdy Creek Hatchery trap (Smith River, one per year in three consecutive years (A. Van Scoyk, CDFW, pers. comm. 2009), 1994-1996. A single chum salmon was also identified in the headwaters of the North Fork Smith River in Oregon during spawning surveys in winter 2013-2014, the first known recording of this species so far upstream in this basin (Garwood, Larson, and Reneski 2014). Chum salmon are also observed on an irregular basis in other coastal streams, such as Redwood and Lagunitas creeks in Marin County (Ettlinger et al. 2005) and Prairie Creek in Humboldt County (R. Bellmer, CDFW, pers. comm. 2010). Regular surveys of spawning salmon on Lagunitas Creek since 1995 have observed 1-3 chum salmon every year, including individuals on redds (Ettlinger et al. 2005), although there has been no evidence of successful spawning (Ettlinger et al. 2015). Chum salmon have been observed in the Russian River in 6 of 16 years; a total of 16 fish were counted passing through a weir (S. Chase, SCWA, pers. comm. 2016). There is no evidence of spawning although the counting weir was well upstream of much potential spawning habitat.

Trends in Abundance: Chum salmon are abundant from Washington on north, with many runs supported by hatchery production (Johnson et al. 1997). They have probably always been uncommon in California; there is only limited evidence for spawning, although systematic efforts to find observe spawning or collect young have been few. There is evidence of spawning in the South Fork Trinity River. In the period 1985-1990, 1-3 adults were seen or captured every year except 1988 and juveniles were taken on at least six occasions; one pair was observed spawning in 1987, and one fish caught in 1990 was spawned out (Mills et al. 1997). USFWS sampling crews collected 21 chum juveniles and 2 fry in the Trinity River and 4 juveniles in the

Klamath Estuary during 1991 (T. Kisanuki, unpubl. data), but they are easy to overlook among the thousands of other salmon taken in the traps. In the West Branch of Mill Creek, a tributary of the Smith River, 1-8 spawning chums were observed in each of 10 years between 1980 and 2002, entering the stream with Chinook salmon during early to mid-December during high stream flows (Waldvogel 2006). In 2001-2002, both adults and juveniles were observed (Stillwater Sciences 2002). The fact that Mill Creek has had chum spawning reported for so many years is presumably in part a function of observers being present and in part a function of its estuarine position, an attractive location for chum salmon. Even though they are not observed every year, the frequency of observations suggests that alternate spawning areas may also be present in the main stem Smith River or its other tributary streams during years when spawning habitat is not accessible in Mill Creek.

There apparently was once a small run in the Sacramento River, with spawner estimates of 34-210 fish annually in the 1950s (Mills et al.1997). Subsequent records have been spotty (Moyle 2002) and they are rarely seen in salmon surveys. Curiously, chum salmon juveniles have been captured recently in the Napa River, indicating successful spawning (Martin 2007).

Overall, it appears chum salmon spawn, at least sporadically, in California streams from San Francisco Bay north to the Oregon border. Recent evidence suggests that the only California rivers used by chum salmon for spawning on a regular basis have been the South Fork Trinity, Klamath and Smith rivers, although the numbers of fish in each river is (was) small. It is highly likely that chum salmon were more abundant and widely distributed along the California coast in the past. However, chum salmon abundance has always been low in California compared to other salmon, few observers are aware of them, and juveniles are easy to overlook, so there are no reliable trend data available on their abundance. It is reasonable to think, however, that they once maintained small populations in the Sacramento River and various coastal rivers that have been extirpated in the last 50-70 years and that existing populations are likely to be extirpated in the near future, if they have not been already.

In Oregon, Nehlsen et al. (1991) list 10 populations at high or moderate risk of extinction, the closest to California being in the Elk River, in southwestern Oregon (Curry County). Johnson et al. (1997) indicate these populations have largely been extirpated with populations persisting in streams in the Tillamook region (NOAA Fisheries 2014).

Overall, chum salmon likely maintained a number of small populations in California, most of which were lost due to wide-scale changes in California rivers in the past 150 years. Today most occurrences in California are likely strays from the large populations that exist in the Pacific Northwest from both natural and hatchery production.

Factors Affecting Status: The apparent historical rarity of chum salmon in California makes it difficult to identify factors that have negatively affected their abundance. However, chum salmon historically spawned in the lower reaches of river systems in Oregon and California (Salo 1991) and these are the reaches most likely to be degraded by human activity, such as logging, road building, mining, channelization, and draining of estuarine marshes. There is so little information available on them in relation to chum salmon that no further discussion is merited beyond what is presented in Table 1.

If California populations are largely driven by fish 'straying' from more northern populations, especially those with large proportions of hatchery fish, then their abundance would also be related to factors such as ocean conditions, hatchery production, and status of populations in the northern part of their range.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of chum salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is low. See methods for explanation.

Factor	Rating	Explanation
Major dams	Low	Most populations well below major dams but altered flows could be an issue.
Agriculture	Medium	Estuaries highly altered for pasture, long-term degradation of watersheds.
Fire	Low	Fires may increase siltation of spawning areas.
Grazing	Low	Long-term degradation of watersheds increases siltation.
Rural/ residential development	Low	Most rivers where chum salmon are encountered are in rural areas, but impacts of residential development are unknown.
Urbanization	n/a	
Instream mining	Low	Gravel mining and gold dredging could alter habitat.
Mining	n/a	
Transportation	Medium	Roads are present or near most streams.
Estuary alteration	High	Loss of rearing habitat occurs in estuaries as habitat becomes degraded or constricted.
Logging	High	This has had a major impact on known spawning watersheds such as South Fork Trinity River.
Recreation	Low	Lower reaches of rivers where chum spawn are often a focus of fishing and other recreation.
Harvest	Low	Some commercial and recreational harvest; poaching may occur.
Hatcheries	Low	Some possibility that fish in California waters could be strays of hatchery origin.
Alien species	n/a	

Effects of Climate Change: Moyle et al. (2013) consider chum to be "critically vulnerable" to climate change, while admitting very little is known about likely effects. Climate change effects could be minimal on the freshwater portions of the life cycle because most spawning takes place in the lower reaches of rivers, in the coastal fog belt, which are likely to remain cool, except the South Fork Trinity River. On the other hand, even small changes in flows or temperatures and/or small changes in ocean conditions could eliminate small, fragile populations. Poor ocean conditions (e.g., reduced upwelling and food availability, higher temperatures) in particular could reduce survival in the ocean and reduce connections to more northern populations.

Status Score = 1.6 out of 5.0. Critical Concern or Extirpated. Johnson et al. (1997, p. 164) reported chum salmon as being extinct in California and all populations in Oregon as being "depressed or extinct." There is very limited evidence to support the presence of 2-3 very small self-sustaining populations (in Smith, Klamath, and Trinity rivers) in the state, which, if they exist, are all threatened with extinction. However, given the paucity of data, the certainty of the status of these populations is low (Table 2). The alternative, however, is to admit they are extirpated from the state as a viable species with records resulting entirely from fish straying from elsewhere. In this case, spawning in California streams would take place mainly when populations are high in the ocean and ocean conditions are favorable for fish to stray from more northern populations. At present, there is no hard evidence to support either hypothesis, so the conservative course of action is to assume chum salmon populations continue to exist in California and to take actions to enhance them as the southernmost populations of the species. Because California populations are not regarded as a distinct ESU, due to lack of information, chum salmon are not considered to be threatened in California by AFS (Jelks et al. 2008). The southernmost large runs, in the Columbia River and Hood Canal, Washington (summer run) are listed by NMFS as threatened (NOAA Fisheries 2014).

Table 2. Metrics for determining the status of chum salmon in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is low. See methods for explanation.

Metric	Score	Justification
Area occupied	2	If chum salmon are still maintaining populations, there are 2-3 (Smith, Trinity, Klamath rivers).
Estimated adult abundance	1	There is little evidence that any population is more than a handful of spawners, perhaps 6-20 in most years.
Intervention dependence	2	No effort is currently being made to specifically protect chum salmon runs; unlikely for species to persist in CA without intervention.
Tolerance	2	Southern populations have fairly narrow spawning habitat requirements; young require functioning estuarine habitats for rearing.
Genetic risk	1	California populations, if not already extirpated, are extremely small and vulnerable to inbreeding depression and other genetic problems. Genetic studies needed on CA populations.
Anthropogenic threats	2	2 High threats; severity of most threats unknown.
Climate change	1	Small changes in flows or temperatures and/or small changes in ocean conditions could eliminate the populations.
Average	1.6	11/7.
Certainty (1-4)	1	Information is very limited.

Management Recommendations: The chum salmon is an enigmatic, cryptic species in California. Arguably, it is already too late to protect whatever spawning populations once existed and recent records all seem to be of individuals straying from more northern populations. However, surveys in the South Fork Trinity, Klamath, and Smith rivers should be conducted to determine if fish still spawn there. If present, they will benefit from maintaining suitable habitat, flow, and water quality to protect and enhance as a group the imperiled salmonids (including summer steelhead) in those rivers. Once key spawning areas are known, specific plans should be established for enhancing conditions. The management of Mill Creek in the Smith River system may be a model for management of similar streams that might support chum salmon (Stillwater Sciences 2002). However, in general, it is likely that chum salmon, like other salmon, will benefit from restoring the larger coastal estuaries, such as that of the Eel River, to more naturally functioning systems, away from their present state as diked and drained pastureland with narrow channels and breached lagoons.

Genetic studies on California and Oregon chum salmon are needed to determine if they are self-sustaining or are just part of the larger population in the ESU, with southern populations maintained by 'strays.'

PINK SALMON
Oncorhynchus gorbuscha

Critical Concern or Extirpated. Status Score = 1.6 out of 5.0. There are probably only one or two self-sustaining populations in California. It is highly likely of pink salmon will disappear from California streams, except as strays from northern populations, within the next 25-50 years, if they have not already.

Description: Pink salmon are the smallest of the Pacific salmon; adults are usually less than 60 cm SL (2.5 kg). Maximum recorded length is 76 cm SL (6.3 kg). They are distinguished from other salmon species by black oval markings on the tail and back. The number of gill rakers is 16-21 on the lower, first gill arch. The mouth is terminal with sharp teeth on jaws, vomer, palatines, and on the tongue. The dorsal fin has 10-16 complete rays, the anal fin, 13-19, the pectoral fins, 14-18, and the pelvic fins, 9-11. There are 147-198 scales along the lateral line. Branchiostegal rays number 10-15 on either side of the jaw.

Fresh marine-phase fish appear steel blue to blue-green on top, with white bellies and silvery sides. The back and sides have large black spots as do the adipose and caudal fins. Spawning males have a pronounced hump behind the head and the snout is greatly enlarged and hooked. The body color becomes darker as they mature, especially on the head and back. Reproductive females lack the conspicuous hump of the males and resemble trout in general body shape. Their sides are olive green, with long, dusky, vertical markings. Scales in mature pink salmon are deeply embedded. Juveniles in fresh water are small (mostly <40 mm TL) and lack parr marks, unlike all other salmon.

Taxonomic Relationships: This species was first described in 1792 by Johann J. Walbaum from the Kamchatka Peninsula in Russia. Nothing is known about the genetic identities of California fish or how they relate to more northern populations. However, biochemical differences have been observed among pink salmon stocks in different river systems in North America (Hard et al. 1996). Russian workers also have noted genetic differences among stocks in different geographical areas (Omel'chenko and Vyalova 1990). Hard et al. (1996) state that the southernmost populations are in Puget Sound, Washington, and, with one exception, they only have spawning runs on odd years. These odd-year fish are regarded by NMFS as a distinct ESU which is supported in part by a hatchery on the Hood Canal. Presumably California fish are most closely related to members of this Washington ESU, although the presence of some even-year fish in California suggests that the relationships among ESUs may be complex.

Life History: The life history of pink salmon is well known, so this account briefly summarizes information in Scott and Crossman (1973), Heard (1991) and Moyle (2002). Pink salmon live for two years although occasionally three-year-old fish are reported. The adults move into fresh water between June and September and spawn from mid-July to late October, depending on the geographic location. Spawning in California has mainly been recorded in October (Fry 1967, C. Bell, ACoE, pers. comm. 2003) although mature fish have been found in California streams as early as August (Skiles et al. 2013). Most pink salmon spawn in the intertidal or lower reaches of streams and rivers, but upstream migrations of 100-700 km occur in some rivers. In 1891, several mature pink salmon were taken from the McCloud River, over 500 km from the ocean (Skiles et al. 2013). Spawning takes place in flowing water between 20 and 60 cm deep. Six redds in the

Russian River were excavated by females in shallow water, in fairly fine gravel (Fry 1967). While the redd area is typically defended by a large, dominant male, smaller, subordinate males may also contribute to spawning (see Heard 1991 and Moyle 2002).

A female lays 1,200-1,900 eggs during the 3-5 day spawning period. Both males and females die after spawning. Embryos hatch after 4-6 months of incubation, which would mean in February and March in California. Alevins emerge a month or so later, after the yolk-sac has been absorbed. The 30-40 mm-long fry move downstream into the estuary as soon as they can swim, although M. Sparkman (CDFW, pers. comm. 2011) found a smolt that was 67 mm FL in Redwood Creek (Humboldt Co.). Fry out-migration takes place at night and fish can reach the estuary in just one night. In the estuaries of larger rivers, they form large schools and may remain for several months before moving out to sea. Most juveniles do not feed in fresh water but in estuaries and the ocean, they consume small crustaceans and other invertebrates. As they grow larger, they feed increasingly on small fish, squid, and shrimp.

Pink salmon roam much of the North Pacific Ocean; tagged individuals have been captured 2,700 km (1,700 mi) from where they were tagged (Omel'chenko and Vyalova 1990), suggesting many swim great distances in their short life spans. However, they generally return to their natal streams for spawning. The discrete two-year life span of pink salmon results in genetically distinct populations, even odd- and even-year spawning runs in the same river system. Odd and even-year runs, however, do not necessarily occur in all rivers used by pink salmon. Fisheries in Washington and Oregon have caught significant numbers only in odd years, and in California most records of pink salmon are also for odd years (Hallock and Fry 1967), although Redwood Creek apparently has supported a small run on even-numbered years (Sparkman 2005).

Habitat Requirements: Spawning streams for pink salmon have shallow, riffle sections with small gravel substrates, where spawning takes place at depths of 30-100 cm and velocities of 30-140 cm/sec (Heard 1991). Spawning can take place at temperatures ranging from 5 to 18°C but at the southern end of their range, spawning takes place in winter, when temperatures in coastal streams are generally <10°C. Incubation has been recorded at 3-15°C, with warmer temperatures shortening incubation time. Pink salmon fry head out to sea shortly after they emerge from the gravel and absorb their yolk sacs, generally spending less than a few days in fresh water if travel distances are short. Rearing temperatures are likely to be similar to incubation temperatures (Heard 1991).

Environmental tolerances have not been studied for California populations, but factors limiting their distribution and abundance in California are likely to be (1) temperatures >15°C for spawning and incubation, (2) intragravel dissolved oxygen in spawning areas of less than 90% saturation, (3) current velocities over spawning areas of less than 30 cm/sec, (4) potential spawning areas dominated by sand and silt, rather than clean, coarse gravel, (5) water excessively turbid with silt, and (6) limited ocean access during spawning and fry outmigration periods (Heard 1991).

Distribution: In Asia, pink salmon are found in coastal streams from Korea through Japan to Siberia (Heard 1991). Along the Pacific coast of North America they spawn in streams from the MacKenzie River in the Yukon Territory (Canada) south along the coast of Alaska and British Columbia to California. Isolated oceanic records have been documented as far south as La Jolla (Hubbs 1946). However, the largest runs on the southernmost end of their range are in streams tributary to Puget Sound (Hallock and Fry 1967, Hard et al. 1996). Successful spawning by pink salmon has been recorded from Oregon streams and they are caught in small numbers in the commercial fishery off the Oregon coast (Hard et al. 1996).

There are a number of reports of pink salmon from California, from the 19th century to present, and from the Big Sur region to the Oregon border. Most records are from odd years, perhaps reflecting a connection to the mostly odd year runs in Puget Sound. Unless otherwise indicated, the records discussed in the following paragraphs are from Skiles et al. (2013).

South of San Francisco Bay. The southernmost stream record of adult pink salmon is in Big Creek and its tributary Devils Creek, Monterey County, at the upper end of the Big Sur Coast. In September and October, 2011, three pink salmon were identified from the river. In August, 2011, four pink salmon adults were captured in the Salinas River, about 7 km up from the mouth. Snyder (1931) thought the Salinas River was the southern end of the pink salmon range, but did not provide any records of their occurrence there. However, small numbers have been reported from the San Lorenzo River, a bit further north (Scofield 1916).

Sacramento River. During the 1800s, pink salmon were reported in the Sacramento River, "... which it [*sic*] ascends in tolerable numbers in October" (Calif. Comm. of Fish. 1881, p. 54). During the 1930s, commercial fishermen on the Sacramento River reportedly captured a dozen or more pink salmon in some seasons (Hallock and Fry 1967). In the period 1949-1958, 38 pink salmon were recorded from the Sacramento River system; this included 12 fish from Coleman National Fish Hatchery, 4 in Mill Creek (Tehama Co.) and 3 at Nimbus Fish Hatchery on the American River (Hallock and Fry 1967). More recent occurrences of pink salmon have been infrequent. One was seen in the American River (T. Mills, CDFW, pers. comm. 1995) and three more (males) were taken on that river on separate occasions (R. Ducey, CDFW, pers. comm. 1995). Regardless of the limited sightings, spawning does occur on occasion. Seven juvenile pink salmon were captured at the state J.E. Skinner Fish Protective Facility near Tracy in March 1990 (D. McEwan, CDFW, pers. comm., 1990). Occasional pink salmon continue to be caught from the river and its tributaries, such as two that were caught in the Yuba River in September, 2011 (D. Massa, unpubl. obs.).

Russian River. The first published report of pink salmon in the lower Russian River (Sonoma Co.) was a 1937 sighting by Taft (1938). Since then, there have been intermittent reports of their presence. In October, 1955, there was evidence of a small run, including observations of redds and spawned-out fish (Fry 1967). Pink salmon have also been observed in 2003 and 2008 (Chase et al. 2005; S. Chase, SCWA, pers. comm. 2008).

North Coast rivers. Spawning pink salmon have been reported from a number of streams north of San Francisco Bay besides the Russian River: including Lagunitas Creek (Marin Co.), Garcia River and Ten Mile rivers (Mendocino Co.), Mad River, and Redwood Creek and its tributary Prairie Creek (Humboldt Co.) (Taft 1938, Smedley 1952, Roedel 1953). A pink salmon caught in the Mad River was reported in the *Arcata Union* (Sept. 6, 1928) which stated that "this species had been frequently taken in the Mad River by net fishermen many years earlier." In 1937, Taft (1938, p198) reported "many quite large schools" in Ten Mile River and "several hundreds" spawning in a 3 km reach of the lower Garcia River. In recent years, the most consistent occurrences seem to have been in odd years in the lower Garcia River, although not all years were surveyed. In 2003, 23 pink salmon redds were documented in one incomplete survey. Sparkman (2005, 2015) captured small numbers of juvenile pink salmon in out-migrant traps in Redwood Creek in 2000, 2002, 2004, 2005, 2008, 2011, 2013, and 2014, suggesting spawning was taking place in both even and odd years. Spawning adults were observed in the watershed in 2008 and 2010. According to Sparkman et al. (2015, p70): "Based on our trapping data in Prairie

Creek and Redwood Creek…Pink Salmon are present and reproducing, albeit in small numbers."

Trends in Abundance: In Alaska and Canada, pink salmon are extremely abundant and support major commercial fisheries. Generally, the odd year runs are bigger than even year runs, even in the same streams or regions. California is the southern edge of their range so pink salmon have never been common here. However, given that pink salmon spawn in the lower reaches of streams during the fall, when few observers are likely to be present, and given that their young go out to sea immediately after emerging from the gravel, spawning pink salmon in coastal streams would be easy to overlook, especially when outmigrant traps are located some distance above the estuary. Nevertheless, in the late 1880s, pink salmon were included in the salmon catch sent from the north coast to San Francisco markets (U.S. Comm. Fish and Fisheries 1892). As indicated in the distribution section, they were widely distributed in California both historically and in more recent times. Overall, it seems highly likely that pink salmon were once common enough in California to support small runs in several rivers. They appear to be much less common today than they were historically and it is not certain how much present day records depend on straying from more northern populations versus home-grown production.

Persistence of pink salmon in California seems unlikely without special management of spawning reaches of the Garcia River, Redwood Creek and similar streams, as well as their estuaries. Use of conservation hatcheries to enhance whatever populations exist may be required at some point. If climate change results in a northward shift of the southern boundaries of spawning anadromous fishes, pink salmon will probably disappear from California for good.

Factors Affecting Status: The sparseness of historical data on the abundance and distribution of pink salmon in California makes assessment of factors affecting their status difficult. In fact, it is not certain whether any population, as defined by McElhany et al. (2000), still exists in California, although the population in the Redwood Creek watershed seems persistent, if small. Pink salmon historically may have been a species that occurred in California mainly as a 'sink' population from sources further north. If so, then their abundance in the state would have mainly reflected the abundance of populations in Washington and British Columbia, which have mainly odd-year runs. On the other hand, if pink salmon have (or did have) self-sustaining populations in California, as seems likely, their tendency to spawn only short distances upriver from the ocean makes them extremely vulnerable to the general degradation of estuaries and the lower reaches of coastal rivers. This degradation results from farming, logging, channelization, gravel mining and other human activities, many of which continue. The timing, location, and short duration of spawning also make them very hard to observe.

Examples of threats facing pink salmon in just three streams in California include:

- Lower Russian River and its lagoon have been altered by gravel mining and lagoon breaching. The river also has an altered flow regime, with many diversions, and some pollution from agricultural and urban sources. The lower river is much less suitable for pink salmon spawning.
- The Garcia River watershed has been highly impacted by logging in the upper reaches and pasturage in the lower reaches. The spawning areas are unprotected.
- The Redwood Creek estuary is mostly confined into narrow channels as the result of diking and draining large areas for pasture.

Generic threats are presented in Table 1 with no further discussion because of the lack of hard, pink salmon-specific information. Because whatever populations still exist are so small, stochastic factors, such as exceptionally large scouring flow events, can have major impacts.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of pink salmon. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factors	Rating	Explanation
Major dams	Low	Probably reduced distribution in larger rivers such as the Sacramento and Mad rivers but effects minor.
Agriculture	High	Alteration of land for farming has reduced spawning and rearing habitat; diversion of water has reduced flows and warmed water.
Fire	Low	Most watersheds where pink salmon occur are in the coastal fog belt, and therefore major fires are unlikely.
Grazing	Low	Long-term degradation of watersheds possible.
Rural /residential development	Low	Little impact.
Urbanization	N/A	
Instream mining	Medium	Gravel mining (e.g., Russian River) and gold dredging could alter habitat.
Mining	N/A	
Transportation	Medium	Roads are present along or near most streams.
Estuary alteration	High	Pink salmon often spawn just above tidal influence; alteration of estuaries has reduced spawning and rearing habitat.
Logging	Medium	This has been a major impact in the past on all watersheds used.
Recreation	Low	Lower reaches of rivers where pink salmon spawn are often a focus of fishing and other recreation.
Harvest	Low	Some commercial and recreational harvest.
Hatcheries	N/A	
Alien species	N/A	

Effects of Climate Change: Moyle et al. (2013) rated pink salmon "critically vulnerable" to the effects of climate change, making extirpation from the state likely as water temperatures and flows change. Because they spawn in streams close to the ocean in the coastal fog belt, pink salmon should in theory be less vulnerable to climate change than many species. However, three factors may contradict this: (1) their habit of spawning in fall, a period when rainfall may become even more variable than it is today (2) the possibility that climate change may also reduce coastal ocean productivity, so it is less suitable for rearing of juveniles, and (3) sea level rise may severely alter the lower reaches of rivers and their estuaries, making them less suitable for pink salmon spawning and rearing. These factors could result in a general northward shift of spawning salmon populations and eliminate pink salmon from California completely.

Status Score = 1.6 out of 5.0. Critical Concern or Extirpated. Pink salmon were considered by Hard et al (1996), Moyle (2002) and Augerot and Foley (2005) as extirpated from California, except for strays from northern populations. They were listed as a Species of Special Concern by the California Department of Fish and Wildlife in 1995, although they were considered to be "extinct or endangered in California" (Moyle et al. 1995). Continued uncertainty as to their status led to them being removed from the revised report (Moyle et al. 2015). Katz et al. (2013) gave their status as endangered (status score of 1.3 using metrics similar to those in Table 2). The status given here, with a certainty rating of '2' (of 4), is based on the reports of a spawning run in the Garcia River and the presence of juveniles over a 15 year period in the Redwood Creek drainage. These observations suggest that small populations still exist in the state and have been overlooked or ignored. Assuming there are regular spawning populations, their small size indicates high vulnerability to extirpation. It is highly likely reproducing populations will disappear completely from California streams within the next 25-50 years without positive action. However, it is possible that populations have naturally gone extinct during long periods of drought and then become re-established through straying from more northern populations.

Table 2. Metrics for determining the status of pink salmon in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is low. See methods for explanation.

Metric	Score	Justification
Area occupied	1	Only confirmed from Garcia River and Redwood Creek.
Estimated adult abundance	2	Numbers very uncertain, so this is a best guess.
Intervention dependence	3	Largely unstudied, but some intervention needed if this species is to persist.
Tolerance	1	Short life cycle, dependent on 1-2 streams.
Genetic risk	1	If a local population, then risk is high.
Climate change	1	Rated "critically vulnerable" by Moyle et al. 2013.
Anthropogenic threats	2	2 High and 3 Medium threats; loss of important spawning and crucial estuarine habitat.
Average	1.6	11/7.
Certainty (1-4)	2	Very limited documentation.

Management Recommendations: The first step in managing this species is to determine the status of populations in the Redwood Creek watershed and in the Garcia River through continuing existing studies or adding additional studies (e.g. estuarine monitoring of juveniles), if needed. The lower reaches of the Ten Mile, Russian and other potential spawning rivers should be thoroughly surveyed for spawners at the appropriate time of year (mid-September through November). If viable spawning populations exist, as they seem to in at least the Redwood Creek watershed, then habitat, flow, and water quality should be protected. In addition, it is especially important to protect potential spawning areas in lower reaches of coastal streams and restore estuaries (e.g. Redwood Creek) as functional rearing habitat for salmonids in general. Last, genetic studies on pink salmon in California should determine affinities to northern populations.

STEELHEAD

CENTRAL CALIFORNIA COAST STEELHEAD
Oncorhynchus mykiss irideus

High Concern. Status Score = 2.0 out of 5.0. Central California Coast steelhead DPS populations are in long-term decline, and face extinction in the next 100 years without significant investments in monitoring, habitat restoration, and water management.

Description: Central California Coast steelhead are anadromous coastal rainbow trout, and are made up of populations downstream of manmade or natural barriers throughout their range. A description of juveniles and adults is similar to that of steelhead in the Northern California winter steelhead account.

Taxonomic Relationships: The Central California Coast (CCC) steelhead DPS is a complex group of populations inhabiting a region that has been the recipient of hundreds of thousands of out-of-basin juvenile steelhead releases over decades. CCC steelhead have also been used as a source for numerous transfers into the neighboring South-Central Coast steelhead DPS (Bjorkstedt et al. 2005). Using microsatellite markers Garza et al. (2014) found that juvenile steelhead from CCC streams were distinct from those from other northern California DPSs, with a closer relationship to more southerly steelhead populations. Within the Russian River, samples of steelhead show two genetic patterns: mainstem river and headwaters. In the mainstem, steelhead below natural barriers are not different from each other or from fish found above recently constructed dams. However, six steelhead populations above natural barriers were significantly different genetically from other populations, suggesting long term isolation and limited genetic diversity (Deiner et al. 2007).

More recently, Garza and colleagues (Garza et al. 2014) studied microsatellite DNA in coastal California steelhead and found five distinct groups delineated by major geographic features along the shore. The five groups include: 1) Big Sur creeks to the mouth of San Francisco Bay, 2) San Francisco Bay tributaries to the Russian River, 3) Gualala River to Usal Creek, 4) Big Creek to Mad River (except Freshwater Creek), and 5) Freshwater Creek to the Oregon border. They also found that geographic distance among populations was directly related to genetic similarity, providing evidence that populations in smaller streams are reliant on migrants from nearby basins for persistence. Finally, more northerly populations had larger populations and more genetic diversity than southern populations (Garza et al. 2014).

Life History: CCC steelhead trout show a tremendous amount of juvenile and adult life history variation to match the varied systems they inhabit, though all adult runs occur during the winter. Shapovalov and Taft (1954) identified 32 different combinations in the amount of time steelhead spent in fresh and salt water, although most of the fish were of four types (freshwater years/saltwater years): 2/1 (30%), 2/2 (27%), 3/1 (11%), and 1/2 (8%). The remaining 28 life history combinations comprised less than 5% of the run. Shapovalov and Taft (1954) observed steelhead entering Waddell Creek as early as late October following the opening of the lagoon three to six weeks earlier. However, the majority of CCC steelhead enter rivers later in the season, typically between late December and February when flows are highest. CCC steelhead are mostly ocean-maturing ecotype fish, and enter rivers in reproductive condition (Hodge et al. 2014). Most spawning typically occurs during late spring (February to April) to which reduces the negative effects of winter scouring flows that are common to the small, short streams along California's central coast. This late spawning strategy also permits CCC steelhead to spawn in upper portions of seasonally flowing watersheds, which are encountered in the southern portion of their range. On the Russian River, steelhead enter freshwater between November and February (Fry 1973). Shapovalov and Taft (1954) observed that 3+ year old fish (35%) and 4+ fish (46%) comprised the majority of spawners. The opposite seems to be true at the more northern end of the DPS range, where younger fish predominate. Typically in late-December through February, following moderate to large storms and subsequent lagoon breaching in small, flashy coastal streams (e.g., Mendocino coastal streams), female adult steelhead will often complete their spawning cycle rapidly and migrate back to sea in as little as two weeks if flow conditions allow. Male adult steelhead tend to have a much longer spawning cycle, as they attempt to maximize their spawning opportunities (J. Fuller, NMFS, pers. comm. 2016). Steelhead are iteroparous, but only 17% of Waddell Creek spawners spawned more than a single time (Shapovalov and Taft 1954).

Development of steelhead eggs is dependent upon water temperature. Shapovalov and Taft (1954) estimated hatch time to be 25-35 days, with emergence of fry after 2 to 3 weeks for alevin development. Hayes et al. (2008) found juvenile growth rates were influenced by variables including flow, temperature, young-of-year (YOY) coho salmon and YOY steelhead densities. Age 0+ steelhead move into deeper water as they grow. Juvenile steelhead and coho salmon often use similar habitats, mostly pools, to oversummer in coastal streams with abundant riparian vegetation, woody debris, and other instream shade and cover. As a result, they often get trapped together in pools as flows in riffles become subsurface in fall.

On Waddell Creek, Shapovalov and Taft (1954) observed a bimodal juvenile emigration pattern, with peaks in early January and mid-March, although they moved downstream during all seasons of the year. In general, older age classes of juveniles migrated earlier. For example, on Scott Creek, Hayes et al. (2011) found that larger smolts moved downstream from February to March and emigrated to the ocean, while smolts moving in April through June were smaller and tended to rear in the estuary. CCC steelhead smolts quickly adopt a saltwater tolerant physiology (Satterthwaite et al. 2012). Hayes (2008) described three life history pathways prior to ocean entry. Some juvenile steelhead emigrated to the estuary after spending only a few months in the upper watershed, while a second group spent one to two years rearing in the upper watershed before emigration. A third group reared for at least a year in the upper watershed, followed by downstream migration and immediate ocean entry without estuarine occupancy. These life history pathways are not discrete, however, and represent a continuum of opportunistic strategies based on a variety of factors including smolt density, prey abundance, temperature, and streamflow. Hayes and colleagues (2011) also recently described a fourth life history strategy, called "double-smolting," whereby summer recruits to estuaries migrated back upstream as water quality conditions declined in the estuary through the summer and fall months, adjusting their osmoregulation physiology as needed to allow them to rear in either fresh or brackish water.

Smoltification of juvenile steelhead seems to occur after a size threshold (100-110mm FL) has been reached, and is accompanied by physiological changes such as increased levels of Sodium-Potassium-ATPase processing enzymes in the bloodstream (Hayes et al. 2011). In Waddell Creek and the San Lorenzo River, steelhead typically reach age 1+ before they are large enough to undergo smoltification. In the Napa River, smolts are very large, and are capable of changing their average size at smolting over time to adapt to changing environmental conditions. From 2010-2016, median steelhead smolt length decreased noticeably from an average of 186mm to 170mm (J. Koehler, NRCD, pers. comm. 2016). Due to potentially restrictive summer habitat availability, age 1+ and 2+ steelhead juveniles are not as common in the CCC steelhead streams as in streams further north (Smith 2002). Limited growth during the summer was observed in age 1+ steelhead in the upper Scott Creek watershed, and most emigrated to exploit rapid growth opportunities in estuaries prior to reaching age 2 (150mm, Hayes et al. 2008).

Smith (2002) found favorable conditions for rapid growth in productive lagoons and estuaries at the mouths of streams with high summer flow, although these can be tempered by elevated mortality risks and density dependent growth variables (NMFS 2016). Estuaries along the Central California Coast are variable in size, but sandbar formation typically occurs in the early summer and they become seasonal freshwater lagoons during low flow conditions. These areas constitute relatively small portions of steelhead habitat, but seem to be a critical nursery area for juvenile steelhead in spring and summer months. In the Russian River estuary, which does not always close at its bar, steelhead preferred middle and upper portions of this habitat and were almost exclusively captured at confluences with tributaries (Cook 2005). More recent study (Fuller 2011) found more widespread usage of estuaries depending on season and water quality conditions such as temperature and dissolved oxygen content. In this habitat, juvenile steelhead increase in average size until mid-summer, then decrease in size, suggesting YOY continue to enter the estuary while larger smolts either emigrate or move upstream (Cook et al. 2005). In Scott Creek, Bond (2006) found juveniles emigrated into the estuary at all sizes, but larger smolts had a higher survival rate at sea based on mark-recapture studies. YOY juveniles remained in the estuary until it became a closed freshwater lagoon, and experienced high growth rates and a doubling of fork length (206mm mean FL). The growth rates of juveniles in the estuary varied among years and also appeared to be density-dependent (Hayes 2008). Residence times in the Russian River for a relatively small sample size of juvenile *O. mykiss* ranged from 4 to 121 days, which significant growth observed as residence time increased (Fuller 2011). In addition, upstream movements of acoustic tagged wild steelhead and captures of untagged estuary steelhead in fall suggest that long estuarine residency and accelerated growth resulted in half-pounder-like life history traits similar to those of more northern populations (Fuller 2011, see Northern California winter steelhead account). Bond (2006) found that juvenile steelhead in Scott Creek larger than 150mm FL, while comprising less than 50% of the juvenile population of the estuary, have a significant survival advantage in the ocean, and accounted for 85% of returning adults.

Tradeoffs, such as those between mortality and growth rates, likely drive life history variation in steelhead. Similar-sized juvenile steelhead engage in potentially-inherited bet-hedging behavior to avoid the high-risk (increased predation)-high-reward (faster growth) closed lagoon habitats in spring and summer months (Satterthwaite et al. 2012). Faster freshwater growth can increase marine survival, but larger size can decrease survival in fresh water. By measuring growth and survival in Central California Coast streams, Satterthwaite (2012) could predict the most frequent age of smoltification in steelhead (age 2+). Smaller fish reared in the lagoon temporarily before migrating back upstream to rear, whereas larger fish mostly went out to sea or stayed in the lagoon before eventually undertaking an ocean migration. Recent work (Boughton et al. 2017) discusses the important tradeoffs of growth opportunities with challenges (predation, poor water quality, etc.) in detail and frames the topic in terms of bioenergetics costs and rewards.

Habitat Requirements: CCC Steelhead require similar freshwater spawning and rearing sites as described in the Northern California winter steelhead account. Leidy (2007) found the abundance of CCC steelhead juveniles in the San Francisco Bay Area was positively correlated with elevation, stream gradient, and percent native species, but negatively correlated with average and maximum depth, wetted channel width, water temperature, open canopy cover, and the total number of fish species (Leidy 2007). This indicates that steelhead were mainly found in small, coldwater streams with few pools, which may be partially an artifact of the urbanization of the lower reaches of the streams. The most apparent limiting factor in streams supporting steelhead throughout the DPS is over-summering habitat for yearlings. These fish require deep water with overhead cover for protection from predators. In the Napa River watershed, steelhead were found to mostly use transitional gradient habitats from 1-6% slope in Redwood, Dry, Milliken, Sulphur, Napa, and York creeks, (J. Koehler, NRCD, pers. comm. 2016). Mainstem river habitats, such as the Napa River and Sonoma Creek, are capable of supporting spawning and rearing steelhead opportunistically, if flows are sufficient to provide cover from predators. Juvenile steelhead have been sampled in mainstem rivers near tributaries in the spring while feasting on Sacramento sucker (*Catostomus occidentalis*) eggs (J. Koehler, NRCD, pers. comm. 2016).

CCC steelhead require cool water, though these fish manage to grow and even thrive in warmer waters than their Northern counterparts. The optimal temperature range for juvenile steelhead growth is 15-18ºC (Moyle 2002). While cool water is typically found in headwaters and marine-influenced coastal regions of the DPS range, these steelhead will tolerate warmer temperatures if food is abundant. Smith and Li (1983) observed juvenile CCC steelhead moving into riffles when temperatures became stressful because of increased feeding success, despite higher energetic costs. Lagoon habitats, which become closed off from the ocean in spring and summer in most locations, provides heterogeneous thermal habitats, where steelhead can easily move between stratified cooler and warmer habitats. Inflows in these habitats must be sufficient to breach lagoon bars at least every few years to allow migration of adults and smolts. Generally, CCC steelhead juveniles are absent from waters that exceed 25-26°C for even short periods. For adult steelhead, lethal temperatures are 23-24°C (Moyle 2002).

Distribution: The CCC steelhead DPS includes all populations below natural and manmade barriers from the Russian River (Sonoma Co.) south to Aptos Creek (Santa Cruz Co.), although this delineation may change based on new genetic data (Garza et al. 2014, Figure 1). The most recent status review from the National Marine Fisheries Service (NMFS) identified watersheds that harbor essential populations (those that have physical or biological features essential to conservation of the species) that must be bolstered to stabilize for recovery, including (from North to South): Russian River, Salmon, Stemple, Walker, and Lagunitas creeks North of San Francisco Bay; Corte Madera, Novato, Sonoma, Suisun creeks, Petaluma and Napa rivers, Alameda, Coyote, and San Francisquito creeks, and Guadaulupe River within the boundaries of

San Francisco Bay; and Pilarcitos, San Gregorio, Pescadero, Waddell, Scott, Soquel, and Aptos creeks, and San Lorenzo River, South of San Francisco Bay (NMFS 2016).

Figure 1: Map of Central California Coast steelhead with diversity strata boundaries.

Figure 1. Watershed regions supporting different populations (diversity strata) of Central California Coast steelhead From NMFS 2016, Fig. 1, pg. 12.

Despite new insights into their distribution, NMFS has not changed the DPS' geographic boundaries, pending further review of additional information (NMFS 2016). The CCC steelhead DPS remains centered on populations in the Russian River and San Francisco Bay (Spence et al. 2007). In the Interior Region, the upper Russian River mainstem reaches upstream of Big Sulphur Creek provide sufficient habitat and isolation to support an independent population, while tributaries such as Mark West, Dry, and Macamas Creeks historically had potentially independent steelhead populations. Lower Russian River tributaries with potentially viable populations such as Austin Creek and Green Valley Creek are included in the North Coastal

Region, with tributaries around Tomales Bay. These populations were all historically dependent upon dispersal from Russian River and San Francisco Bay populations, although some contain sufficient habitat to be designated potentially independent populations (Spence et al. 2007).

Within the San Francisco Bay Coastal and Interior Region, independent populations are/were found in the Guadalupe and Napa rivers, as well as in far inland San Leandro, San Lorenzo, Coyote, and Alameda creeks. Populations of non-hybridized *O. mykiss* still reside in the upper reaches of streams that feed storage reservoirs, such as Upper San Leandro Reservoir upstream of Chabot Dam (EBMUD 2008) and are included in the DPS. Steelhead in drainages of San Francisco, San Pablo, and Suisun bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers are also part of this DPS. This region includes coastal temperate habitats dominated by redwood forests as well interior Mediterranean habitats covered by chaparral and oak woodlands. Numerous San Francisco Bay tributaries historically harbored small populations, but most currently lack sufficient habitat for self-sustaining populations. In general, the most important San Francisco Bay steelhead tributaries were Suisun Creek (Solano Co.), Napa River (Napa Co.), Sonoma Creek (Sonoma Co.), Corte Madera Creek (Marin Co.), San Francisquito Creek (San Mateo Co.), Guadalupe River and Coyote Creek (Santa Clara Co.), and Alameda Creek (Alameda Co.), and most continue to harbor populations of resident *O. mykiss* that can give rise to anadromous offspring (Leidy et al. 2007, M. Leicester, CDFW, pers. comm. 2016). Historically, fish in the headwaters of these systems could re-colonize downstream areas, but are now largely cut off by dams (e.g., Warm Springs-Russian River, Calaveras-Alameda Creek, and Guadalupe-Guadalupe River). One notable exception, coastal Scott Creek (Santa Cruz Co.) offers 23km of accessible, suitable stream habitat (Satterthwaite et al. 2012) and its fish populations have been studied for decades (Shapovalov and Taft 1954).

In the ocean, little is known about CCC steelhead habitat usage, as only a few CCC steelhead have been captured in trawl surveys along the California coast (Brodeur et al. 2004). It is hypothesized that California steelhead migrate to cool water offshore of the Klamath-Trinidad coastline before migrating to the North Pacific feeding grounds, similar to Northern California steelhead; they are generally encountered in trawl surveys much farther offshore, and in fewer numbers, than Chinook or coho salmon (Harding 2015).

Trends in Abundance: CCC steelhead abundance data is very limited due to the large number and diversity of watersheds across the DPS, but numbers appear to be less than 10% of historical estimates throughout the region. During the early 1960s, CDFW (CDFG 1965) estimated that about 94,000 steelhead spawned in this DPS, with most spawning occurring in the Russian (50,000) and San Lorenzo rivers (19,000). In 2006, NMFS estimated that only 14,100 annual spawners remain. Trend analysis is difficult because little empirical data is available. Most sampling efforts are conducted by municipal utilities per mitigation mandates for operating water storage reservoirs. Juvenile sampling, while highly variable annually and geographically, often represent the only data for abundance estimates throughout the DPS. Information on available habitat also provides insight into population status; most streams in the DPS are listed as impaired under the Clean Water Act. There is little sign of major habitat improvement, despite many local efforts, so CCC steelhead populations are still assumed to be declining.

The Russian River probably once supported the third largest steelhead run in California, and was historically the most important for CCC steelhead. Steelhead abundance in the Russian River has declined from an estimated 50,000 in the 1960s to less than 7,000 in the 1990s (Busby et al. 1996; Good et al. 2005), and is likely lower now. To mitigate for lost habitat above Lake

Sonoma, the U.S. Army Corps of Engineers uses Warm Springs Fish Hatchery (Sonoma Co.) to release about 50,000 smolts per year; these hatchery fish account for nearly 95% of returning adults each year (NFMS 2016). CDFW also operates Coyote Valley Hatchery at Lake Mendocino (Mendocino Co.), which releases approximately 200,000 yearling steelhead into the East Branch of the Russian River each year to meet a compensation goal of 4,000 returning adults. However, this goal has only been attained once since operations began in 1992 (CDFW 2016). The private Bill Townsend Conservation Hatchery raises fish from approximately 30,000 fertilized eggs from the Coyote Valley facility for release every year (CDFW 2016). Their effectiveness in boosting steelhead abundance in the Russian River is unclear (Figure 2).

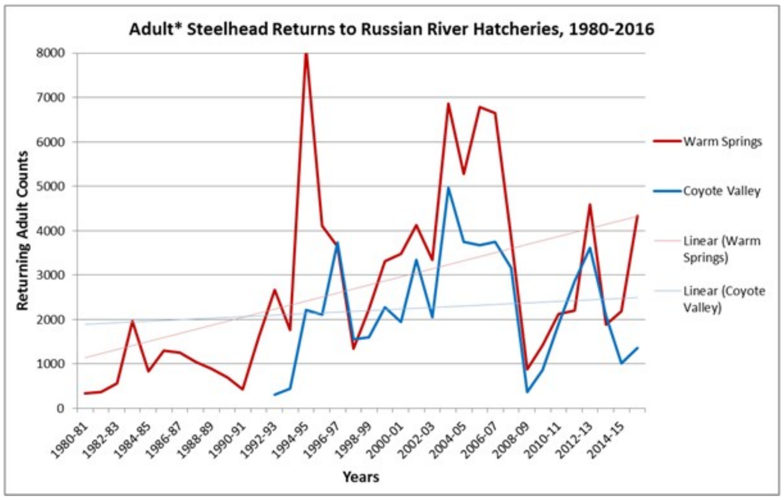


Figure 2. Adult steelhead returns to Warm Springs and Coyote Valley Hatcheries. From R. Watanabe, CDFW 2016. *Half-pounders are not included on this graphic.

Historically, tributaries in Sonoma and Marin Counties were estimated to contain a combined 12,000 adults annually. The annual run in Lagunitas Creek was about 500 fish during the 1990s (McEwan and Jackson 1996), and has remained so (Figure 3).

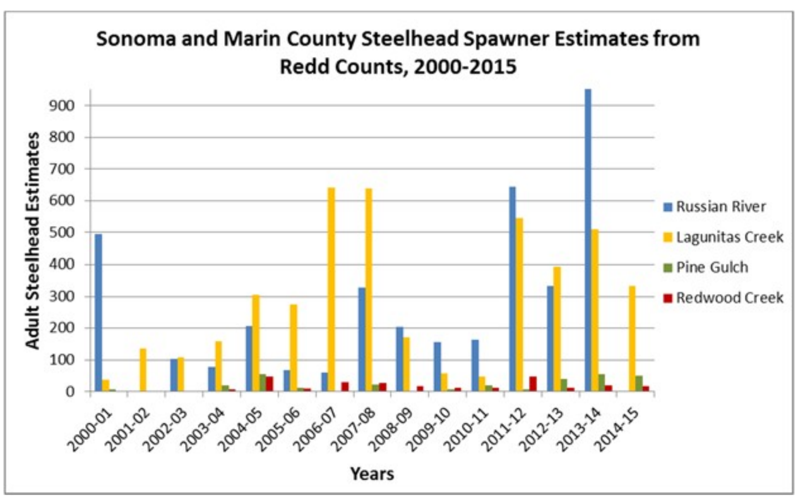


Figure 3. Numbers of spawning adult steelhead in Sonoma and Marin County streams, as estimated from redd surveys. Redd surveys conducted by CDFW; data compiled by The Nature Conservancy and NMFS. Data from D. Logan, NMFS 2016.

Redd counts provide one indicator of adult steelhead abundance in the Lagunitas watershed (Figure 4). Reintroduction of coho salmon (*Oncorhynchus kisutch*) and restoration efforts in the nearby Walker Creek basin since 2003 have increased monitoring and effort in the region to restore populations; over 1,700 steelhead juveniles and several dozen adult redds were observed in mainstem Walker Creek in 2013 (Garcia and Associates 2013).

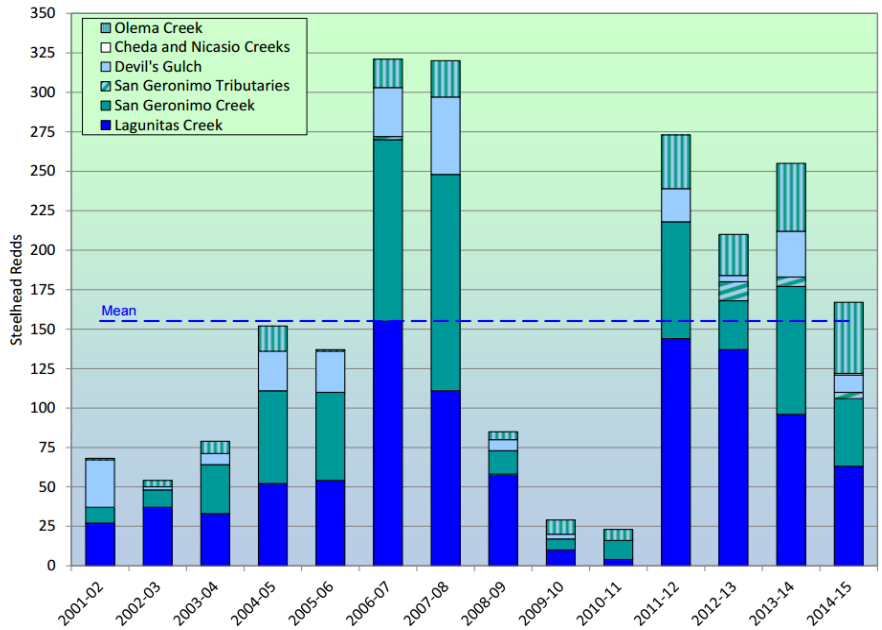


Figure 4. Steelhead redds, Lagunitas Creek watershed 2001-15. MMWD 2015, Fig. 3, pg. 17.

For San Francisco Bay tributaries, information on steelhead numbers is largely lacking, and these small, isolated populations are at a high risk of extinction due to the prevalence of impassable dams and barriers, reduced flows, and degraded water quality from runoff and pollution (NMFS 2016). Most existing information comes from opportunistic sampling or observations. For example, a public sighting of two adult steelhead was confirmed below an impassable public transportation weir in Alameda Creek, the first confirmed sighting in that watershed since 2008 (*East Bay Express* 2016). CDFW has not documented spawning or rearing of steelhead in southern San Francisco Bay tributaries since the drought began in 2012 (M. Leicester, CDFW, pers. comm. 2016). In the northern portion of the bay, the Napa Resource Conservation District has used rotary screw trap data to estimate that between 100 and 1000 juvenile steelhead exit the Napa River watershed each year (J. Koehler, NRCD, pers. comm. 2016). However, 2014 smolt totals trapped per day dropped significantly, possibly due to the ongoing drought and reduced trapping efficiency at lower flows (Figure 5).

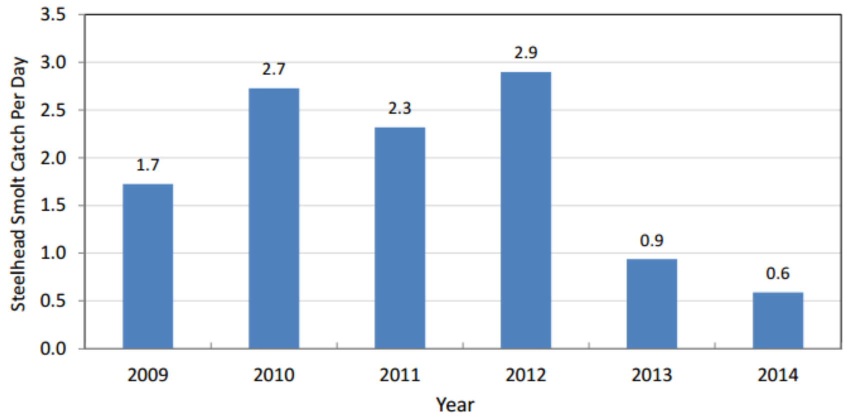


Figure 2. Steelhead smolt catch-per-unit-effort (CPUE) in the Napa River RST from 2009-2014.

Figure 5. Rotary screw trap catch per unit effort of steelhead smolts in the Napa River, 2009-2014. From: NRCD 2016 Figure 2, pg. 4.

In San Mateo County, Pescadero Lagoon provides important wetland habitat for juvenile steelhead, including larger smolts that contribute disproportionately to the number of returning adult spawners. However, since 1995 this habitat has become the site of nearly annual fish kills when the lagoon breaches in late fall or early winter each year (CDFW 2014). With historic drought in California from 2012-2016, the lagoon has either not breached at all or breached for such a short time that passage of adult and juvenile steelhead to the ocean was not possible. In 2013, state, federal, and public partners gathered to rescue 54 adult steelhead trapped in the closed lagoon, and transported them over the sandbar into the ocean (NMFS 2013). In 2014, biologists estimated that all of the 170 mostly two-year-old steelhead smolts in the lagoon perished that summer (CDFW 2014). These conditions were similar to those observed in nearby San Gregorio and San Lorenzo (Santa Cruz Co.) lagoons. Drought may compound poor water quality conditions by trapping juveniles and post-spawn adults in lagoon habitats throughout the region. In 2015, biologists counted dozens of dead juvenile steelhead smolts in anoxic water in Pescadero Lagoon, prompting calls for further restoration study (*San Jose Mercury News* 2016).

CCC steelhead have also steadily declined in abundance south of the mouth of San Francisco Bay. YOY sampling on Gazos Creek in Half Moon Bay has revealed a downward trend since 2000 (Smith 2016). Waddell Creek, a potentially independent population in Santa Cruz County, averaged about 500 adults between 1933 and 1942 (Shapovalov and Taft 1954). In the San Lorenzo River in downtown Santa Cruz, creel surveys in 1953-54 ranged from 1,895-5,645 steelhead caught by anglers to between 1,035-1,816 captured between 1970 and 1973 (Johansen 1975). The most recent estimates for Waddell Creek are less than 100 adults per year, with recorded YOY densities less than 50% of 1995-1998 averages (93 YOY/30m) every year since 1999 (Smith 2016). In the San Lorenzo River, the recent average hovers around 500. San Vincente, Scott, Soquel, and Aptos Creeks all average below 200 fish annually, but sampling efforts are few and far between. Only Scott Creek currently has a full life cycle monitoring station in place to collect data on steelhead and coho salmon. Juvenile sampling of YOY juveniles since 2002 (Smith 2016) and recent adult returns (TNC and NMFS 2016) show a continuing downward trend, which is due in part to the ongoing drought (Smith 2016, Figure 6).

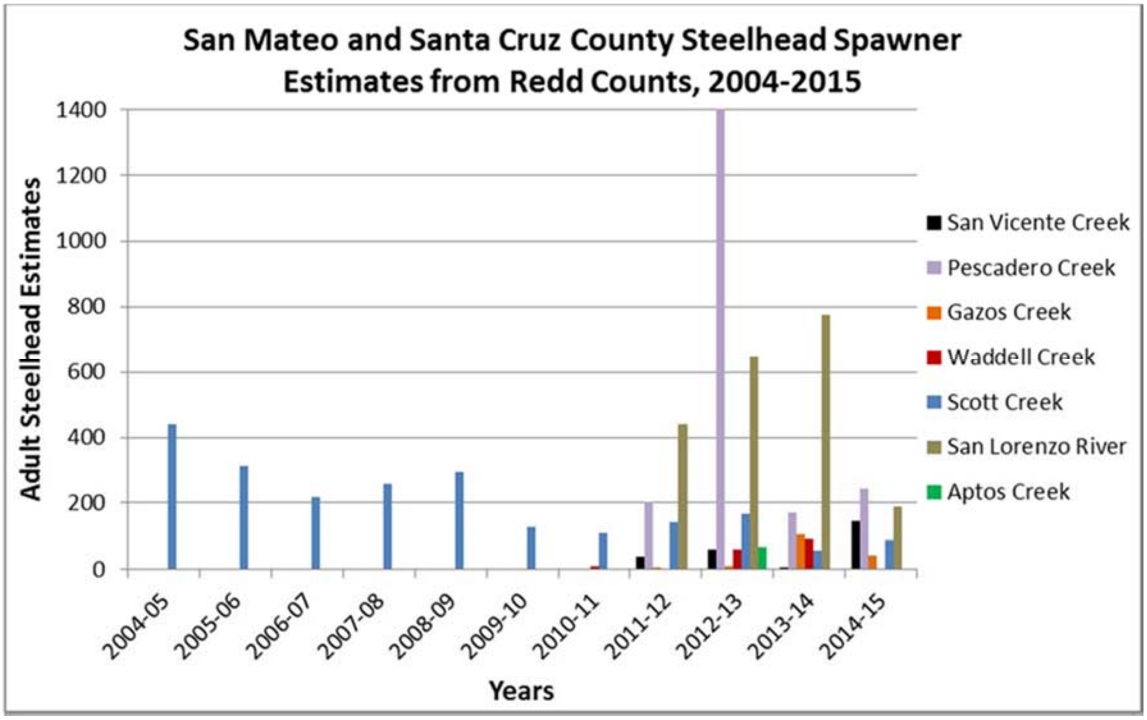


Figure 6. Steelhead spawner estimates based on redd surveys conducted by CDFW; data compiled by The Nature Conservancy and NMFS. From: D. Logan, NMFS 2016.

From 1999 to 2013, students on the Monterey Bay Salmon and Trout Project (Santa Cruz Co.) at Kingfisher Flat Hatchery on a tributary to Scott Creek released 33,840 steelhead smolts to supplement low populations. This program has been shuttered since 2013/2014 (NMFS 2016).

Factors Affecting Status: Small populations of steelhead still occur in watersheds throughout the DPS range, but they are limited by dams and other barriers, degradation of existing habitat through land use practices, hatcheries, and other factors. The cumulative and synergistic effects of these threats make it difficult for CCC steelhead populations to contend with ongoing drought and fluctuating ocean conditions, and limit their ability to recover in the face of climate change.

Dams. Across the CCC steelhead DPS, dams have significantly reduced the amount of accessible habitat for juveniles and adults and reduced streamflows. For the DPS as a whole, 22% of historical habitat is estimated to be blocked by human-made barriers (Good et al. 2005). In the Russian River, Coyote and Warm Spring dams both block historical spawning and rearing habitat. Dry Creek has lost 56% of its habitat, Mark West Creek (7%), and the upper Russian River (21%) (Spence et al. 2007). In the San Francisco Estuary, approximately 58% of historically occupied streams no longer support anadromy, although presumably related resident populations do exist in many headwaters (Leidy et al. 2005). Watersheds around San Francisco Bay that have lost habitat include: Novato Creek (22%), Napa River (17%), Walnut Creek (96%), San Pablo Creek (72%), San Leandro Creek (80%), San Lorenzo Creek (48%), Alameda Creek (95%), Coyote Creek (49%), Guadalupe River (21%), Stevens Creek (54%), San Francisquito Creek (33%), and San Mateo Creek (83%). In Tomales Bay, accessibility is also a problem in Lagunitas Creek (49%) and Walker Creek (26%).

Steelhead tend to rear and spawn in smaller headwater tributaries in upper portions of watersheds, and so regaining access to this habitat is critical for recovery of the DPS (Bjorkstedt et al. 2005). None of these dams allow passage, and what water is allowed to flow through them is often reduced in quantity and quality from what it would be without dams in place (NMFS 2016). Whether for flood control, water storage, or hydropower generation, dams trap sediment and lead to loss of instream/riparian habitat for steelhead (NMFS 2016). Dams also dramatically change the hydrograph of the streams on which they occur, with larger dams especially removing peak flows that bring steelhead in from the ocean to spawn and help propel juveniles downstream in spring. Related diversions typically reduce summer flows, reducing habitat and increasing water temperatures, and make it more difficult for steelhead to survive through the warmer summer months.

Dams also play a major role in sustaining remaining downstream habitat for steelhead. For example, in the Russian River, releases from the Eel River and Mendocino Reservoir for downstream urban and agricultural diversion may actually increase summer flows in places. While dam operations perpetuate extinction risk in many CCC steelhead populations by restricting access to habitats, they may also regulate flow and water temperature in summer and fall months and offer suitable rearing habitat immediately downstream (M. Leicester, CDFW, pers. comm. 2016, NMFS 2016). The Russian River is the first National Oceanic and Atmospheric Administration (NOAA) Habitat Focus Area to coordinate restoration efforts among federal, state, and private entities for the benefit of all salmonids, and incorporates adaptive management to allow flushing flows to benefit ESA-listed coho and Chinook salmon as well as steelhead (NMFS 2016). Downstream of the San Francisco Bay rim dams, tributaries such as Alameda, Coyote, and San Francisquito creeks have some flow directly downstream of dams, dry middle reaches in urban areas, and accreted flow in lower sections. These disturbed watersheds offer very small windows of opportunity to allow adult passage, but little opportunity for rearing or getting smolts back into ocean through dry-back zones. Even with adjusted dam operations to allow strategic adult attraction flows in the fall and pulse flow releases to benefit juvenile outmigration in the spring, recovery of CCC steelhead in San Francisco Bay tributaries is daunting with so many dams in place (M. Leicester, CDFW, pers. comm. 2016).

Urbanization. Degradation of habitat in most watersheds is a significant threat to CCC steelhead through urbanization, expansion of vineyards and other agriculture, road building, logging, mining, sewage discharge, and other actions. For instance, numerous tributaries and the mainstem Russian River are currently listed as impaired water bodies under the Clean Water Act due to high levels of sedimentation, aggravated water temperatures, presence of pathogens, and poor water quality. Urban development throughout the DPS, especially in San Francisco Bay tributaries, intrudes on floodplains and destroys riparian habitat, especially on estuarine wetlands and riparian corridors, which contribute to reductions in water quantity and quality. Without societal efforts to reduce water usage in urban and agricultural areas, critical over-summering habitats will not be available for juvenile steelhead. This will reduce the life history diversity expressed by juveniles in the CCC steelhead DPS and increase their susceptibility to climate change. In the San Francisco Bay area, CWA-listed impaired watersheds include the Guadalupe River, San Francisquito, Stevens, and Sonoma creeks as well as the Petaluma and Napa Rivers.

Logging. Historical and ongoing timber harvest in Santa Cruz County watersheds continue to have legacy impacts on steelhead habitat such as sedimentation as a result of the associated roadbuilding that occurred in mountainous areas (NMFS 2016). Logging practices have led to the CWA listing of San Mateo County coastal steelhead creeks (Pomponio, Pilarcito, and Pescadero) for poor water quality. Legacy impacts of logging include removal of old-growth trees, which provide important cover for adult and juvenile salmonids. Large wood in streams provide important habitat features for steelhead, yet throughout the CCC steelhead DPS, logjams continue to be removed due to concerns over flooding and recreational hazards. Because significant portions of the CCC steelhead DPS are heavily developed and riparian areas are being lost, woody debris removals are reducing cover and pool formation, and increasing conditions unfavorable to juvenile steelhead.

Estuarine alteration. CCC steelhead are unusually dependent on the estuaries (lagoons) at the mouths of their streams for growth and survival at all life stages. These habitats are shrinking as they fill with sediment from upstream land uses and are encroached upon by urbanization and agriculture. For example, over 90% of critical wetland and estuarine habitat in the San Francisco Bay region has been drained and/or filled (NMFS 2016). This results not only in less habitat, but shallower, less complex (increased vulnerability to predators), and warmer habitat that is increasingly vulnerable to pollution events and drought. In addition, the natural summer sand barriers are frequently artificially breached, resulting in sudden draining of lagoons and large-scale reduction in habitat quantity and quality (Moyle and Smith 1995). For example, CDFW has been studying the impacts of low freshwater inflows, agricultural pollutant inputs, and reduced dissolved oxygen on the fish communities in Pescadero Lagoon for years (CDFW 2014). Agency investigators found that the duration of lagoon bar closure and marsh inundation, as well as reductions in freshwater inflows to the lagoon all play a role in contributing to the extent and duration of anoxic conditions at breaching, which have led to fish kills in most years since 1995 (CDFW 2014). These stressors have likely been exacerbated by the ongoing drought in California, which reduces inflows and leads to earlier bar closure than under average water years, trapping adults and juveniles alike. Highways impacts many estuaries in the CCC steelhead DPS through direct pollution, channelization via bridge and roadway construction, erosion, and sedimentation.

Agriculture. While mostly urbanized, CCC steelhead DPS watersheds have been heavily impacted by agricultural water diversions, especially in the northern portion of the range (Sonoma, Marin, Napa, and Solano counties). Cattle ranches, dairies, vineyards, orchards, and other agricultural users have placed high demands on limited water supplies through diversions and groundwater pumping, which have led to habitat reduction, degradation, and fragmentation, and non-point pollution. Return flows from agricultural uses to streams are often warmer and contain organic compounds that reduce the suitability of habitat for salmonid survival. Irrigated agriculture, ranches, dairies, municipalities, and other water users throughout the Napa and Sonoma Valleys have ditched, channelized, diverted and pumped water over time to support the burgeoning population and wine industry (M. Leicester, CDFW, pers. comm. 2016). South Bay municipalities draw out groundwater to support agricultural and other human uses in Santa Clara and Alameda counties, which reduces the supply of cool, subsurface flow into small, critical steelhead rearing streams such as tributaries to Coyote Creek and the Guadalupe River.

Hatcheries. There are currently two large artificial propagation programs for CCC steelhead: the Warm Springs Hatchery (Dry Creek, Russian River) and upstream Coyote Valley Hatchery (East Branch Russian River) (CDFW 2016). While the stated goals of the Warm Springs and Coyote Valley hatcheries are to contribute to future abundance and spatial structure in the CCC steelhead population. Due to the low number of wild spawners remaining in the limited available natural habitat left below dams, it is more likely that domestication selection is reducing genetic diversity and effective population size in the Russian River watershed with potentially negative effects on the remaining wild populations as well, through interbreeding with 'stray' hatchery fish. The influence of past frequent plants of hatchery steelhead from out-of basin is not well understood but is probably minimal. Currently, NMFS is undertaking a Hatchery Genetic Management Plan review for both active programs to try to understand and reduce these impacts on the genetic integrity of fish in the wild (NMFS 2016).

In addition, the Kingfisher Flat Hatchery, located on Scott Creek, operated during 1982-2014. This facility, which was run by a the Monterey Bay Salmon and Trout Project, collected wild coho salmon and steelhead from Scott Creek and the San Lorenzo River for spawning and rearing. It was closed down in 2014. It effects, positive or negative, on the steelhead population are not known.

Harvest. As a DPS with Threatened status under the Endangered Species Act, CCC steelhead must be kept in the water and released unharmed if accidentally caught. Therefore, harvest is low, and fishing closures on small coastal watersheds and San Francisco Bay tributaries such as Sonoma Creek helped to reduce stress on wild fish during the exceptionally dry years of 2014-2015. On the Russian River, where hatcheries support what has become a popular but essentially put-and-take fishery, anglers can harvest two hatchery-origin steelhead per day that have had their adipose fins clipped. Some inadvertent mortality on wild fish while legally fishing is likely, although poorly understood in the Russian River. Commercial fisheries rarely catch steelhead at sea (Harding 2015).

Predation. Since the CCC DPS has such low abundance, natural rates of predation may significantly impact small, fragmented populations, making recovery more difficult. For example, NMFS researchers estimated that higher numbers of sea lions at the mouth of the Russian and San Lorenzo rivers likely cause significant predation on CCC steelhead, and that western gulls or even raccoons may have higher rates of predation on juvenile steelhead in recent low water years than is typical (NMFS 2016). Degraded habitats resulting from myriad anthropogenic stressors can favor alien species over natives, which can increase predation pressures on juvenile steelhead from species such as basses (*Micropterus* and *Morone* spp.), especially in lower reaches of watersheds (Leidy 2007).

Status Score = 2.0 out of 5.0. High Concern. Most CCC steelhead populations are very low in abundance (e.g., San Francisco Bay tributaries) and far below recovery thresholds, though there is not enough information to determine if their extinction risk has changed since the last NMFS status update in 2011 (NMFS 2016). Throughout most of the DPS range, populations are now small enough to be susceptible to stochastic events, and are at risk of permanent shifts in population dynamics and genetics (NMFS 2016). Every indication is that downward trends in all populations will be accelerated by climate change. CCC steelhead were listed as a threatened species on August 18, 1997; their status was reaffirmed for fish downstream of impassable barriers in 2006 (71 *Federal Register* 834, Jan. 5, 2006, NMFS 2006). They have no special status in California except as a sport fish with limited take for hatchery-marked fish. The NMFS draft recovery plan states that CCC steelhead have a low potential for recovery due to the demands of transportation, land use practices, and urbanization, which are not likely to be reduced in the heavily-populated DPS range. The plasticity of life history strategies observed in CCC steelhead will likely guarantee their persistence in the far reaches of larger watersheds they inhabit, but it is likely extirpation of steelhead from most currently occupied watersheds will occur over the next 25-50 years unless large-scale restoration actions are coordinated and implemented.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of California Central Coast steelhead. Factors were rated on an ordinal scale, where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Major dams on the Russian River and many other small dams and diversions present throughout DPS.
Agriculture	Medium	Diversions for agriculture likely limit population sizes, especially at the margins of the DPS boundaries.
Grazing	Low	Some impacts in rural portions of CCC steelhead range.
Rural residential	Low	Diversions for rural residential use likely impact rearing and oversummering habitat for juveniles.
Urbanization	High	The CCC DPS range is heavily urbanized and increasing demands for limited water degrade its quantity and quality.
Instream mining	Low	Gravel mining still occurs regularly on the Russian River.
Mining	Low	Heavy metal contaminants from historical mining reduce the suitability of estuaries and lagoons for juvenile rearing.
Transportation	Medium	Nearly every watershed in the DPS is crossed by major roads and highways, often several times.
Logging	Low	Mostly legacy impacts of timber removal that leave streams devoid of valuable large wood cover.
Fire	Low	Not a major threat to coastal streams or Bay tributaries.
Estuary alteration	High	Every estuary and lagoon in the DPS has been significantly altered by urbanization and roadbuilding.
Recreation	Low	Recreation may negatively impact behavior of low numbers of steelhead but are unknown.
Harvest	Low	Harvest of wild CCC steelhead is prohibited.
Hatcheries	Medium	Effects of hatchery steelhead on wild populations is not known but probably negative.
Alien species	Low	Predation by alien species may be significant in San Francisco Bay and the lower Russian River watershed, especially during periods of drought.

Effects of Climate Change: The impacts of a changing climate will exacerbate the decline of CCC steelhead primarily through reductions in the temporal and spatial availability and accessibility of usable habitat. Moyle et al. (2013) rated this fish as "highly vulnerable" to the effects of climate change, likely leading to extinction by 2100 if present trends continue and their range or populations decline. The reasons for this evaluation are multiple and complex.

Climate change is expected to increase demands for existing water supplies in California, as well as increase remaining water temperatures beyond lethal limits, especially during summer and fall months. As temperatures rise and precipitation patterns shift, steelhead will be exposed to greater periods of higher water temperatures and more flow variability, eventually outpacing their remarkable ability to alter the timing of life strategy transitions to avoid such exposure (Wade et al. 2013). In general, climate modeling suggests that regions in lower latitude and with lower elevation will be subject to the greatest increase in duration and intensity of high air and water temperatures (Wade et al. 2013). The CCC steelhead occur at very low elevations, near the southern edge of steelhead range; there are not many potential headwater habitats at higher elevations accessible to CCC steelhead to reduce their exposure to higher temperatures without removal of dams. Fortunately, most watersheds are coastal or tributary to San Francisco Bay, which may help moderate fluctuations in air and water temperature more than inland watersheds. Providing access to a variety of habitats to allow expression of all life history strategies remains the best approach for building resiliency to climate change in populations (Wade et al. 2013).

In saltwater, where steelhead spend a majority of their lives feeding and growing, climate change is likely to increase sea surface temperatures and ocean acidity, contribute to loss of habitat through sea level rise, change the timing and strength of upwelling currents that provide prey, and generally lower marine productivity and salmonid survival off of California's coast (NMFS 2016). The poor ocean productivity observed over the last few years off California's coast associated with El Nino events probably reduced survival of freshwater, estuary, and marine life stages across the DPS by reducing foraging success, and similar impacts are expected if sea surface temperatures rise off California (NMFS 2016). Changes in ocean productivity generally manifest themselves in low numbers of returning adult steelhead in a few years when mostly age-3 and 4 fish return to freshwater to spawn.

Where land and sea meet, a changing climate will likely reduce freshwater inflow to estuaries, which are critical nursery areas for CCC steelhead, altering nutrient cycling and sediment transport (NMFS 2016). Climate change is also likely to cause more frequent and more intense drought, and a reduction in usable estuary habitat due to sea level rise. With low flows, remaining water in estuaries becomes too warm and oxygen-poor to support juvenile steelhead growth and survival, and gives rise to algae blooms that further deplete oxygen at night. Monitoring coastal lagoons for juvenile survival and conducting adult fish rescues when they are stranded in low water years must be continued and expanded to support small and stressed populations of CCC steelhead. Recent restoration projects in San Francisco Bay, coastal lagoons in Pescadero Creek and San Lorenzo River watersheds are improving habitat conditions, but drought impacts continue to threaten these populations with extirpation (CDFW 2014).

California's historic, extreme drought (2012-2016) has highlighted poor water management across the region. In addition to providing some of the lowest precipitation on record for California, the ongoing drought has also been coupled with several of the hottest years ever recorded. The result of these combined forces has revealed that water storage systems along California's Central Coast are inadequate. As drought has continued, reservoir storage has become depleted and caused release of highly turbid, warm water downstream, which has degraded spawning and rearing habitat (NMFS 2016). As a direct result of drought, groundwater withdrawals throughout the DPS range have impaired the volume, timing, extent, and temperature of surface flows in streams, and San Francisco Bay tributaries have been lowered and even run dry due to over-drafting of aquifers (NMFS 2016). The high rates of groundwater pumping that have occurred to make up the deficit of surface water deliveries have starved watersheds of critical cooling hyporheic flows in the summer and fall months. In addition, coastal lagoon habitats have been cut off from the ocean much sooner than in average water years, subjecting juveniles and trapped post-spawn adults to lethal water quality in many areas

(CDFW 2014). Illegal water diversions for marijuana cultivation have been especially stressful to salmonid populations over the current drought, as the plants require large volumes of water to grow during the warmest months. These impacts are likely to continue until surface waters are managed differently and groundwater pumping is regulated and allows aquifers to recharge over time (NMFS 2016).

Table 2. Metrics for determining the status of Central California Coast steelhead, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Multiple watersheds occupied in California but very few viable populations still exist.
Estimated adult abundance	2	The Russian River contains > 1,000 spawners annually, with smaller contributions from other populations, but numbers are declining and supported by hatcheries.
Intervention dependence	2	Habitat restoration and barrier removal are critical to increasing habitat availability.
Tolerance	4	Able to adapt to live in freshwater and estuarine environments.
Genetic risk	3	Widespread but populations increasingly fragmented and isolated, with potential for interbreeding with hatchery fish.
Climate change	1	Extremely vulnerable due to limited access to habitat and cumulative effects of other factors (e.g. urbanization) dams, etc.).
Anthropogenic threats	1	3 High, 3 Medium factors.
Average	2.0	14/7.
Certainty (1-4)	3	Hard numbers are few but status is fairly certain.

Management Recommendations: The CCC steelhead DPS continues on a downward trajectory as the result of being centered in a highly urbanized landscape, with many competing land uses and pressures on limited water resources. Impacts from dams, agriculture, urbanization, logging, hatcheries, and other factors pressure small, fragmented populations in different ways across the large and diverse DPS range. In general, most central coast watersheds are neglected and not prioritized for monitoring or restoration funding unless they also contain coho salmon (e.g., Lagunitas and Scott creeks) (M. Leicester, CDFW, pers. comm. 2016). In the face of cumulative impacts from increasing anthropogenic stressors, including climate change, providing connectivity among diverse, high-quality habitats from headwaters to sea will be critical to the persistence of the CCC steelhead DPS.

Both local and widespread habitat improvements and land use changes are needed across all scales, such as additions of large wood into streams, maintenance of adequate riparian buffers, and limitations on sediment and other pollutants flowing into streams to cumulatively benefit recovery of CCC steelhead. Restoration guidelines have been developed by NMFS and other management partners for bank stabilization, road maintenance, and instream gravel mining projects to help minimize impacts of land use in this heavily urbanized DPS range. To enhance summer and overwintering survival of CCC steelhead, improvement in stream habitat complexity, as well as recruitment and retention of large wood, is important across the DPS. The

California Forest Practices Rules are currently inadequate for protection of riparian habitat, which shades streams, limits sedimentation, and provides large wood into streams. Actions that enhance riparian and upslope habitats will increase food supplies for juvenile steelhead, decrease siltation into the stream, and reduce solar exposure of streams, especially during warmer months.

In addition, headwaters must be protected and connected to downstream areas subject to summertime dry-back, and managed in a coordinated way to maximize benefits of conservation and restoration activities that increase resiliency to climate change (Wade et al. 2013). Regulating releases from existing dams to mimic natural flow regimes must be instituted where possible to benefit all life stages of salmonids. At a broad scale, public education and engagement should be prioritized so citizens are made aware of the plight and distribution of CCC steelhead near where they live and work. For example, along Adobe Creek, tributary to the Petaluma River, a local school works with CDFW to incorporate an egg rearing and juvenile stocking program into their curriculum. Where practicable, these programs have the potential to help connect citizens better understand their impacts on fish species and help protect them.

Across all populations in the DPS, more monitoring will be essential to understanding baseline population dynamics and how recovery actions can be implemented to support CCC steelhead. Stalling implementation of CDFW's statewide coastal salmonid monitoring program prevents collection and use of comprehensive abundance and trend information for management. Reliable assessments of steelhead-bearing streams and estuaries are severely lacking throughout the DPS currently, and hinder recovery efforts. Toward that end, CDFW should continue to integrate data collection with partners to streamline statewide data collection and management, obtain reliable information on core and peripheral populations, and help managers understand how and when specific habitats are used that allow expression of diverse life histories. This information could inform creation of an adaptive Fisheries Management and Evaluation Plan (FMEP) that prioritizes and defines criteria for taking specific actions to benefit recovery of the DPS. For example, at the northern edge of their range in the Tomales Bay Watershed (Walker, Lagunitas, Olema creeks), there is documented steelhead usage of main-stream habitat, but they probably use smaller tributaries opportunistically in wet years. Likewise, interior bay watersheds such as Suisun, Corte Madera, and Novato creeks are not monitored at all currently. These peripheral habitats require protection because they can play important role in providing habitat in the future (R. Watanabe, CDFW, pers. comm. 2016). Basic stream gaging, and stream connectivity studies should be undertaken to help understand the habitats available to these populations. Until more basic monitoring is carried through, several important questions with management implications, such as abundance trends in priority watersheds, how and when steelhead use San Francisco Bay habitats, where, when, and why do steelhead and coho salmon co-occur in the DPS, how hatchery operations and stocking practices impacting wild steelhead, and which barriers to migration should be removed, will be left unanswered (NMFS 2016).

CDFW, National Parks Service, NMFS, other management agencies, and private citizens have identified specific actions that will benefit CCC steelhead but need to be coordinated and implemented. The reality in Central California is that ESA-Endangered coho salmon are the number one priority for funding and projects, but there is potential to get piggyback benefits for steelhead through coho restoration projects. Wherever possible, monitoring and restoration that benefits all native salmonids, and not only coho, should be prioritized. For starters, management partners can work together to ensure that there is enough water left in streams during critical oversummering periods: there are several tools available to accomplish this. First, AB 2121 minimum flow requirements can protect flows for salmonids downstream of dams. Second, Fish and Game Code 1537, which mandates that dam operators keep fish in good condition downstream, is being wielded currently in court on water operations on the Guadalupe River in San Jose, and shows promise for setting precedent for flow restoration on other San Francisco Bay rim dams. Third, the recent Sustainable Groundwater Management Act (SGMA), which requires creation of groundwater management plans that account for and ensure recharge of aquifers, should also be aggressively implemented. Currently, several CDFW-funded projects statewide are focused on promoting voluntary options storage on private property, switching from direct diversion in summer/fall months to using storage tanks to collect water during wet seasons. NMFS and CDFW developed the Voluntary Drought Initiative Program (VDI) to help incentivize water conservation, fish rescue, and flow augmentation in Green Valley Creek, Dutch Bill Creek, Mill Creek, Mark West Creek, and Porter Creek over 1,900 acres of vineyards in the Russian River Valley, reducing water demand by 25% over 2013 levels (NMFS 2016). The 2014 Russian River Frost Protection Regulation controls harmful stream stage changes during low-flow fall periods for frost control over 3 years, and will likely improve fry survival across watersheds in Napa and Sonoma Counties. Other projects focus on providing subsistence flows in streams, and while they are mostly voluntary, they have seen some success and should be replicated wherever possible. Recovery of the CCC steelhead DPS depends upon such partnerships between agencies, municipalities, and private landowners to coordinate conservation activities.

Restoration of headwaters, active channels and floodplains, and estuaries/lagoons that encourage life history diversity yield resiliency to threats such as climate change can be supported in populations by providing habitat complexity (NMFS 2016). Replacement of the Highway 1 bridges over Scott and Waddell creeks offers the potential to restore residual lagoon depth, which can enhance rearing habitat and increase juvenile survival (Smith 2016). Illegal lagoon breaching, which has occurred on several CCC watersheds since 2008, should be fiercely prosecuted. Barrier removal projects that benefit native species should also be expedited where possible. The town of Saint Helena is slated to remove a long-standing earthen dam that has blocked passage for many adult steelhead each year from valuable spawning and juvenile rearing habitat. The U.S. Forest Service's Forest Practice Rules need to be updated to reduce impacts of decommissioned and active logging roads to reduce sedimentation and siltation and help contribute large wood to streams to benefit all salmonid life stages. In addition to re-introducing wood to watersheds, snags and logjams must be monitored so they do not turn into passage barriers themselves, as has happened on Gazos and Scott creeks recently (Smith 2016).

Next, CDFW can implement the new Hatchery and Genetics Management Plan for the Warm Springs/Coyote Valley hatcheries using the best available science. The most recent findings suggest that the current DPS boundaries are not sufficient for conservation or management of coastal steelhead in California; they should be updated to account for gene flow, population dynamics, and opportunities for future adaptation and persistence (Garza et al. 2014). In addition to minimizing the impacts of hatchery fish on the genetic integrity of wild steelhead within the DPS, NMFS and CDFW can enforce low-flow fishing closures and regulation changes throughout the DPS to minimize fishing impacts during times when juveniles and adults are highly susceptible. Last, all management partners can work with law enforcement to implement provisions in Proposition 64 to minimize impacts of marijuana cultivation on fish.

Finally, the heavily altered San Francisco Bay lies at the heart of the DPS and remains integral to efforts to meaningful recovery. Restoration of Bay salt ponds and tidal marshes has increased recently, with massive projects such as the South Bay Salt Pond Restoration Project currently underway in Alviso, Ravenswood, and Eden Landing, and myriad projects to benefit wildlife, such as migratory birds, surrounding San Pablo Bay. In the South Bay, biologists have been tagging and tracking juvenile steelhead in the Guadalupe River watershed since 2013 to assess former salt pond habitat functionality. In an effort to reduce toxicity associated with methyl mercury creation and entrainment in former salt ponds and degraded marsh habitats that can be lethal to wildlife, researchers have breached tide gates for extended periods of time to allow steelhead and other species volitional passage between brackish waters of the bay and more freshwater environments further upland in the watershed. As large-scale restoration of San Francisco Bay continues and expands to benefit native species, passage and habitat suitability for juvenile and adult CCC steelhead should be incorporated and prioritized.

CENTRAL VALLEY STEELHEAD

Oncorhynchus mykiss irideus

Moderate Concern/Low Concern. Status Score = 3.0/4.6 out of 5.0. This taxon, which includes resident trout, is in little danger of extinction but current Central Valley stream conditions appear to favor resident life histories over anadromous life histories. Over time, life history diversity could be lost increasing extinction risk for the taxon as a whole. The first score only reflects the status of the steelhead life history, including both hatchery and wild fish, while the second score reflects the potential for maintaining large, highly adaptable rainbow trout populations that include steelhead as a life history option.

Description: Steelhead and rainbow trout vary greatly in color and body shape (Moyle 2002). Juvenile trout display 5-13 oval parr marks centrally located along the lateral line, with interspaces being wider than the parr marks. The color of the dorsal and anal fins ranges from white to orange, and there is little or no spotting on the slightly forked tail. The head is blunt with a short jaw that does not extend past the eye. Historically, adult CV steelhead rarely exceeded 60 cm FL. Newly arrived adults appear silvery, sometimes showing an iridescent pink to red lateral line, and have a square-shaped caudal fin with radiating spots, which is unlike other salmonid species within the Sacramento-San Joaquin Rivers. Many small, black spots also cover the back, adipose, and dorsal fins. As adults remain in freshwater they darken, taking on the greenish hue on the back and the iridescent pink and red sides characteristic of resident rainbow trout. The scales are small, with 110-160 pored scales along the lateral line. Basibranchial teeth are absent, with 16-22 gill rakers on each arch and 9-13 branchiostegal rays. Steelhead typically have 10-12 primary dorsal fin rays, with 8-12 primary anal rays, 9-10 primary pelvic rays, and 11-17 primary rays making up the pectoral fin.

Taxonomic Relationships: Central Valley (CV) steelhead are part of the coastal rainbow trout complex in the Central Valley and are broadly defined as anadromous rainbow trout from the region. However, the CV steelhead does not hold up well as a distinct taxonomic unit.

NMFS (1998) found that CV steelhead formed an Evolutionary Significant Unit (ESU) that was genetically distinct from the Central Coast ESU, which includes fish found in tributary streams to San Francisco Bay. The ESUs also included non-anadromous rainbow trout, so to "clarify" the situation the ESU designation was changed in 2005 to a Distinct Population Segment (DPS), which only listed threatened anadromous (sea-run) individuals (NMFS 2016). To further complicate the listing designation, only fish "originating below" dams and other barriers are counted. Resident trout are not included as part of this DPS and are therefore not listed under the ESA even where they interbreed with sea-run fish. All *O. mykiss* above Central Valley reservoirs are also excluded from the DPS, even steelhead-like forms (adfluvial rainbow trout) with spawning migrations into tributary streams and which use reservoirs like below-dam steelhead use the ocean. These migratory fish would presumably be part of the DPS if the dams did not exist. The changes happened despite the official DPS Policy, which states that a group of organisms form a distinct population segment if it is "markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, and behavioral factors." [61 *Federal Register* 4722 (Feb. 7, 1996)]. A basic problem is that CV steelhead are not "markedly separated" from resident rainbow trout. For more explanation of distinctions between an ESU and a DPS, see the Northern California winter steelhead account.

Curiously, the CV steelhead DPS officially includes steelhead from the Feather River and Coleman National fish hatcheries. "NMFS determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the DPS" (NMFS 2016, p 4.). NMFS excluded fish reared in the Nimbus Fish Hatchery on the American River because these fish had been heavily hybridized with fish from the North Coast, primarily Eel and Mad rivers. These fish were brought to the hatcheries to increase adult size and improve angler satisfaction. The hatchery plans to eventually stop rearing the mixed strain fish and replace them with steelhead closer to the native strains, perhaps from above-dam populations (NMFS 2016). Mokelumne Hatchery recently switched from raising fish from the American River Hatchery to using Feather River Hatchery stocks prompting NMFS (2016) to recommend including these fish in the DPS as well.

In contrast to patterns typical of coastal steelhead, a recent genetic study found that Central Valley *O. mykiss* above and below dams within the same watershed were not each others' closest relatives. Most below dam samples clustered closely with hatchery steelhead (Pearse and Garza 2015). The authors also found no relationship between genetic and geographic distance among below-dam populations suggesting that a long history of translocations of hatchery trout and steelhead and pervasive habitat alteration have effectively "scrambled" populations creating a single heavily hatchery-influenced below-dam population.

Central Valley steelhead is not a well-defined taxon. The mish-mash of which populations and life history forms to include or exclude from the DPS comes with some curious assumptions, namely that: (a) the resident and anadromous forms do not significantly interbreed, (b) that steelhead from various hatcheries do not stray and mix, (c) that steelhead-like fish above reservoirs have diverged significantly from steelhead below the rim dams so they no longer serve as part of the DPS, and (d) decades of widely planting resident rainbow trout of hatchery origin (multiple strains) have not influenced the genome of the DPS. Faults with each of these assumptions are addressed below.

Resident vs. anadromous. There is a conundrum in the relationships among steelhead and resident rainbow trout (Kendall et al. 2015). In general, the two forms are complementary because anadromous trout can give rise to resident trout and vice versa on a regular basis and the flexible life history has a strong genetic basis (Pearse et al. 2014). Recent research shows that anadromy is a polygenic (multi-gene) trait in rainbow trout with moderate heritability. This means that a large population of resident rainbow trout, such as the one in the lower Sacramento River, can continue to have individuals that express anadromy even if there is selection against behavior in many years (M. Miller, UC Davis, pers. comm. 2017). But the anadromy trait can also be maintained at low levels through the effects of environmental variation. If survival is low in the ocean or during down-stream migration for an extended period, then resident trout will have an adaptive advantage. If conditions in the ocean promote high survival and growth, then progeny of high-fecundity steelhead females will do well in fresh water. Resident trout can also thrive above natural barriers (e.g., landslides) and steelhead can recolonize streams from which resident fish have been eliminated from natural causes (e.g. volcanic eruptions). According to NMFS (2006) "It is unclear how long an *O. mykiss* population can persist if [it is] dependent entirely upon the productivity of resident fish in a dynamic freshwater environment, even if the resident forms are abundant (*Federal Register* 71(3), p 844.)."

Pearse et al. (2014) indicate that residency has a genetic basis reflecting strong selection pressure against anadromy in populations with little contact with sea-run fish. Conversely, in southern California many, if not most, returning "steelhead" likely originate as migratory smolts produced from resident headwater trout populations many of which persist *above* man-made and natural barriers to anadromy. The polygenic nature of the anadromy indicates that the trait can persist for a long time in a large resident population. This has been demonstrated in an Argentina river flowing to the Atlantic, where steelhead have developed from resident fish, apparently of California origin, with resident and migratory fish forming one interbreeding population (Pascual et al. 2001). In some cases, such as in the Calaveras River, the anadromous forms may be mainly female, while males remain as resident fish (McEwan 2001).

All Central Valley Hatchery steelhead are adipose fin clipped and it appears that a majority of steelhead, or at least of steelhead caught by anglers, are of hatchery origin. Steelhead resulting from natural spawning make up only a small percentage of fish returning to hatcheries or nearby streams in the Central Valley (NMFS 2016). Resident populations in tailwaters (e.g. in Sacramento River between Keswick dam and Red Bluff) generally contain few fish that had a steelhead mother (Zimmerman and Reeves 2000, Zimmerman et al. 2009). Presumably the large size, rapid growth rates and correlated greater fecundity of these resident fish, combines with low survival rates of fish that out-migrate through valley rivers and the Delta to go to the ocean, strongly selects for the resident life history in these habitats. It should be noted that some large resident trout enter hatcheries and are inadvertently used as part of the steelhead breeding program (NMFS 2016).

Straying of hatchery fish. Hatchery fish of all salmonids are notorious for straying to non-natal rivers at fairly high rates, especially if smolts are released at a location other than the hatchery. Thus, in the Central Valley where hatchery fish have often been released at downstream locations straying should be expected. Accordingly, straying of hatchery fish is presumably one reason why CV steelhead show little genetic structure among populations (Pearce and Garza 2015). In an earlier report to CDFW, Garza and Pearse (n.d.) found that genes from Eel River fish were found in both resident and anadromous fish throughout the Sacramento Basin, indicating widespread straying of fish raised at Nimbus Hatchery on the American River from Eel River stock.

Rainbow trout above dams. A number of large reservoirs that block former steelhead streams (e.g., Berryessa Reservoir) have steelhead-like fish that migrate from the reservoir into tributaries to spawn. This expression of the anadromy gene complex increases in frequency in direct proportion to reservoir size (Leitwein et al. 2016), hence the propensity of rainbow trout in the Great Lakes to take on an adfluvial life history. These adfluvial populations are now isolated from below-dam forms and presumably are on their own evolutionary pathways, although individuals may be washed downstream of dams and interbreed with fish in the CV rivers below. While genetic work on rainbow trout above reservoirs has not focused on migratory forms, wild rainbow trout of unspecified origin are genetically distinct from lineages below the dams (Pearse and Garza 2015), providing a reason not to include them in the DPS. These fish would have been part of the DPS before dams were constructed and today may better represent the pre-dam steelhead genome than the lineages presently below the dams. No evidence was found that out-of-basin rainbow trout stocked to support fisheries have contributed to the genomes of the above-reservoir rainbow trout populations. We do not know if the migratory form in reservoirs are of hatchery origin or developed from natural populations present when the dams were built. Research is needed on the genomes and life histories of migratory fish in and above reservoirs, including their relationships with resident forms.

Genetic impacts of hatchery fish on wild populations. In general, wide-scale planting of catchable-size hatchery rainbow trout above dams has not resulted in long-term introgression of hatchery genes into wild resident rainbow trout populations in California (Pearse and Garza 2015). Presumably hatchery populations become adapted to hatchery conditions and their domesticated genomes do not do well in the wild. Even where they survive long enough to breed with wild fish it is assumed that their off spring which inherit domesticated genes have limited reproductive success in the wild thus limiting the flow of hatchery genes into wild trout populations. Below dam CV populations, in contrast, appear to have heavily introgressed with hatchery stocks (Pearse and Garza 2015).

These diverse lines of evidence indicate that the Central Valley steelhead is not a well-defined taxon, or even a DPS. We see two basic ways the taxon can be treated for management: (a) continuing with the status quo or (b) recognizing that all rainbow trout in the Central Valley belong to one broad, genetically diverse taxon, perhaps an ESU.

Status quo. This means to continue to recognize virtually any sea-run rainbow trout in the Central Valley as CV steelhead, except those from the American River. Rainbow trout populations above dams (including adfluvial forms) would continue to be excluded from the DPS. This strategy keeps steelhead of hatchery origin as the mainstay of the run, with only occasional individuals originating from tail-water rainbow trout populations or above dam populations. If the status quo is to be pursued, improved monitoring techniques and genetic methods will need to be widely used to limit interbreeding with America River steelhead or large resident rainbows that sometimes enter hatcheries masquerading as steelhead.

One broad taxon. The reality of the Central Valley is that below dams resident rainbow trout and steelhead, regardless of origin, form one interbreeding population (Pearse and Garza 2015). Central Valley hatchery programs strongly select for the anadromous life history while habitat conditions in the highly altered contemporary rivers strongly select for the resident life history in the year-round, cool tail water habitats directly below dams. Until recently, two hatcheries (Nimbus, Mokelumne) also selected for larger fish, favoring the introduced north coast phenotype over the Central Valley phenotype. Thus the anadromous steelhead life history is maintained in the Central Valley largely through artificial selection. In addition, rainbow trout living in tailwaters below dams also experience a very different selection regime than their pre-dam predecessors once did, one that seems to favor a resident life history.

Careful investigation of adfluvial populations above major dams – which likely better represent ancestral CV steelhead lineages than populations presently below dams – could result in finding individuals of native ancestry that could contribute migratory life history diversity to valley floor populations. This overall approach of acknowledging the biological interdependence of different life history traits could better support steelhead fisheries without concerns about 'contamination' of CV steelhead genomes with genetic material from other populations.

Life History: CV steelhead, like all steelhead populations, exhibit flexible and diverse life history strategies, which allow for persistence in spite of variable conditions in rivers and the ocean (McEwan 2001). General aspects of steelhead life history are portrayed in Moyle (2002) and the North Coast winter steelhead account, while interactions of steelhead with resident trout are reviewed in Kendall et al. (2015). While we accept recognition of all rainbow trout below dams as a unified population in the Central Valley, in this account we focus on fish with the anadromous steelhead life history and their interactions with resident trout living in tailwater habits below Central Valley dams. Broader accounts can be found in Moyle (2002) and in the coastal rainbow trout account. This account necessarily over-simplifies the adaptive life history variations in rainbow trout.

At present steelhead found in the Central Valley undertake spawning migrations during winter. There is indication that before the era of large rim dams that summer-run (late-maturing) steelhead, such as those that still exist in the Klamath River, once existed in the system (McEwan 2001). In the American River, summer steelhead apparently migrated upstream in May-July, and were fairly abundant (Gerstung 1971). Because summer steelhead over-summer in deep pools found in mid- to high elevation streams, they were extirpated by the large dams that blocked migration into upstream areas, despite an effort to propagate them (Gerstung 1971). For winter steelhead, peak immigration seems to have occurred historically from late September to late October, with some creeks such as Mill Creek showing a small run in mid-February (Hallock 1989). Juvenile CV steelhead generally migrate out of the system from late December through the beginning of May, with a peak in mid-March. There is a much smaller peak in the fall (Hallock 1961).

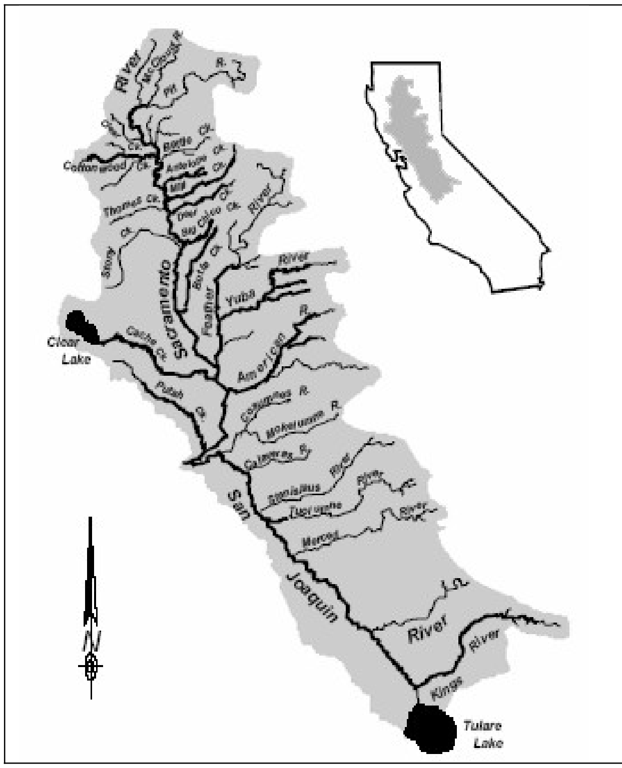
Juvenile CV steelhead are opportunistic, voracious predators on anything available in their rearing streams, from aquatic and terrestrial insects, to small fish, frogs and mice (Merz and Vanicek 1996, Merz 2002). However, benthic aquatic insect larvae are the mainstay of their diet, especially those of caddisflies (*Trichoptera*), midges (*Chironomidae*), and mayflies (*Ephemeroptera*). Below reservoirs, zooplankton may be important as well. Diets shift with season and size of juveniles. At times, salmon eggs, juvenile salmon, sculpins, and suckers may be important prey for yearling steelhead (Merz 2002) and these high calorie, seasonal forage items are especially important for growth. Curiously, Merz (2002) did not observe a change in average prey size with fish size, and even adult steelhead were observed feeding on small insects. In the Mokelumne River, Merz (2002) found that most juveniles tended to have relatively limited movement within their rearing area.

CV steelhead historically spent 1 (29%), or 2 (70%) years within their natal streams, with a small percentage (1%) spending three years before becoming smolts and migrating out of the Sacramento-San Joaquin system (Hallock 1961). It is not known to what extent, if at all, this anadromous life history diversity is still present today. As discussed in other accounts, the relationship between anadromous and resident forms of the same species is complex, but populations that have both basic life history strategies, as is true in the CV, are likely to have an evolutionary advantage. Resident fish persist when ocean conditions cause poor survival of anadromous forms, while anadromous forms can recolonize streams in which resident populations have been wiped out by drought, fire or other natural disasters. Anadromous steelhead produce many more than do resident fish and thus improve gene flow among rivers, increasing genetic diversity of the meta-population. Eggs are large which confers an advantage on sea-run females, which attain greater size and therefore produce more and larger eggs than their resident counterparts. Small resident males, on the other hand can still produce sufficient sperm to fertilize many eggs from large females. For this reason anadromy seems to confer a greater benefit on females (Ohms et al. 2014).

Habitat Requirements: The habitat requirements of CV steelhead *sensu lato* are similar to those of Central Coast steelhead and coastal rainbow trout. Water quality is a critical factor during freshwater residence, with cool, clear, well-oxygenated water needed for maximum survival (Moyle 2002). Optimal spawning temperatures are 4˚-11˚C, with embryos starting to die at 13˚C (McEwan and Jackson 1986). Fry, after emerging from gravel, usually migrate into shallow (< 36 cm) areas such as stream edges or low gradient riffles, often in open areas with large substrates (Everest and Chapman 1972, Everest et al. 1986, Fontaine 1988). With increasing size, fry move into higher-velocity, deeper, mid-channel areas in the late summer and fall (Fontaine 1988). Fry prefer water depths of 25-50 cm, and optimal growth occurs at temperatures of 15-19°C (Richter and Kolmes 2005). Juvenile steelhead (ages 1+ and 2+) prefer deeper water in summer than fry, and show a preference for pool habitat, especially deep pools near the thalweg with ample cover, as well as higher-velocity rapid and cascade habitats (Bisson et al. 1982, 1988; Dambacher 1991). In general, juveniles and resident adults prefer complex habitat with large physical structures such as boulders, submerged clay and undercut banks, and large woody debris that provide feeding opportunities, segregation of territories, refuge from high velocities, and cover from fish and bird predators. These features are characteristic today of small tributaries and they are uncommon in rivers below major dams. However, much of the complex cover in the Sacramento and San Joaquin rivers and their tributaries was removed in the 19^{th} century as part of 'desnagging' efforts to improve channels for navigation. CV steelhead now spawn in mainstem rivers, as do resident fish. Merz (2002) observed good growth and feeding of presumed steelhead in the Mokelumne River below Camanche Dam. The Sacramento River above Red Bluff supports resident rainbow trout, with some becoming steelhead.

Distribution: CV steelhead were historically part of a single meta-population with resident rainbows. The meta-population consisted of distinct populations in the Sacramento and San Joaquin Rivers and in most of their tributaries (Figure 1). Lindley et al. (2006) modeled the likely historical distribution of steelhead in the Central Valley based on habitat characteristics, and concluded there were possibly 81 discrete populations from the San Joaquin Valley north to the Pit River drainage, although a number of the 'populations' they identified are in areas not accessible to anadromous fish, indicating the close link between anadromous and resident fish.

The distribution of steelhead life history in the Central Valley today is greatly reduced from the historical distribution. This is the result of impassable dams and water diversions that block access to historical spawning and rearing areas (Figure 2). Estimates on the loss of habitat for Central Valley salmonids ranges from 80-95% (Clark 1929, CACSST 1988, Yoshiyama et al. 2001, Lindley et al. 2006). Populations of resident rainbow trout that have anadromous components within them are found in the Upper Sacramento River and tributaries, Mill, Deer, and Butte Creeks, and the Feather, Yuba, American, Mokelumne, Tuolumne, and Calaveras Rivers (McEwan 2001). A wider implementation of monitoring programs would probably turn up other populations, as has happened on Dry Creek, Auburn Ravine and the Stanislaus River (McEwan 2001). The Cosumnes River, which historically had steelhead, provides rearing habitat to non-natal steelhead from adjacent basins in wet years (NMFS 2014). However, most fish living below dams in tailwaters are resident rainbows; where cold water exists above dams, resident and adfluvial rainbows occupy such habitat.



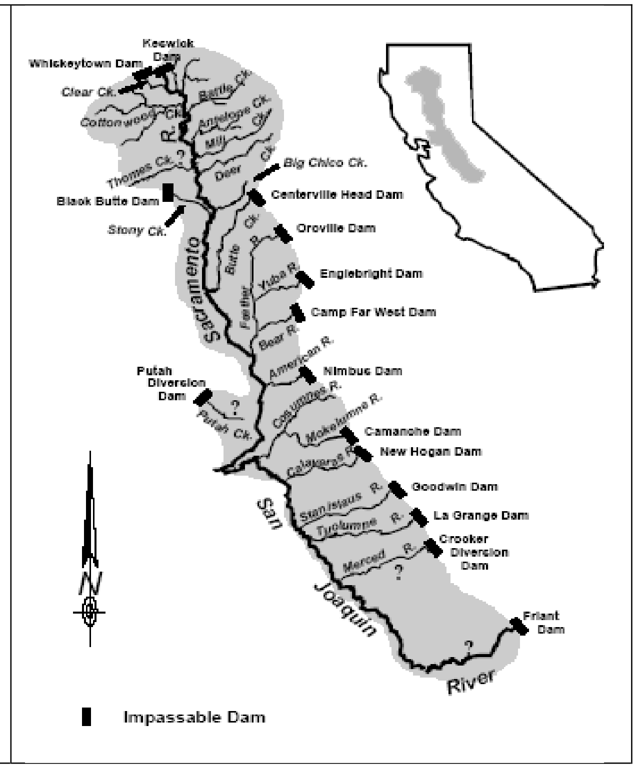


Figure 1. Historical distribution of steelhead in Central Valley drainages. Thick lines represent stream reaches that have documented historical evidence of steelhead. Thin lines represent likely distribution of steelhead based on documented occurrence of Chinook salmon or lack of natural barriers above documented steelhead occurrences. Shading represents an estimation of historical range within which steelhead likely occurred in numerous small tributaries not shown on map. From McEwan 2001.

Figure 2. Present distribution of steelhead in Central Valley drainages. Shading represents an estimate of present range within which steelhead likely occur including tributaries not shown on map. Question marks denote streams and stream reaches where steelhead currently may have access but their presence is unknown. From McEwan 2001.

Trends in Abundance: Data on the historical abundance of CV steelhead is lacking because of a combination of factors, including: (a) adult migration takes place during winter high flows, when streams were turbid, (b) steelhead do not aggregate in large numbers like the more conspicuous Chinook salmon, (c) early observers did not discriminate between steelhead, resident trout, and small salmon – they tended to be lumped as "salmon-trout" - and (d) much of 'steelhead' spawning apparently took place in smaller streams, after which many adults headed back to the ocean. Given the sheer size of available habitat and abundance of food resources for juvenile steelhead (*e.g.* small salmon, larval suckers and minnows, abundant insects generated by dead salmon nutrients), an estimate of annual adult numbers in 50,000-100,000 range would be reasonable.

The apparent precipitous decline of steelhead that lead to listing was obtained by looking at returns to the upper Sacramento River, which are based mainly on counts from fish ladders and hatchery returns, from an average of 6,574 fish in 1967-1991 to an average of 1,282 from

1992 to 2006 (Figure 3). Data from Coleman National Fish Hatchery on Battle Creek, (Sacramento River Rkm 534), shows a small but relatively stable return of 100-300 natural-origin adults per year (Figure 3). NMFS (2016) estimated total annual runs of steelhead to be around 4,600, including fish returning to hatcheries, with an additional 1,700 fish returning to the Nimbus Fish Hatchery and American River. However, all estimates are highly uncertain and should treated as best guesses.

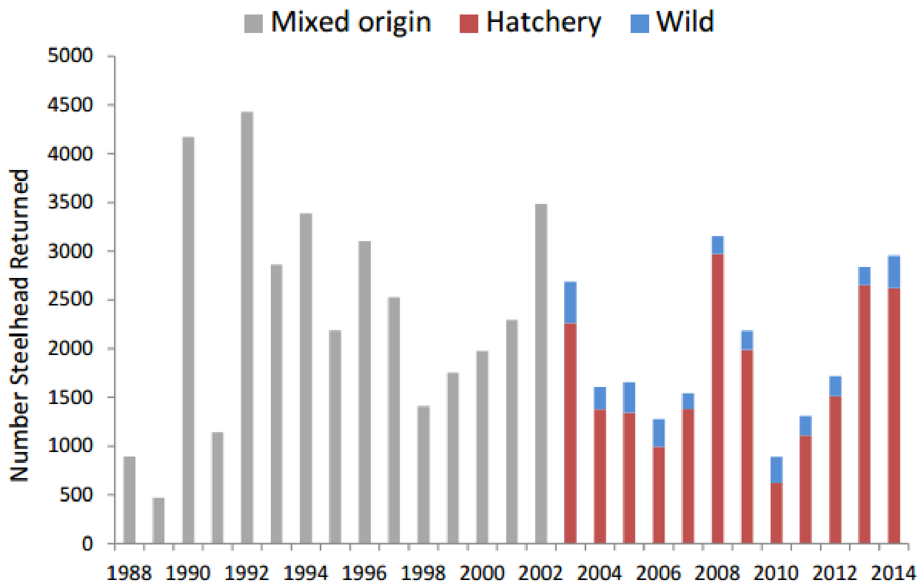


Figure 3. Adult steelhead returns to Coleman National Fish Hatchery, 1988-2014. From NMFS 2016, Fig. 1, pg. 12. *Prior to 2003, origins of returning fish were not classified.

Factors Affecting Status: Many stressors have contributed to the decline of steelhead life history in the Central Valley, including: major (rim) dams, diversions, barriers (small dams and other structures), levees and bank protection, dredging and sediment disposal, mining, contaminants, alien species, fisheries, and hatcheries (Upper Sacramento FRHAC 1989; Reynolds et al. 1989, 1993; CALFED 2000; CMARP Steering Committee 1999; McEwan 2001). Most of the factors affect steelhead in a manner similar to Chinook salmon, so are treated in the Central Valley fall Chinook salmon account.

Dams. Probably the single largest cause of loss of rainbow trout exhibiting steelhead life history has been the loss of historical holding, spawning and rearing habitat now above impassable dams. It is likely that somewhere between 80 and 95% of steelhead spawning and rearing habitat is no longer accessible to anadromous individuals. This habitat was mainly smaller tributary streams at higher elevations, but steelhead also likely ascended many mainstem rivers to higher elevations (McEwan 2001). Where cold releases from dams are present throughout the summer, resident populations of trout are present, which support tailwater fisheries. Migratory steelhead life histories have largely been replaced in the Central Valley by resident trout who spend their entire lifecycle in the tailwater releases from dams. Upstream reservoirs created by these same dams often support populations of migratory adfluvial trout.

Agriculture. In reality, agriculture is the single largest cause of steelhead decline because most dams in the CV were constructed in good part to provide water for irrigation. Agriculture also dried up some streams (e.g., San Joaquin River), created diversions that sucked up fish and water, dumped pollutants and warm-return water into streams, and generally degraded water quantity and quality in low elevation streams. On the other hand, summer flows below dams, either to satisfy downstream diverters or to protect fisheries, have created additional habitat for salmonids, especially rainbow trout. The Sacramento River, for example, has higher flows in summer than it did historically, and now maintains a wild trout fishery from Keswick Dam to Red Bluff. Although steelhead use these tailwaters, the resident life history is favored. In addition, concern for the affects of large diversions on salmon and steelhead has led to extensive screening of most large diversions to reduce mortality (Moyle and Israel 2005). NMFS (2016) emphasizes the many projects that are in place to mitigate for the impacts of agriculture.

Fire. Wildfires are having an increasing impact on streams supporting native rainbow trout populations above dams. NMFS (2016, p.35) states: "...Drier-than-normal conditions can increase the intensity and severity of wildfires. According to CalFire (www.calfire.ca.gov), in 2014, fire crews responded to 4,266 fires which burned over 191,000 acres (which was similar to the year-to-date average of 4,508 wildfires on 109,888 acres burned), and in 2015, there have been 6,284 fires and over 307,595 acres burned." How this landscape level change will affect rainbow trout and steelhead is not known, but it may contribute to increasing water temperatures and reduced cold water pools in reservoirs (see Climate Change).

Hatcheries. There are four hatcheries that raise steelhead in the Central Valley producing on average 1.5 million yearlings per year: Coleman National Fish Hatchery on the Sacramento River, Feather River Hatchery, Mokelumne River Hatchery, and Nimbus Hatchery on the American River (McEwan 2001). Steelhead reared in the first three hatcheries are considered to be part of the DPS (NMFS 2016). Williamson et al. (2011) regard hatcheries as a major threat to sustainable wild steelhead populations. The fish produced by hatcheries can negatively affect wild steelhead and rainbow trout in a number of ways including, displacing wild juveniles through competition and predation, competition between hatchery and wild adults for limited spawning habitat, hybridization with fish from outside the basin, and spread of disease. The first two effects are well documented for salmonids and may be responsible for the estimate that only 10-30% of returning steelhead in the upper Sacramento River are of natural origin (Reynolds et al. 1990). However, it is likely that, in the long run, hatcheries are causing a gradual decline in survival of both hatchery fish and naturally-spawned fish of hatchery origin. Reproductive fitness in steelhead can decrease rapidly when fish are raised in hatcheries. Araki et al. (2007) estimate that fitness of steelhead decreases almost 40% per generation of hatchery culture. When wild fish are brought into hatcheries there is a reproductive loss of 15% in the first generation and a further loss of 37% with each successive generation. This research indicates a major problem with using hatcheries to maintain or restore wild populations: steelhead of hatchery origin are quite different from steelhead of wild origin when it comes to long-term persistence in California streams and rivers. Currently, there seems to be strong natural selection favoring resident rainbow trout life history in tailwaters, presumably in part due to the poor ability of steelhead of either hatchery or wild origin to survive and contribute to populations.

The use of steelhead from outside the Central Valley as hatchery broodstock is well documented, although the effects of outside stocks on wild fish are not known. Outside stocks have been used in all four hatcheries, but Busby et al. (1996) and Pearse and Garza (2015) found that Coleman Hatchery and Feather River Hatchery fish are genetically most similar to naturally spawning Central Valley steelhead while Nimbus hatchery fish are most similar to Eel River steelhead. The Mokelumne River Hatchery fish at that time was rearing fish from the Nimbus Hatchery but has subsequently switched to rearing fish derived from returnees to the hatchery.

Whether or not these fish have actually harmed CV steelhead is not known; it can be argued that the increase in genetic diversity that results from hybridization, which is widespread in the CV steelhead/rainbow trout population, is potentially a positive attribute, allowing more flexibility for adapting to climate and other change. However, outbreeding depression between natural- and hatchery-origin fish could lead to an overall loss of genetic diversity in the population.

Estuarine alteration. Predation on steelhead smolts migrating through the highly altered lower Sacramento River and Delta may be heavy because of lack of cover, low outflows, and other factors. Confusing hydrology during periods of low-outflow in winter/spring may also direct smolts to areas with unfavorable conditions, such as agricultural ditches and fields. High rates of mortality of out-migrating smolts may select against the steelhead life history strategy.

Fisheries. Harvest of naturally spawned steelhead is prohibited within the Central Valley. Take is limited to one hatchery fish per day and every hatchery fish is marked. Because hatchery fish are raised for harvest and are not particularly suitable to augment wild stocks, their catch is not detrimental to the steelhead population as a whole. It is not clear what affect the incidental catch and release of wild steelhead has on the CV steelhead population as a whole, but some mortality is most likely occurring. However, steelhead are buffered somewhat by the presence of large populations of resident rainbows, which can reach 60+ cm FL, which make up most of the rainbow trout caught. See NMFS (2016) for discussion of angling effects on CV steelhead.

Alien species. Predation on steelhead and other salmonids by striped bass (*Morone saxatilis*), largemouth (*Micropterus salmoides*) and smallmouth bass (*Micropterus dolomieu*), and other alien predators is just one cause of mortality of juvenile steelhead, as is predation by native predators from fish to birds to sea lions. Such predation may be a major proximate cause of death, but the ultimate causes are severe habitat alteration (Grossman 2016).

Table 1. Major anthropogenic factors limiting, or potentially limiting, (a) viability of the steelhead (SH) life history in the Central Valley and (b) viability of the entire rainbow trout meta-populations, including both steelhead and resident (RRT) populations. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating a	Rating b	Explanation
Major dams	High	Low	Much former SH habitat is above dams; but RRT use much of former SH habitat; RRT in tailwaters.
Agriculture	Medium	Low	Habitats have been degraded through diversions, warm return water, and associated pollutant inputs.
Grazing	Low	Low	Livestock are pervasive on public and private lands bordering trout streams.
Rural/ residential development	Low	Low	Cumulative effects of numerous roads and rural development can negatively affect habitats.
Urbanization	Medium	Low	SH must pass through river sections leveed to protect cities, water diversions reduce flows overall but increase flows in summer. RBT fisheries are valued in tailwaters.
Instream mining	Low	Low	Gravel mining has impacts in limited areas.
Mining	Low	Low	Legacy effects of hard-rock mining are potentially severe in localized areas.
Transportation	Medium	Low	Roads present along many streams.
Logging	Low	Low	Both legacy effects and on-going impacts degrade aquatic habitats; most logging upstream of dams.
Fire	Low	Low	Fires have regional affects on water quality above dams.
Estuary alteration	High	N/A	The Delta /SF Estuary is a partial barrier to migration of SH, with poor water quality etc.
Recreation	Low	Low	Boating and other recreation may alter behavior.
Harvest	Low	Low	Recreational fisheries take mainly hatchery SH and RRT.
Hatcheries	Medium	Low	SH largely maintained by hatcheries; wild trout have legacy hatchery affects (genetics).
Alien species	Medium	Low	Hatchery SH juveniles eaten by striped bass and other predators (including natives).

Effects of Climate Change. Moyle et al. (2013) did not rate the effects of climate change on CV steelhead because of uncertainty of what fish should be included in the rating. Climate change models generally agree that the climate is/will be more variable than in the past 100 years, with long droughts more frequent, lower stream flows with higher temperatures in summer in most years, and larger floods in some years. This variability will increase the difficulty of managing large dams to maintain sufficiently large pools of cool water that can enable cold-water flows for salmonids through the hot, dry summers, including the cold water needed to maintain hatchery operations for steelhead. During droughts, habitat for rainbow trout in general will shrink although resident rainbow populations are capable of rapid (2-3 years) recovery if favorable conditions return. Presumably steelhead will have additional difficulties in maintaining their life history under extended periods of low flows, even in hatcheries. Our general prediction for climate change effects is that resident rainbow trout will persist in most places, but with more variable and often smaller population sizes. The persistence of CV steelhead life history, however, is problematic without major hatchery inputs.

Drought and climate change are related, and more frequent and intense droughts are likely to become more common in California. Drought has had a negative impact on Central Valley steelhead populations recently (NMFS 2016). From 2012-2016, historic drought likely reduced limited habitat quality and range for CV steelhead. In the lower American River, drought reduced coldwater pool upstream of Folsom Dam, impacting water releases and thus reduced survival of wild steelhead parr (NMFS 2016). While steelhead populations in the Central Valley have historically overcome periodic droughts, current low abundance and productivity may cause some populations to become extirpated during long dry spells (NMFS 2016). Prospects for re-establishing these populations in highly altered landscapes and habitats throughout the Central Valley are troubling.

Status Score = 3.0/4.6 out of 5.0. Moderate Concern/Low Concern. The first score only reflects the status of the steelhead life history, including both hatchery and wild fish. The second score reflects the potential for maintaining large, highly adaptable rainbow trout/steelhead populations below dams, assuming the steelhead life history would develop or disappear in response to changing environmental conditions. We view the second score as representing a Central Valley rainbow trout meta-population that includes steelhead life history as one option. Other options include adfluvial life history in large rivers, and various resident life history options. Using the NMFS definition of CV steelhead, Moyle et al. (2008) and Katz et al. (2012) rated the status of CV steelhead as 2.5 and 2.4, respectively, using similar rating systems.

The DPS was first listed as a threatened species under the ESA by NMFS in 1998 and was reevaluated and confirmed in 2005, 2010 and 2015. It is managed by CDFW as a sport fish with limited take of hatchery fish. NMFS (2016, p 3) affirmed the status of CV steelhead as a threatened species, based on the following summary:

> "Many watersheds in the Central Valley are experiencing decreased abundance of CV steelhead. Dam removal and habitat restoration efforts in Clear Creek appear to be benefiting CV steelhead as recent increases in non-clipped (wild) abundance have been observed. Despite the positive trend in Clear Creek, all other concerns raised in the previous status review remain, including low adult abundances, loss and degradation of a large percentage of the historical spawning and rearing habitat, and domination of smolt production by hatchery fish. Many other planned restoration and reintroduction efforts have yet to be implemented or completed, or are focused on Chinook salmon, and have yet to yield demonstrable improvements in habitat, let alone documented increases in naturally produced steelhead. There are indications that natural production of steelhead continues to decline and is now at very low levels. Their continued low numbers in most

> hatcheries, domination by hatchery fish, and relatively sparse monitoring makes the continued existence of naturally reproduced steelhead a concern. We therefore conclude that CV steelhead remain listed as threatened, as the DPS is likely to become endangered within the foreseeable future throughout all or a significant portion of its range."

The NMFS status of threatened is based on a rather narrow view of steelhead that depends on their numbers being independent of the rest of the rainbow trout complex of which they are part, a requirement for a DPS. This status is not based on biology but on a somewhat arbitrary distinction between anadromous and resident rainbow trout, reflecting only the populations of fish below the rim dams around the Central Valley. The decision notes that both natural and hatchery steelhead are in decline, suggesting that the life history strategy itself is not sustainable. It also implies that low numbers could be an effect of limited monitoring as well. In contrast, the tailwater 'resident' trout populations appear to be thriving, as are populations above dams. The status of adfluvial trout populations above dams is not known, but their life history is similar to that of steelhead except a reservoir with abundant forage fishes takes the place of the ocean. Pearse et al. (2014) indicate that resident status can evolve very rapidly from ancestral steelhead.

NMFS (2016) indicates that while hatchery steelhead are part of the DPS, their main concern is naturally produced steelhead, although continuing to raise hatchery fish appears to directly conflict with that goal. Below dams in the Central Valley, resident trout and steelhead, including hatchery steelhead, form one genetic population, without much structure and featuring significant genetic input from introduced North Coast hatchery steelhead. Clearly, the selection pressures in the current CV environment favor fish adapted for resident life history, which has a genetic basis and can evolve rapidly (Pearse et al. 2004). Fish with the low rated steelhead life history will be abundant only if conditions that favor steelhead improve, but the life history seems likely to continue to be expressed at low rates regardless.

Table 2. Metrics for determining the present status of steelhead/rainbow trout populations in the CV. Score a is for the steelhead life history whether of wild or hatchery origin; score b is for all naturally produced rainbow trout below dams, whether migratory or non-migratory. For individual metrics, 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is moderate to high. See methods for explanation.

Metric	Score a	Score b	Justification
Area occupied	4	5	Steelhead are present in small numbers in at least 5 rivers, plus 3 hatcheries.
Estimated adult abundance	3	5	Steelhead include hatchery fish; based on NMFS estimate.
Intervention dependence	2	4	Steelhead life history increasingly dependent on hatcheries; valley floor rainbow trout depend on tailwaters below dams.
Tolerance	4	5	Rainbow trout are one of most tolerant salmonids but conditions in lower rivers/Delta may exceed tolerances of migrating steelhead.
Genetic risk	4	4	Appears to be one genetically diverse population with little genetic structure.
Climate change	2	4	Populations below dams that maintain cool water pools most threatened, especially steelhead life history. Upstream populations less so.
Anthropogenic threats	2	5	Score of 2 if only steelhead considered, 5 if entire meta-population considered.
Average	3.0	4.6	a = 21/7; b = 32/7.
Certainty (1-4)	3	4	Well studied but steelhead/rainbow trout interactions need more study.

Management Recommendations:

Habitat. Current habitat conditions and dam operations in Central Valley rivers select against the anadromous steelhead life history, while hatchery operations select for anadromy. We must trust in the natural adaptability of rainbow trout and focus management on sustaining diverse, fishable, wild trout populations in tailwaters downstream of dams. When restoration actions improve conditions so that the migratory life history is once again favored, steelhead abundance will rebound. Habitat conservation efforts should be focused on the few streams that have track records of producing wild steelhead, such as Deer and Battle creeks. However**,** management of rivers to benefit naturally spawning Chinook salmon, especially late fall, winter, and spring runs should also benefit rainbow trout by providing both habitat protection and cold-water flows. For example, coldwater releases below Shasta Dam for winter-run Chinook provide habitat for a thriving, highly prized resident trout population.

NMFS (2016) provides a detailed, but not exhaustive, list of the many large-scale projects underway or proposed that should benefit steelhead, even if aimed mainly at Chinook salmon. These included restoring Clear Creek and Battle Creek, tributaries to the Sacramento River. In Clear Creek, barriers have been removed and passage to upstream areas improved, increasing steelhead numbers (NMFS 2016). For Battle Creek, dams have been identified for removal, with those remaining having passage improved. The Clear Creek and Battle Creek projects should be regarded as first steps towards a broader program of stream restoration, with more actions focusing on steelhead. In addition, significant actions associated with the operation of the State Water Project in the Central Valley under a 2009 NMFS Biological Opinion are likely to positively benefit CV steelhead survival.

Hatcheries. Interbreeding between hatchery and natural-origin steelhead likely reduces the reproductive capacity and inhibits the recovery of natural steelhead populations in the Central Valley. Hatchery steelhead should be excluded from the DPS and hatchery management should focus on reducing interactions between hatchery fish and naturally spawning steelhead while producing fish to support the fishery. Studies are also needed on why so few juvenile hatchery steelhead survive after being released. Delisting hatchery stocks would also allow loosening permitting restrictions on sampling of steelhead and rainbow trout for scientific research to better support recovery of naturally spawning populations.

A major effort should be made to understand the genetics and life histories of rainbow trout populations in and above reservoirs to determine the value of resident and adfluvial fish for future restoration efforts and well as simply to document their presence. We recommend against using "CV steelhead," however defined, in attempts to restore anadromous fish above dams (such as trap and haul programs), as proposed by NMFS (2016). Such programs do not work well where they have been tried (Lusardi and Moyle 2017) and steelhead face the particular problem of already-established rainbow trout populations above dams that include large adfluvial fish. In addition, genetic evidence suggests that current above-dam populations may better represent historical CV steelhead lineages than do anadromous individuals returning to the Central Valley today.

KLAMATH MOUNTAINS PROVINCE SUMMER STEELHEAD

Oncorhynchus mykiss irideus

Critical Concern. Status Score = 1.9 out of 5.0. Klamath Mountain Province (KMP) summer steelhead are in a state of long-term decline in the basin. These stream-maturing fish face a high likelihood of extinction in California in the next fifty years.

Description: Klamath Mountains Province (KMP) summer steelhead are anadromous rainbow trout that return to select freshwater streams in the Klamath Mountains Province beginning in April through June. Summer steelhead are distinguishable from winter steelhead by (1) time of migration (Roelofs 1983), (2) the immature state of gonads at migration (Shapovalov and Taft 1954), (3) location of spawning in higher-gradient habitats and smaller tributaries than other steelhead (Everest 1973, Roelofs 1983), and more recently, genetic variation in the Omy5 gene locus (Pearse et al. 2014). Summer steelhead are nearly identical in appearance to the more common winter steelhead (see Northern California coastal winter steelhead).

Taxonomic Relationships: For general relationships of steelhead, see Northern California coastal winter steelhead account. In the Klamath River Basin, salmonids are generally separated primarily by run timing, which has been shown recently to have a genetic basis (Kendall et al. 2015, Arciniega et al. 2016, Williams et al. 2016, Pearse et al. *In review*). The National Marine Fisheries Service (NMFS) does not classify Klamath River basin steelhead "races" based on run-timing of adults, but instead recognizes two distinct reproductive "ecotypes." Steelhead ecotypes are populations adapted to specific sets of environmental conditions in the Klamath Basin based upon their reproductive biology and timing of spawning (Busby et al. 1996). However, differences in run-timing of steelhead are not accounted for in current NMFS Distinct Population Segment (DPS) criteria, which lumps all Klamath Mountains Province steelhead together for management purposes.

Genetic and non-genetic factors (physical, physiological, ecological, and behavioral) indicate that the ocean-maturing ecotype (winter-run) steelhead are distinct from the stream-maturing ecotype (summer-run) fish. Genetic samples from fish collected between the Klamath River estuary and the river's confluence with the Trinity River support discrete migrating populations based primarily on timing of freshwater entry and state of maturity (Papa et al. 2007). Studies of the KMP steelhead Distinct Population Segment (DPS) indicate that KMP summer steelhead are more closely related to winter steelhead in the KMP than to summer steelhead from other basins (Reisenbichler et al. 1992, Pearse et al. 2014). Studies of summer and winter steelhead demonstrate greater levels of differentiation between spatially isolated reproductive populations (Papa et al. 2007, Pearse et al. 2007, Prince et al. 2015). Recent evidence suggests that early run timing has evolved, presumably independently, in several basins throughout California, including in the Klamath Mountains Province, and further indicates that summer-run life history evolved from winter-run fish in the same basin (Arciniega et al. 2016).

Pearse et al. (2007) analyzed genetic samples from 30 sites throughout the Klamath River watershed and three Trinity River sites. Results indicated that geographically proximate populations were most similar genetically, even when taking the influence of hatcheries into account from Iron Gate Hatchery on the Klamath and the Trinity River Hatchery. Steelhead sampled from the mouth of the Klamath were most similar to those in other coastal streams (Smith River, Wilson Creek), and samples from downstream of the Trinity River confluence (Turwar, Blue, Pecwan, Cappell, and Tully creeks) expressed limited gene flow with steelhead sampled upstream. Further, populations sampled in the middle regions of the Klamath River basin had genetics that clustered closely together with fish from the Trinity River and the Trinity River Hatchery, perhaps due to a history of egg transfers to the hatchery (Busby et al. 1994). At the upper portion of the watershed in California, fish from the Shasta and Scott rivers were genetically distinct from steelhead sampled in other mid-Klamath basins, and clustered closely to steelhead from Iron Gate Hatchery, suggesting that influence of hatchery gene flow (possibly from straying) to these nearby tributaries with summer-run fish. These fish have a section of the genome that has evolved separately in both steelhead and spring-run Chinook salmon (Pearse et al. 2016). Over time, positive selection and straying likely have caused this favorable mutation to radiate outward and give rise to steelhead and salmon along the West Coast of the United States that express a continuum of run timings to exploit habitats during different times of the year. In the future, KMP stream-maturing steelhead, labeled summer steelhead here, should be recognized as a distinct DPS and managed separately from winter steelhead based on differing life histories, morphologies, genetics, susceptibilities to environmental changes, and conservation needs (Hodge et al. 2013, Prince et al. 2015). A question remains, however, as to the assignment of the distinct fall run steelhead in the Klamath and Trinity rivers (Box 1).

The decision not to include fall-run fish as a discrete run with their own account, but in addition to the discussion of summer-run fish, was made based on a review of historical timing of adult steelhead returns in the Klamath/Trinity Basin, reports from NMFS and CDFW, literature review, and discussion with biologists. Busby, Wainwright, and Waples (1994) summarized the complex task of discerning discrete steelhead runs in the KMP:

> "Run-type designation for steelhead in the Klamath and Trinity Rivers continues to be perplexing, particularly with respect to what is historically called fall-run steelhead. Everest (1973) and Roelofs (1983) contend that spring and fall steelhead of the Rogue, Klamath, Mad, and Eel Rivers are in fact summer steelhead based on lack of segregation at spawning, and the observation that sport fisheries for fall steelhead are limited to rivers with summer steelhead. However, other biologists classify fall steelhead separately (e.g., Heubach 1992) or as winter steelhead." pg. 18.

The authors also note that, "In the Klamath River Basin, some biologists refer to fall-run steelhead; disagreement exists as to whether fall-run steelhead should be considered as summer-run, winter-run, or as a separate entity. In this status review, we consider fall-run steelhead from the Klamath River Basin to be part of the summer run" (pg. 51), citing the shared stream-maturing ecotype of summer- and fall-run fish. The authors also agreed at the time that the winter-run (ocean-maturing) steelhead encompassed the most abundant runs in the basin.

However, Hopelain (1998) used scale analysis to determine run timing and life history strategies in KMP steelhead, albeit with small sample sizes. However, he assumed that the race of each steelhead examined could be assigned based on river location and time of capture. He also noted that the fall steelhead run is by far the largest of the three runs (summer, fall, winter) of steelhead in the basin, which contradicts the consensus finding just 4 years prior.

More recently, NMFS (2009) noted that "populations in the Basin are… comprised of three distinct runs; summer, fall and winter" (pg. 7) in their annual report to Congress. There is still confusion over how names are assigned to steelhead with a diversity of life histories and run timings. Without updated scale analyses or genetics studies, fall-run steelhead in the KMP lack sufficient information to be scored and discussed in a separate account.

It is likely that the steelhead that made it into the upper Klamath Basin before construction of Copco Dam were summer steelhead. The other alternative is that the upper basin steelhead were anadromous or fluvial redband trout (*O. mykiss newberrii*). The genetic relationship of KMP steelhead to these redband trout, which show migratory and resident life history variations, has also not been determined. With the four lower Klamath dams (Copco 1, Copco 2, J.C. Boyle, and Iron Gate) scheduled for removal beginning in 2020, this relationship bears further study because redbands will again have unimpeded access to the Pacific Ocean and summer steelhead will have access to some of their presumed historical spawning grounds.

Box 1. Fall-Run Steelhead: The Klamath-Trinity's Unrecognized Run?

Steelhead in the Klamath Mountains Province return to fresh water in every month of the year (W. Sinnen, CDFW, pers. comm. 2016) and represent a continuum of life history strategies and run timings (Hodge et al. 2016). The presence of an apparently distinct fall run of steelhead in the Klamath and Trinity rivers represents an interesting problem in evolution and management. It is unclear if a discrete run of fall-run steelhead historically existed in the basin. In the 1990s, steelhead that entered freshwater between August and November were referred to only occasionally as "fall steelhead" (Burgner et al. 1992). This peak occurs considerably later than summer steelhead, as revealed by snorkel surveys in tributaries (which occur in August), and earlier than the bulk of winter-run, ocean-maturing fish that peak in January. While summer-run adults typically migrate far upstream into headwater reaches of tributaries, fish returning in the fall cannot access these waters due to low flows, and thus hold mostly in mainstem river habitats until stream flow increases (W. Sinnen, CDFW, pers. comm. 2016). These stream-maturing fish have undeveloped gonads, and have thus been classified as summer steelhead by NMFS (Busby et al. 1996).

It is not clear whether hatchery-origin fish support this run, although thousands of unmarked fish of presumed wild origin are counted each year at Willow Creek Weir on the mainstem Trinity River during fall (W. Sinnen, CDFW, pers. comm. 2016). These fish are considered fall-run steelhead by CDFW because the weir cannot be left in place during the highest flows of the year when winter-run fish return (TRP 2014). Very small populations of steelhead with fall-run timing also exist in upper Klamath tributaries, though these fish may be offspring of hatchery strays (W. Sinnen, CDFW, pers. comm. 2016). Selection pressures from dam and hatchery operations (Abadia-Cardoso et al. 2013), intra- and inter-basin transfers of fish over decades, and associated straying of returning adults have likely played a role in the run-timing of adult steelhead in the KMP.

It is unknown if fall-run fish exemplify a shift in early maturation and run-timing of winter steelhead, or a protracted migration window of stream-maturing ecotype summer steelhead in response to alterations to streamflow over the last several decades of dam operations. The latter is most likely given that the early maturation phenotype is the result of a rare set of genes that are unlikely to have been 'invented' in steelhead more than one or two times in the basin. Spatial and temporal overlap of this intermediate life history strategy with the more discrete summer- and winter-run fish has likely muddled previous analyses and created more uncertainty. Regardless of the origins, conservation and management implications of fall-run life history displayed in KMP steelhead require further study; additional information, especially regarding genetics and scale analyses, is needed to warrant a separate account and scoring of these fish.

Life History: Steelhead are coastal rainbow trout that undergo physiological changes as juveniles, become anadromous, and migrate from the ocean to return to spawn in freshwater. Steelhead are iteroparous and can spawn several times throughout their lives, though after each successive spawning run decreases (W. Duffy, HSU, pers. comm. 2017). The Klamath and Trinity rivers are unusual for California in that they allow juvenile and adult steelhead to enter and exit the estuary in every month of the year (W. Sinnen, CDFW, pers. comm. 2016). The diversity of habitats, from inland spring-fed systems such as the Shasta River to snowmelt-driven tributaries of the Trinity River, underpin the variability in life history strategies in KMP steelhead. The cues for early migration and smoltification in steelhead that determine when they arrive on spawning grounds have been linked to a specific portion of their genome known as Omy5 (Pearse et al. 2016). This finding indicates that the basic life history diversity expressed in a given run of steelhead has its basis in a common ancestor and can be passed on to offspring. Across the spectrum of run-timing strategies, two basic reproductive strategies exist in KMP steelhead: ocean-maturing and stream-maturing. Ocean-maturing winter steelhead are most common. They enter fresh water with well-developed gonads and spawn relatively soon thereafter, while stream-maturing fall or summer steelhead enter fresh water with immature gonads, requiring several months to mature and then spawn (Burgner et al. 1992, Busby et al. 1996). These two ecotypes often overlap in spawning timing, which can confound differentiating between summer, fall, and winter steelhead for management. For example, it is likely that summer, fall, and winter fish appear at the Trinity River fish weir simultaneously.

In the KMP, summer-run steelhead are uncommon compared to ocean-maturing winter-run fish, but continue to persist in sub-basins of the Klamath Mountains Province. These fish are distinguishable from winter steelhead on the basis of adult migration and their morphological and physiological differences. They typically enter rivers in spring (April-June) and migrate upstream through early summer. In the Trinity River, however, summer steelhead enter between May and October. Summer steelhead are found in the Trinity River tributaries by June and in the mainstem Trinity above Lewiston by August. In the Klamath River, summer steelhead ascend into summer holding areas during a similar period. These holding areas are typically deep bedrock pools in canyon reaches of streams with some overhead cover and subsurface flow to keep temperatures cool. Once in upper reaches of cool tributaries, summer steelhead mature over several months in deep pools (Busby et al. 1996, Shapovalov and Taft 1954). They spawn in upstream regions that are largely not used by winter steelhead (Roelofs 1983) including smaller tributary/headwater streams. Spawning in the Trinity River peaks in February, earlier than winter steelhead, which peak in March. Spawning begins in late December and peaks in January (Roelofs 1983) throughout the Klamath Mountains Province (Figure 1). Maximum-recorded age of steelhead in recent studies is seven years, and female fecundity has been estimated at 2,000 to 3,000 eggs per fish (Hodge et al. 2015). In the Klamath drainage, 40 to 64% of the total spawning population are repeat spawners (Hopelain 1998). In the Salmon River, fully one-third of returning adult steelhead are repeat spawners, compared to about 15% in other tributaries (Hodge et al. 2015).

Steelhead race	KRSIC (1993)	Hopelain (1998)	USFWS (1979)	Busby et al (1996)	Moyle (2002)
Spring/Summer	May- July	March-June	April-June		April- June
Fall	August- October	July-October	August-November		
Winter	November- February	November-March	November-February		November-April
Stream-maturing				April- October	
Ocean-maturing				September-March	

Figure 1. Classification of different run-timings and reproductive ecotypes of steelhead found in the Klamath River Basin. From: Moyle 2002, Figure 3, pg. 4.

Early life history of summer steelhead in the Klamath River basin is presumably similar to the better-understood summer steelhead in the Eel River (see Northern California coastal summer steelhead account). Based on their occupancy of headwater streams with relatively low (<50 CFS) winter flows (Roelofs 1983), the fry move out of these smaller natal streams into larger tributaries soon after emerging. Scale studies suggest the majority of juvenile fish from the Middle Fork Eel River become smolts at two years old and return at age 3 and 4 (Puckett 1975). Average lengths for steelhead smolts entering the ocean ranges from 200mm (Trinity River) to 270mm (Shasta River) (Hodge et al. 2015).

Half-pounders: Winter and summer steelhead are all known to give rise to offspring that may exhibit a "half-pounder" life history strategy. While the half-pounder life history seems to be most closely associated with summer steelhead (Lee, 2016), they do not mature or reproduce while in the river. The presence of half-pounders over-summering with adult summer steelhead is not typically considered in the literature (Kesner and Barnhardt 1972, Hopelain 1998). However, annual snorkel surveys of summer steelhead in late summer in the Salmon, New, and South Fork Trinity rivers and their tributaries regularly encounter apparent half-pounders (L. Cyr, USFS, pers. comm. 2016). Frequently, the half-pounders outnumber adult steelhead during these surveys, but it is also possible that they are mistaken for resident rainbow trout, which confounds abundance estimates of these fish (D. Lee 2016). A relatively low proportion of half-pounder-sized steelhead are resident fish in all but the Scott River (Hodge et al. 2014). In the Klamath River, half-pounders are considered stream-maturing fish like summer steelhead because they are not mature when they enter freshwater holding habitat (Lee 2016). The relative contribution of winter-run and summer-run steelhead to offspring that exhibit the half-pounder life history is unknown, but seems to be more common to summer-run fish (Lee, 2016). However, differences in run timing between half-pounder fish and summer steelhead require more study. The presence of half-pounder fish is uncommon above Seiad Valley on the Klamath River, (Hopelain 1998) and summer steelhead are also not found in tributaries above this area. Individuals expressing these two life histories are often counted together because they both oversummer in pool habitat censused during snorkel surveys in the Salmon, South Fork Trinity, and New rivers and Wooley Creek in July-August (L. Cyr, USFS, pers. comm. 2016).

Habitat Requirements: Juvenile habitat requirements of summer-run steelhead seem to be similar to the more common winter steelhead (see Northern California coastal winter steelhead account). However, over-summering habitat for adult summer steelhead is critical for survival of these fish during periods of climatically and hydrologically unfavorable conditions. They are often found in the same cool tributaries as half-pounder fish and spring-run Chinook salmon, where their habitats overlap (L. Cyr, USFS, pers. comm. 2016). For example, adult summer steelhead in the New River occupy confluence and other pools of moderate size (200-1,000 m^2)

with depths of 1.0 to 1.4m. Although localized areas of cool water (i.e., 0.2 to 3.8°C lower than the mean hourly pool temperature of 18.0°C) are observed in some pools, Nakamoto (1994) found that more important factors influencing summer steelhead habitat use are pool size, low substrate embeddedness (<35%), presence of riparian habitat shading, and instream cover associated with increased velocity through the occupied pools (Baigun 2003). Cover was used by 99% of the summer steelhead observed during the day on the New River, with bedrock ledges and boulders the most frequently used habitat types (Nakamoto 1994).

Spawning habitat for summer steelhead is variable and their consequent temporal and spatial isolation from other steelhead runs maintain some level of genetic differentiation from winter steelhead in the same watershed (Barnhart 1986, Papa 2007, Prince et al. 2015). Summer steelhead often spawn in intermittent headwater streams when sufficient flows are available, from which the juveniles emigrate into perennial streams soon after hatching (Everest 1973). Roelofs (1983) suggested that use of small streams for spawning may reduce egg and juvenile mortality because the embryos are less susceptible to scouring by high flows and predation on juveniles by adults is decreased due to lower densities of predators in smaller streams. Water velocity and depth measured at redds are 23-155 cm sec-1 and 10-150 cm, respectively, and diameters of the gravels are typically 0.64-13 cm. The concept of spawning spatial segregation is based largely on summer steelhead distribution and habitat utilization and inferred from genetic variation, since little is known about the spawning distribution of winter-run steelhead throughout the KMP due to high, turbid flows during spawning.

After feeding for several years in a narrow range of sea surface temperatures in the ocean (typically 8-14°C), a steelhead's spawning migration is triggered and they return to their natal rivers (Harding 2015, Hayes et al. 2016).

Distribution: The KMP summer steelhead range in California encompasses a variety of different habitats from cool fog-belt redwood forests on the coast near the mouth of the Klamath, to hot and arid inland valleys at the headwaters of the snowmelt-fed Scott and spring-fed Shasta rivers, allowing steelhead to adapt various life history strategies to make use of them. For example, the Trinity River is largely fed by snowmelt and runoff, while the Klamath and Shasta rivers are spring-fed at their sources, causing distinct differences in hydrographs and thus summer steelhead distribution and abundance (CDFW 2015). The Smith River is undammed and relies on runoff to fill its banks. The KMP summer steelhead range includes 23 streams including the Klamath River and its main tributaries: the Trinity, Salmon, Scott, and Shasta rivers and other streams north to the Elk River near Port Orford, Oregon (Nelson 2016). In the Klamath River, the upstream limit of steelhead migration is Iron Gate Dam, near the Oregon border. Their historical range likely included tributaries to Upper Klamath Lake, prior to dam construction (Hamilton et al. 2005). In the Trinity River, upstream migration is blocked by Lewiston Dam (Moffett and Smith 1950). Their range also encompasses the Smith River in California and the Rogue River in Oregon. In California, KMP summer steelhead currently inhabit the larger tributaries of the mid-Klamath subbasin (Bluff, Red Cap, Camp, Dillon, Clear, Elk, Indian, and Thompson creeks), the Salmon River, and the Trinity River. In the Salmon River they are found in the North Fork, South Fork, and Wooley Creek. In the Trinity River drainage, populations of summer steelhead are present in Canyon Creek, Hayfork Creek, North Fork Trinity, East Fork Trinity, South Fork Trinity, and New rivers. In addition, the Smith River also supports summer steelhead.

While the majority of a steelhead's life is spent at sea, relatively little is known about their oceanic distribution. Many age-0 California steelhead juveniles apparently spend a year feeding in the California Current off the Klamath-Trinidad region, then move northwest to cooler waters offshore in the North Pacific. NOAA Fisheries' Southwest Fisheries Science Center has been conducting salmon trawling surveys since 2010 along lateral transects from shore from the Gulf of the Farallones to Southern Oregon. In general, steelhead were most abundant in trawls off of the Klamath-Trinidad transect, and catch per unit effort for both juvenile and adult steelhead increased with distance offshore compared to both coho and Chinook salmon (Harding 2015). These surveys show that while at sea, steelhead feed on krill, fish, and amphipods in low densities and in surface waters further offshore than either coho or Chinook salmon prefer (Hayes et al. 2016). The fact that steelhead are rarely caught in commercial fisheries for both coho and Chinook salmon lend further evidence of their differentiated habitat use in the ocean (Hayes et al. 2016).

Trends in Abundance: Little is known about the historical abundance of summer steelhead in the KMP; quantitative records of summer steelhead numbers exist only for recent decades (Roelofs 1983). While the stock status of this stream-maturing run of fish is uncertain (Nelson 2016), given the limited amount of habitat now available since large portions of the upper Klamath and Trinity basins were blocked by dams, it is likely that summer steelhead in the Klamath Basin currently represent only a small fraction of their original numbers and are currently in decline. Some summer steelhead populations (e.g., Salmon River) have declined precipitously in the past 30-40 years (Quiñones et al. 2013), while others have shown increases in recent years (e.g., New and North Fork Trinity rivers; USFS and CDFW 2016). Snorkeling counts, which provide the only abundance estimates for summer steelhead, are prone to numerous problems such as counting half-pounders as adult steelhead, incomplete spatial surveys due to access problems associated with private landownership, illegal marijuana cultivation throughout the KMP, and observational bias by surveyors. Thus survey numbers likely represent the minimum fish present in reaches during specific times, and so are mainly useful for trend analysis. However, the majority of estimates for California populations have been less than 100 fish each for the past decade (new data here).

Despite the highest documented diversity of life history strategies expressed by steelhead in the KMP (Hodge et al. 2016), resiliency has proven elusive for the population as a whole. In 1989-1991, the three-year average exceeded 500 fish in the North Fork Trinity River and New River and Dillon Creek in the middle Klamath River, which also each had more than 500 fish in 1999-2001 and 2002-2004. These two tributaries averaged more than 800 fish in 2009-2012. Three-year averages also exceeded 500 fish for some years in Dillon Creek (2000-2004) and Clear Creek (2001-2003) (T. Jackson, CDFW, pers. comm. 2011) on the order of hundreds of fish and even over one or two thousand fish in some years. For example, the New River had an estimated 2,108 summer steelhead counted in snorkel surveys in 2003. The most recent data in 2006 suggests that no tributary contains more than 1,000 summer steelhead. Out of 1,820 summer steelhead populations surveyed in the Klamath-Trinity basins, eleven averaged <100 fish annually and nine averaged < 20 fish each for the years they were surveyed only Wooley, Dillon, and Clear creeks, the Salmon, North Fork Trinity, and New rivers have averaged more than 100 fish throughout the survey series (Table 1).

Table 1. "Summer Steelhead Totals." Data collected by the Klamath Basin Collaborative Partnership Orleans/Happy Camp Ranger Districts, USFS. *In 1988-89, Bluff and Red Cap counts combined half-pounders and adults. "Other" represents small tributaries between Aikens and Beaver Creek on the Klamath River. From: Cyr, L., USFS, pers. comm. 2016.

Year	Adults	Half pounders	Total Steelhead	Bluff	Red Cap	Camp	Wooley	Dillon	Clear	Elk	Indian	Thompson	Grider	Other
1985	457	-	**457**	5	-	-	290	-	162	-	-	-	-	-
1986	428	-	**428**	-	-	-	-	-	428	-	-	-	-	-
1987	900	17	**917**	-	-	-	285	77	524	31	-	-	-	-
1988	1433	36	**1585**	91	25	-	362	299	693	69	46	-	-	-
1989	1503	36	**1620**	58	23	18	245	38	934	150	154	-	-	-
1990	271	72	**343**	-	-	-	73	74	117	57	21	-	-	1
1991	199	220	**419**	212	2	1	25	88	39	44	8	-	-	-
1992	119	360	**480**	149	31	7	38	-	100	72	82	-	-	1
1993	242	337	**579**	-	-	-	112	161	178	61	67	-	-	-
1994	185	251	**436**	15	4	2	54	-	134	110	117	-	-	-
1995	209	259	**469**	20	3	2	42	122	175	61	39	4	-	1
1996	73	270	**343**	15	6	1	15	91	102	96	-	14	-	3
1997	123	287	**410**	2	1	0	54	180	85	33	42	13	-	0
1998	108	667	**775**	15	6	4	41	151	68	490	-	-	0	0
1999	116	219	**335**	5	3	0	30	209	65	23	-	-	-	-
2000	489	511	**1000**	9	0	0	49	679	186	77	-	-	-	0
2001	1153	753	**1906**	9	2	2	214	929	538	212	-	-	-	-
2002	1728	993	**2721**	35	9	4	288	1108	1034	200	-	-	29	14
2003	913	375	**1288**	31	23	5	288	576	238	55	4	46	0	22
2004	587	456	**1043**	20	20	3	110	437	268	112	-	17	44	12
2005	243	214	**457**	10	10	13	50	216	108	34	-	9	3	4
2006	384	330	**714**	7	6	0	-	448	158	37	30	13	8	7
2007	187	270	**457**	18	4	15	59	58	129	33	87	21	16	17
2008	200	184	**384**	11	0	0	-	-	222	68	71	9	2	1
2009	154	290	**444**	23	2	7	90	107	78	56	42	36	1	2
2010	170	256	**426**	10	2	2	64	119	97	38	51	27	7	9
2011	233	296	**529**	11	2	1	47	166	141	87	70	0	0	4
2012	115	306	**421**	3	4	1	80	119	142	37	29	3	0	3
2013	195	506	**701**	11	1	4	73	113	299	91	36	14	0	59
2014	572	654	**1226**	4	1	2	219	494	269	67	111	27	19	13
2015	236	502	**738**	4	0	1	60	324	99	85	117	23	0	25

The "effective" (breeding) population sizes are likely less than the actual counts, so many populations may be close to or below the minimum size needed for long-term persistence (Lindley et al. 2007). These estimates are of fish holding in pools in midsummer and the number surviving to spawn in winter probably is considerably less because of natural mortality and poaching, which is a major cause for concern due to the low, clear water in most holding habitat. Most of the populations were severely affected by the extraordinary floods of 1964, which filled in many deep pools with sediment and presumably scoured out redds. Although their habitat is gradually recovering from this disaster, the number of summer steelhead has fluctuated widely without any upward trends. The ongoing drought has likely negatively impacted summer steelhead survival to spawning due to lower flows and higher stream temperatures than average over the last several years. Summer steelhead population estimates from each stream in the DPS have likely been less than 1,000 individuals over the last several years (W. Sinnen, CDFW, pers. comm. 2016). The status of each major population is a follows:

Mainstem Trinity River. Moffett and Smith (1950) indicate that summer steelhead were common in the snowmelt-fed upper mainstem Trinity River in the 1940s. This population apparently persisted through the early 1960s but is probably now extirpated (B. Curtis, 1992, CDFG files) due to the effects of Trinity and Lewiston dams. Suitable water temperatures downstream of Lewiston Dam provides habitat for summer steelhead, although the abundance of these fish is not known. It is likely that a large proportion of fish observed in the upper mainstem Trinity River recently originate from the Trinity River Hatchery or their offspring.

North Fork Trinity River. There is little historical information on summer steelhead in this stream, but recent data indicate that the population fluctuates between 200 and 700 fish per year. Summer steelhead distribution has changed relatively little during the recent period of monitoring and the majority of holding habitats have remained in the middle reaches. Their distribution at the upper extent seems to be conditional based upon sufficient flows, while temperature may be limiting in the reaches closest to the mainstem Trinity River confluence (Everest 1997). This stream has been heavily altered by mining, and therefore runs were likely much higher in the past (Roelofs 1983). Canyon Creek, a tributary near the North Fork Trinity River, continues to see small numbers of summer steelhead return each year.

South Fork Trinity River. There is no historical information on summer steelhead in this stream. Recent counts were as low as 34 fish, although in 2006 and 2007 more than 100 fish were observed. It is the only one of 18 stream reaches surveyed in KMP that displays a general upward trend in the snorkel survey abundance index, suggesting it could be a valuable refuge habitat as conditions continue to degrade in surrounding watersheds. Recent surveys on the South Fork Trinity River show summer steelhead were less common than half-pounder steelhead, although similarly distributed (Garrison 2002). This trend is repeats itself throughout the compiled survey data from the entire KMP (Table 1).

New River. This tributary of the Trinity River is home to the largest summer steelhead population in California, although it is highly accessible to humans and was heavily dredged for gold in the past. The estimated average abundance for 1979-2006 was 647 summer steelhead, with an average of 2108 fish in 2003, and 977 in 2004-2006. Availability of cool canyon pools of various types and overhead cover allow summer steelhead adults to successfully oversummer in remote reaches of this stream.

Klamath River tributaries. Summer steelhead populations averaging less than 70 fish are found in six small tributaries: Bluff, Red Cap, Camp, Indian, Thompson, Grider creeks, most with populations of less than 100 fish. Summer steelhead populations in Elk Creek averaged about 110 fishes during the years they were surveyed. Dillon and Clear creeks retain the largest summer steelhead populations on the Klamath River, averaging more than 300 fish annually during the years they were surveyed (1978 and 1980, respectively, Figure 2). While there is no clear trend among the smaller populations, summer steelhead populations on Dillon and Clear creeks became more abundant through the 1990s and were estimated to be over 1000 fishes in 2003. The estimates have decreased significantly over the past few years, and the 2005-2009 counts were 207 and 139, respectively.

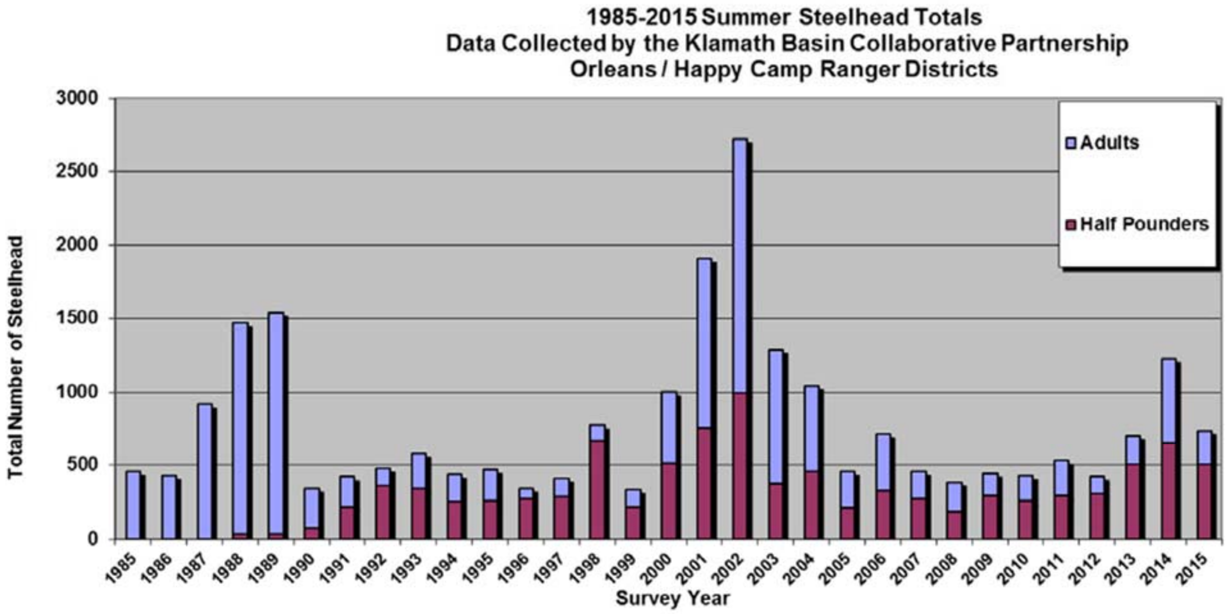


Figure 2. Compiled summer steelhead and half-pounders in the KMP from 1985-2015. The most important tributaries are Wooley Creek (Salmon River), Dillon Creek, and Clear Creek by contribution of returning adults. From: Cyr, L. USFS, pers. comm. 2017.

Salmon River. Despite the presence of suitable spawning and holding areas, the two forks of the Salmon River combined now only support less than 100 summer steelhead fish per year and have included half-pounder fish that could be mistaken as adults. These watersheds were heavily mined during the late 19th century and still impacts runoff and the prevalence of heavy metal contaminants (Klamath National Forest and Salmon River Restoration Council 2002). The 1990 complete census of the Salmon River showed 48 summer steelhead (DesLaurier and West 1990) and the number observed remained very low with a recent increase since 2000. Since 2001, between 100 and 350 summer steelhead and oversummering half pounders have returned to the Salmon River. Correlation trend data between Iron Gate Hatchery steelhead and Salmon River summer steelhead suggest that hatchery stocks are influencing adult escapement trends (Quiñones et al. 2013). Further investigation is needed to explore adult escapement and population trends between hatchery and wild steelhead in the basin.

Wooley Creek. Like the Salmon River, to which Wooley Creek is tributary, this rather inaccessible (to humans) stream has maintained a run of steelhead that is usually 100-400 fish per year. This population did not experience a gradual increase during the 1990s like larger KMP summer steelhead populations, but instead declined to average 50 individuals annually between 1990-2000. The estimated run size recently peaked at 288 fish in both 2003 and 2004, although more recent estimates have returned to approximate the 1990s average.

Smith River. Only 10-20 fish are estimated to occur in each of five tributaries in recent years (Reedy 2005), less than 100 fish total, but this river may never have supported summer steelhead in large numbers (Roelofs 1983, J. Garwood, CDFW, pers. comm. 2016), so these counts may be in line with historical averages. New information continues to become available through dedicated and comprehensive monitoring, which will be formalized in the draft Smith River Fishery Management Plan scheduled for release in 2017.

Overall, KMP summer steelhead numbers in recent decades appear to have ranged between 1,400 and 4,000 fish in the entire KMP system per year. They have since dwindled to likely less than 2,000 returning adults to the entire basin for the past decade. These estimates almost certainly represent only a small fraction of historical numbers, based on the fact that large areas of formerly accessible habitats are now blocked above dams, that summer steelhead generally utilize these same types of blocked habitats (e.g., smaller tributary headwater streams), and human land and water uses have altered many remaining accessible habitats. Increases in numbers have been documented in some tributaries, such as the South Fork Trinity River in recent years, presumably due to a combination of good ocean conditions, recovering stream habitats, and restrictive sport fishing regulations.

Factors Affecting Status: Summer-run steelhead are exceptionally vulnerable to human activities because adults are conspicuous in their summer holding pools and are present in rivers for extended periods of time. All salmonids are subject to the legacy effects of 19th century hydraulic mining and logging in the KMP, which devastated many watersheds. While steelhead populations may have recovered somewhat from these legacy effects, by the time there was much interest expressed in summer steelhead, their numbers were low again, presumably depressed by pervasive 20th century mining and logging. There is no hatchery production of summer steelhead, so their populations truly reflect local conditions. In contrast, fluctuations in the more recently documented fall-run fish are likely responses to a combination of recent stressors and selection pressures on hatchery-origin fish in the Trinity basin. Other more general factors are discussed under North Coast winter steelhead and Upper Klamath-Trinity River spring Chinook; the latter often share habitat with summer steelhead.

Dams. Three dams that directly affect KMP steelhead in the Klamath basin are Iron Gate (Klamath), Dwinnell (Shasta), and Lewiston (Trinity) dams. All are part of larger projects and these three dams have blocked access to large portions of formerly utilized KMP steelhead habitats, especially important spawning and rearing grounds in the middle and upper portions of both systems. These fish probably ascended higher in each watershed than any other salmonid based on their morphological adaptations to hold in lower, faster water and leap higher than other steelhead or Chinook salmon (Hodge et al. 2011). These dams negatively affect all salmonids by limiting access to critical spawning, rearing, and migration habitats, as well as altering flows and increasing water diversions (Lewis et al. 2004) and degrading water quality (Hamilton et al. 2011). Dam operations have decreased the variability, magnitude, duration, and altered timing of flows in the Klamath River. Peak flow timing has also shifted to at least a month earlier than prior to dam construction (Hamilton et al. 2011). Lower flows are of particular concern in the summer because daytime water temperatures can reach 24-26°C across large portions of the Klamath system, reducing available rearing habitat.

However, removal of Iron Gate and other upstream dams under the Klamath Basin Hydroelectric Agreement, and concordant Klamath Basin Restoration Agreement, will open up hundreds of kilometers of potential steelhead habitat in the future; the dam removal project is slated to begin by 2020.

Dwinnell Dam has blocked access to greater than 30km of habitat in the upper Shasta River since its construction in 1928. The dam, in combination with multiple diversions, has decreased the quality and quantity of habitat by reducing flows and disrupting the natural hydrograph, eliminating peak flows that could improve habitat conditions for steelhead and other salmonids (Lewis et al. 2004). Minimum daytime water temperatures in summer below the dam are usually higher than 20°C, peaking above 22-24°C, which can stress steelhead.

Lewiston Dam has blocked access to >170 km of habitat on the Trinity River since 1963. Unlike the precipitation runoff-fed Klamath and Scott rivers or the spring-fed Shasta River, the headwaters of the Trinity in the Trinity Alps is a snow-fed system. Along with Trinity Dam, located just upstream, the dams have greatly reduced flows, but emergency releases in summer months have also decreased water temperatures, and disrupted the natural hydrograph of the main stem Trinity River. In an effort to restore main stem habitat, the Trinity River Restoration Program (initiated in 2000 as part of the Trinity River Record of Decision) was implemented with the goal of restoring up to 48% of flows into the Trinity River. Since its implementation, summer flows have been augmented, habitat improved, the stream channel and floodplain reconnected, and spawning gravel supplemented.

Hatcheries. Two hatcheries currently operated by the California Department of Fish and Wildlife operate in the KMP as mitigation for lost habitat above Iron Gate and Lewiston Dams. While neither hatchery rears summer-run fish, the Trinity River Hatchery primarily uses native brood stock to support steelhead populations and fisheries for them. These fish are likely comprised almost entirely by Trinity River Hatchery fish or their offspring, and their persistence may depend almost entirely on hatchery operations. Numerous documented transfers of fish from outside the watershed have led to potential hybridization and selection against natural gene flow among fall- and winter- run fish (Prince et al. 2015). The hundreds of thousands of smolts released per year over time from Iron Gate and Trinity River Hatchery have been supporting recreational fisheries, but not contributing to stability of the overall population.

The behavioral (competition and predation) and genetic interactions of juvenile hatchery steelhead with wild steelhead on the Klamath and Trinity Rivers have not been fully evaluated in the KMP and require further attention. Abadia-Cardoso et al. (2013) found that hatchery operations in Northern California selected for fish that matured more quickly and returned to spawn at age-2 rather than at age-3 as their natural-origin counterparts in the wild, and shifted the size and percentage of steelhead exhibiting the half-pounder life history in the Trinity River over time (Peterson 2011). Offspring of hatchery fish were also less likely to return to spawn multiple times over their lifetime. These changes all occurred in a relatively short timeframe, on the order of only a few generations. Naman and Sharpe (2012) conducted a review of predation impacts of hatchery-origin steelhead on wild juvenile salmonids in the Trinity River and found predation rates that were orders of magnitude higher than are found in other watersheds throughout the Pacific Northwest. Over 6% of natural-origin Chinook and coho subyearlings in a year class were consumed by hatchery-origin juvenile steelhead in the study reach below Trinity River Hatchery. Hatchery steelhead are released on March 15 every year proximate to thousands of spawning redds, and during a window where few subyearling fish had emigrated, creating conditions for a highly vulnerable source of prey. In addition, despite the fact that access to cold headwater tributaries for summer steelhead has been severely restricted by dams and water operations, which causes more overlap and genetic exchange between remaining constricted populations that can lead to maladaptation in future runs, NMFS lumps all KMP steelhead together for management (Arciniega et al. 2016).

Where possible, the spatial and temporal overlap between predators (hatchery-origin steelhead) and prey (subyearling Chinook and coho salmon) should be reduced by hatchery operations to reduce high rates of predation. Future investigations should drive completion of a

Hatchery Genetics Management Plan, adaptation of Trinity Hatchery operations, and fisheries reintroduction plan after the four lower Klamath dams are removed. Under such a plan, future hatcheries operations should consider implementing operations that take fish of different run timings and into broodstock to maintain phenotypic differentiation, rather than relying on fish only of a specific run-timing (Arciniega et al 2016).

Logging. Both private and public forest lands in the Klamath Basin have been heavily logged in the past century. In the Smith River basin and other protected coastal streams in the KMP, current logging practices are well managed but legacy effects from past, unregulated, timber harvest may continue to reduce steelhead production in some areas. Contemporary logging, along with associated roads and widespread legacy effects from extensive historical timber harvest, has increased erosion rates of steep hillsides that are prone to landslides and mass wasting in this region, greatly increasing sediment loads in KMP streams (Lewis et al. 2004). Logging with its associated roads and legacy effects (see coho salmon accounts) has increased erosion on steep hillsides, greatly increasing sediment loads in the rivers. High sediment loads cause deep pools to fill with gravel, embed spawning gravels in fine materials, and create shallower runs and riffles. All this decreases the amount of adult holding habitat and increases the vulnerability of the fish to poachers and predators. Such practices, by increasing the rate of run-off, may also decrease summer flows, raising water temperatures to levels that may be stressful or even lethal. Poor watershed conditions caused by logging (and mining) were exacerbated by the effects of the 1964 floods in almost all summer steelhead drainages. These floods deposited enormous amounts of gravel that originated from landslides and mass wasting, especially from areas with steep slopes. The action of the floods not only filled in pools, but also widened stream beds and eliminated riparian vegetation that served as cover and kept streams cooler. The gravel accumulated from late 19th century mining and logging and from the flood is gradually being scoured out of pools, but much of it remains. Potential for further mass wasting along the Trinity, Salmon, and Klamath rivers is high, because logging is still occurring on steep slopes and recent forest fires may be contributing to soil instability (increased by road building).

One indirect effect of habitat loss is increased vulnerability of remaining adult fish to predation. As adult populations are reduced and habitat becomes more restricted, it is more difficult for them to withstand the effects of poaching and natural predation, particularly from river otters. Otter predation on summer steelhead is heaviest when populations of suckers and crayfish, the preferred food of otters, are low, such as occurred in the Middle Fork Eel River following the 1964 flood (A. Naylor, CDFW, pers. comm. 1995). The impact of otters on summer steelhead therefore probably varies from year to year, but could be serious during years when steelhead numbers are already low from other causes.

Juvenile KMP summer steelhead spend critical portions of their lives in tributaries where cool, high-quality water was historically common. Recent reports have documented degradation of this habitat and potential impacts to juvenile salmonid production (KNF and SRRC 2002, Cramer Fish Sciences et al. 2010). Accumulation of gravel in streambeds in recent years has reduced the amount of suitable habitat for summer steelhead by reducing available pool habitat and cover. The shallower, more braided streams also may be warmer, potentially reaching lethal temperature levels. During low flow years, emigrating juveniles can suffer heavy mortality when moving downstream, especially if they become trapped in areas with poor water quality and insufficient flows.

Mining. As indicated above, the legacy effects of mining are often hard to distinguish from the effects of logging and other land use that creates roads, removes vegetation, and generally destabilizes the steep slopes of the coastal mountains. Evidence of direct impacts from mining, historical and current, is apparent in many watersheds in the region especially the Salmon River (e.g., extensive tailing piles, active mining claims and associated equipment or refuse piles, cable crossings, etc.), indicating that mining may still affect KMP steelhead habitats by removing spawning gravels, simplifying and channelizing stream reaches, and reducing rearing habitat (Cramer et al. 2010). As a result, the Scott River has been listed as impaired due to excessive sediment and decreased water quality for nearly two decades. Suction dredge mining has been put on hold in California since 2009, banned in 2016 through SB 637, and recently upheld in the state Supreme Court, striking a decisive victory for salmonids throughout the state (see UKTR spring Chinook account) (CDFW 2016). Unfortunately, some illicit suction dredge mining still probably occurs in remote areas in the basin, far from the public view.

Fire. The lower KMP tributaries are within the marine fog belt, with cooler temperatures and higher fuel moisture levels that inhibit wildfires. However, inland portions of KMP watersheds are subject to frequent and intense fires (e.g., Forks, Salmon, and Corral complex fires, 2013) that, under predicted climate change scenarios, are likely to increase in frequency and intensity. Fires can increase water temperatures of important holding and rearing headwater streams, cause landslides, increase sediment loading, and remove shading canopy cover, all to the detriment of steelhead. Large rainfall events can quickly mobilize the debris from steep slopes and bury spawning and rearing habitats in headwater reaches.

Recreation. Recreational activities in KMP steelhead streams include: angling, boating, gold panning, swimming, hiking, and other outdoor activities. The impacts from recreation upon steelhead, especially at the population level, are likely minimal. Intensive motorized boating (e.g., lower Klamath River) may disrupt movement patterns and, potentially, habitat utilization, but this has not been substantiated.

Harvest. Current fishing regulations prohibit the take of wild steelhead and only hatchery (adipose fin-clipped) steelhead may be harvested. Commercial fisheries operating in the Pacific Ocean rarely contact steelhead (Hayes et al. 2016); likewise, the influence of recreational angling on steelhead abundance is not known, but is assumed to be minimal. Angling pressure on steelhead in the Lower Klamath and the Trinity River near Lewiston can be very high, and likely contributes to some mortality through improper handling and stress during legal catch and release fishing. Tribal net fisheries generally do not target steelhead; however, nets are an indiscriminate method of fishing and may capture both wild and hatchery steelhead, especially larger fish, due to the large net mesh size typically deployed for Chinook salmon. Klamath Mountain Province summer steelhead are particularly susceptible to poaching during summer months, because they are large and conspicuous, and aggregate in canyon pools that preclude exit by low stream flows. In these susceptible habitats, steelhead can be snagged by anglers from the banks or speared by divers. Roelofs (1983) indicated that the most stable populations of summer steelhead are in the most inaccessible streams on public lands, whereas those that are showing signs of severe decline are in areas that are most easily accessible. Roelofs also indicated that poaching was a factor affecting populations of summer steelhead in, at least, the North Fork of the Trinity, New River, and some tributaries to the Klamath River, although current levels of poaching are largely unknown.

The impact of marine (commercial and recreational) fisheries on steelhead is poorly known as they are rarely contacted in ocean fisheries (Hayes et al. 2016); however, these activities may account for some mortality. There are likely hundreds of wild steelhead hooked and handled by recreational fishermen in the KMP each year; these actions can stress fish, alter behaviors temporarily, or even lead to mortality if water temperatures are in excess of 20 degrees Celsius, which they often are in the KMP during summer months (Taylor and Barnhart 2010).

Agriculture. Agriculture, especially for alfalfa irrigation, has affected many KMP streams by altering flows and degrading water quality. Flows in many streams within the KMP steelhead range have been decreased by agricultural diversions and pumping from wells adjacent to streams. In some streams, this may be the biggest factor affecting steelhead abundance. Diversions for intense agriculture, particularly in the low-gradient Scott and Shasta rivers, decrease flows and return "excess" water to rivers (Lewis et al. 2004), thereby reducing the amount of suitable habitat. Return water is typically much warmer than that in the river, after passing through ditches and fields, and is also often polluted with pesticides, herbicides, fertilizers, or animal wastes. Although many diversions in the Scott and Shasta valleys are screened to prevent juvenile salmonid entrainment, screening has not been adequately evaluated. Better agricultural practices and appropriate mitigation measures could dramatically improve salmonid production in the Shasta and Scott valleys (Lewis et al. 2004).

Large-scale marijuana cultivation on public lands in the KMP, which is one of the most heavily used areas of the state for illegal cultivation, may be significantly impacting riparian and aquatic habitats through water diversion, increased sediment inputs, fertilizer and herbicide or pesticide inputs and solid waste inputs (trash dumps or abandoned growing supplies). Cannabis water usage estimates vary widely, but demands for water during the hot summer growing season contribute directly to reduced surface flows, groundwater flows and recharge, and reduction of habitat availability and quality. In a neighboring watershed, this cultivation and illegal withdrawal of water was estimated to reduce low flows by nearly a quarter compared to the seven day average for the summer months, which are critical to summer steelhead. This issue requires further investigation and is confounded by safety risks and subject to insufficient law enforcement involvement, limiting the opportunities to document and reduce impacts from this widespread activity (Bauer et al. 2015). With marijuana cultivation legalization likely in the near future in California, more effort must be placed on understanding, quantifying, and reducing the extent and magnitude of impacts on steelhead habitat in the Klamath Mountains Province.

Grazing. Livestock grazing is common throughout KMP watersheds and, in certain areas, contributes to degradation of aquatic and riparian habitats. Stream bank trampling and removal of riparian vegetation by livestock can cause bank sloughing, stream channel lie-back and head-cutting in meadows, leading to increased sediment loads and higher water temperatures in streams (Spence et al. 1996). Impacts may also include reduction in canopy cover (shading) over stream channels, siltation of pools necessary for juvenile rearing (Moyle 2002), or sedimentation of spawning gravels. Feral cattle near Blue Creek in the lower Klamath may trample riparian habitats and degrade water quality, though these impacts are highly localized (Beesley and Fiori 2008). In areas grazed by large herds or where grazing occurs for extended periods without rotation or exclusion fencing, fecal matter from livestock can also impair water quality and increase nutrient loading, leading to eutrophication (Power et al. 2015).

Transportation. Most KMP steelhead streams are paralleled or crossed by roads, often in many locations. Unsurfaced and unimproved roads (mining, logging, rural residential access) are abundant in the Klamath and Trinity basins and culverts associated with road crossings block access to habitat in many streams, while runoff of fine sediments and pollutants associated with roads can degrade water and habitat quality.

Alien species. Naman and Sharpe (2012) attempted to evaluate the impacts of hatchery releases of juvenile salmonids on natural-origin juveniles in the Trinity River. While they could not quantify predation by invasive brown trout (*Salmo trutta*) on juvenile Chinook, coho, or steelhead in the basin, the impacts are likely small. In addition to direct predation, brown trout may compete with other native salmonids at all life stages for food, rearing and spawning habitat (NMFS 2014). In addition, there is a run of American shad (*Alosa sapidissima*) in the mainstem Klamath River that supports a small recreational fishery, though the impacts of these potential predators on juvenile steelhead are not known at this time (W. Duffy, HSU, pers. comm. 2017).

Table 2. Major anthropogenic factors limiting, or potentially limiting, viability of populations of KMP summer steelhead. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Major dams block access to large areas of spawning and rearing habitat on the Klamath, Trinity, and Shasta rivers, altering stream temperatures, flow, habitat availability and quality.
Agriculture	High	Agriculture and water diversions in the KMP, especially for marijuana cultivation, reduce flows and degrade water quality.
Grazing	Medium	Cattle/livestock grazing may have substantial but localized impacts, especially in the Shasta and Scott river valleys.
Rural/ residential development	Low	Rural development widely dispersed but increasing in the region.
Urbanization	Low	Minimal urban development within the KMP.
Instream mining	Low	Suction dredging is now banned throughout California, but KMP watersheds still suffer legacy effects of past gold mining.
Mining	Low	Impacts from hardrock mines and effluent appear to be low, but legacy impacts from hydraulic mining and dredging (Scott River) have changed its productivity and suitability for spawning.
Transportation	Medium	Most primary streams have roads along almost their entire length and many crossings; roads along rivers degrade water quality and simplify habitats, leading to erosion and runoff of fine sediments.
Logging	Medium	Logging is pervasive in KMP watersheds and degrades habitats; legacy effects in areas without recent logging continue to limit steelhead production through sedimentation and loss of cover.
Fire	Medium	Wildfires are common in KMP watersheds and can result in high sedimentation, exacerbating other habitat alteration stressors; fire frequency and intensity predicted to increase with climate change.
Estuary alteration	Medium	The Klamath River estuary is relatively unaltered; however, the Smith River estuary has lost ~50% of its historical rearing habitat (Quiñones and Mulligan 2005).
Recreation	Low	Habitats used by summer steelhead for holding are particularly sensitive to recreational use because they provide few hides.

Harvest	Low	The sport fishery in the KMP is well regulated; it is illegal to take wild steelhead, though poaching may be limiting in some areas.
Hatcheries	Medium	KMP hatcheries produce nearly one-million juvenile steelhead a year; interactions between wild and hatchery steelhead are detrimental to the recovery of wild stocks.
Alien species	Low	Alien species are somewhat common within KMP watersheds, but impacts to steelhead are unknown.

Effects of Climate Change: Climate change is likely the single largest contributing factor to the long-term decline of stream-maturing life history expression found in summer steelhead in the KMP. Climate change is already having significant impacts on summer steelhead by reducing stream volume, increasing stream temperatures, and altering seasonal flow patterns of water in watersheds containing summer steelhead, which will likely lead to further reduction in suitable upper watersheds that steelhead occupy. The timing of peak flows in the basin has already shifted nearly a month earlier than existed historically (Cayan et al. 2001, Stewart et al. 2005). Flows in snowmelt-fed rivers (e.g., Salmon River, some tributaries to the upper Trinity River) in the Klamath Basin usually peak in winter with a second, smaller, peak in spring and then gradually decrease to their lowest levels in summer. If changes in flow regimes continue at the current rate, then streamflows in the Klamath River Basin are expected to decrease by 10-50% in the spring and summer, while the frequency of extreme high and low flows are predicted to increase by 15-20% (Leung et al. 2004, Kim 2005). Altered flow regimes, due to changes in precipitation patterns, may impair salmonid embryo development and juvenile survival. Extreme high flows can scour redds, flush juveniles into suboptimal habitats before they reach critical size, and alter juvenile outmigration timing to miss the spring oceanic phytoplankton bloom (Mote et al. 2003). Fine (< 4 mm) sediment introduced by intense storm events and associated runoff can smother redds, preventing oxygen from reaching developing embryos or acting as a physical barrier to fry emergence (Furniss et al. 1991). Decreases in summer and fall flows may increase mortality of migrating adult stream-maturing fish and juvenile mortality through stranding. Changes in the timing of peak spring and fall base flows may reduce survival of juveniles emigrating from rivers into the ocean (Lawson et al. 2004).

Salmonids rely on cold water pockets as thermal refuges in rivers during juvenile rearing and adult migration and holding when water temperatures exceed 22°C (Strange 2010). In summer, use of thermal refuges may make juveniles less susceptible to disease (Foott et al. 1999). Climate change influences could diminish or eliminate cold-water pockets as temperatures increase. The reduction of suitable freshwater habitat is also expected to result in a northward and/or higher elevational shift in the range of cold water fishes (Haak et al. 2010). As a result, steelhead in the KMP may experience local extinctions and range contractions, particularly since most higher elevation, headwater streams are inaccessible behind large dams or due to lower summer and fall flows. Once the four lowermost Klamath dams are removed, access to cooler water in historical northern tributaries will be restored.

While multiple large populations of KMP summer steelhead are found in diverse portions of the Klamath and Trinity river basins, persistence of all these populations is likely only with increased protection and with restoration efforts to improve stream flows, allow accessibility to prime holding and spawning habitat, and maintain cool temperatures in headwater tributaries for both spring Chinook salmon and summer steelhead. Ongoing drought in California has likely contributed to a dip in populations of summer steelhead in the KMP, as lower flows and warmer summer water temperatures likely caused increased mortality before fish could spawn. The cumulative impact of these changes is a likely a continued reduction in suitable habitat available for spawning and over-summering (Moyle et al. 2013). More recent research similar to Yamamoto's (2004) habitat utilization study in the New River should be undertaken to determine if emerging temperature/hydrologic regimes under climate change and drought are impacting summer steelhead habitat use, survival, and productivity.

Status Score = 1.9 out of 5.0. Critical Concern. Only 2-3 summer steelhead populations are large enough to expect persistence for more than 10-25 years under present conditions. Most of the smaller stream-maturing populations are likely to disappear in the near future due to shrinking availability of suitable habitat associated with lower streamflows and higher temperatures throughout the summer months in headwater tributaries in the KMP. The long-term decline experienced by KMP summer steelhead is continuing and their eventual extinction as a distinct life history strategy seems likely in the next 50 years if present trends continue.

KMP summer steelhead have a high likelihood of going extinct within the next 50-100 years in California because of lack of strong protection combined with climate change affecting adult holding and juvenile rearing habitat (Table 3). There is a general lack of coordinated basin-wide management actions to protect them, increasing the likelihood of local extirpations. KMP steelhead are recognized as a US Forest Service Sensitive Species and are a Species of Special Concern of CDFG. However, they are not listed by NMFS because they are considered part of the larger KMP steelhead ESU and therefore not separated from the more abundant winter-run steelhead. The most recent genetic data indicates that perhaps this lumping of all KMP steelhead together for management purposes should be revisited based on their differential susceptibilities to anthropogenic and environmental stresses.

Table 3. Metrics for determining the status of the KMP summer steelhead, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Much diminished from historical distribution.
Estimated adult abundance	2	Populations are very small and isolated.
Intervention dependence	3	No intervention is being undertaken to assist in persistence, but is badly needed.
Environmental tolerance	2	Adults require coldwater refuges and pool habitat with cover that is free from human intervention.
Genetic risk	2	Due to the spatial and temporal separation between summer and winter fish, the summer steelhead life history is in jeopardy of extinction in the KMP.
Climate change	1	Highly vulnerable; temperatures and flows already marginal in many areas and summer steelhead require cold water in the warmest months to survive to spawn.
Anthropogenic threats	1	2 High, 6 Medium factors.
Average	1.9	13/7.
Certainty (1-4)	3	Well-documented.

Management Recommendations: Restoration and management recommendations for all KMP steelhead is discussed at length in the KMP winter steelhead account. Conservation recommendations for summer steelhead have been developed for most populations (Jones and Ekman 1980, Roelofs 1983, McEwan and Jackson 1996), but management for the broader stream-maturing life history is not a high priority because they are not listed under state and federal endangered species acts despite their unique life history and genetics (Prince et al. 2015). Present management focuses on increased monitoring to assess population trends and limiting factors of summer steelhead. However, special management is needed in the few watersheds where summer steelhead are most abundant (New, South Fork Trinity rivers; Wooley, Dillon, Clear creeks); it should focus on reducing human impacts and improving habitats, especially keeping water temperatures down such as protecting accessibility to coldwater seeps and springs, remote canyon pools with depth and overhead cover, and decreased disturbance by humans. Recent efforts to increase flows in the cold, spring-fed Shasta River are likely to help steelhead in that tributary, but similar actions are necessary basin-wide.

Management plans for each summer steelhead population should be compiled in a formal Summer Steelhead Management Plan. This plan should address:

(1) Enforcement of fishing and land use regulations in over-summering areas, especially related to groundwater pumping, illegal diversions, marijuana cultivation, etc.
(2) Watershed management to minimize sediment and maintain healthy water quality
(3) Regulation of adult harvest during migrations
(4) Management of downstream reaches to favor out-migrating smolts
(5) Rebuilding present populations through natural and artificial means, where necessary
(6) Restoration of populations that have become extirpated
(7) Protection of adults and juveniles from predation and poaching

Across all populations, there is a need for accurate censuses to identify the factors that limit their numbers. For an accurate assessment, distribution and abundance of all populations, monitoring must expand and be well coordinated within the KMP. Strategies should incorporate approaches from the Steelhead Restoration and Management Plan for California, as well as build upon comprehensive life-cycle monitoring as addressed in the Smith River Fishery Management Plan Draft of 2015. The Coastal Salmonid Monitoring Plan seeks to integrate methods and data across California coastal watersheds, which will help managers assess and utilize the best scientific information available to them. Additional information regarding the genetics, ecology, and behavior of KMP steelhead is also needed and will help inform management and conservation strategies after the four lower dams on the Klamath River are removed.

Improvement of summer steelhead habitat, and thus habitat for half-pounder steelhead and spring-run Chinook, should become a priority for the Department of Fish and Game and other agencies, as reduction in summer carryover habitat has been repeatedly identified as a critical limiting factor. Land management practices which reduces sedimentation, increases cover, and minimizes changes to summer steelhead over-summering habitat should be strongly enforced. More research on summer steelhead populations in California is badly needed, especially to determine (1) genetic identities of each population, (2) extent of possible summer holding areas, (3) distribution of spawning areas and whether they require special protection, (4) habitat requirements of out-migrating smolts, and (5) effects of poaching, illegal marijuana cultivation, and disturbance from recreation on adults.

The highest degree of protection for KMP steelhead (and other fishes) is found in the Wild and Scenic Smith River (Del Norte Co.), which is the largest river in California without a major dam. In 1990, the Smith River National Recreation Area Act provided some degree of protection for the important watershed. In the Klamath Basin, intergovernmental cooperation among tribes, state, and federal agencies, and non-governmental organizations has played an important role in protecting steelhead habitat. Acquiring large tracts of private lands to protect important watersheds, such as Goose, Mill, and Hurdygurdy creeks is a valuable mechanism for conserving steelhead sanctuaries for holding, spawning, and rearing that benefits other salmonids as well. This has since been replicated on Blue Creek, in partnership with the Western Rivers Conservancy and the Yurok Tribe. The entire lower watershed has been acquired through standard and non-standard means (e.g. carbon credits, New Market tax credits, etc.). The land is being handed over to the Yurok Tribe for management in perpetuity as a Salmon Sanctuary. The entire project includes adjacent lands as well, so when all negotiations are completed, there will be 73 square miles of land and stream managed by the Yurok as salmon sanctuary, climate preserve, and sustainable community forest. It is important to note that this project assures that cold water from the creek will continue to flow into the Blue Pool in the river at the mouth of the creek; this pool serves as a cool-water refuge for salmon and steelhead moving up the river when temperatures are warm and flows low in fall months (Western Rivers Conservancy 2014).

Special management considerations and regulations should be afforded to KMP summer steelhead populations. Fishing in the New River and South Fork Trinity during periods when these watersheds are only occupied by summer steelhead and spring Chinook in the late fall should be banned. The potential impact of hooking mortality from legal catch-and-release fishing in the New River and South Fork Trinity during periods when these watersheds are only occupied by summer steelhead and spring Chinook in the late fall may be high and pose serious threats. Although fishing is prohibited in many areas and fines for violations are high, protection of summer steelhead populations may require special guards or streamkeepers for a number of years, as is the case in the neighboring Rogue River watershed in Oregon.

On the Trinity River, CDFW and partnering agencies and organizations have implemented the Trinity Record of Decision to supply ~50% of annual inflow to the river; historically, up to 90% was diverted. There is still important work to be done. CDFW has not yet implemented several key components that will benefit all steelhead life histories, including:

1. Increasing naturally-produced steelhead through protection of selected subbasins (e.g. Blue Creek refuge) that protect steelhead distribution and diversity.
2. Completing management plans for each subpopulation of summer steelhead.
3. Restoring favorable instream conditions to benefit desired ecosystem functions and the community of fishes, including coho and Chinook salmon and coastal cutthroat trout.
4. Reducing hatchery impacts on wild steelhead populations. An Iron Gate Hatchery Genetic Management Plan has been drafted for coho salmon, and current hatchery operations are being evaluated at the Trinity Hatchery. Adapting hatchery operations to utilize new scientific information will benefit all salmonids, especially after removal of the four Klamath dams.

As the Klamath dams are removed, habitat for spawning and rearing should be restored as quickly as possible. Literature suggests that dam removal will increase diversity of life histories and increase resiliency of the population, but this remains to be seen (Hodge et al. 2015). Hatcheries using naturally-produced resident, ocean-maturing, and stream-maturing steelhead ecotypes to repopulate the newly-accessible habitat should also be carefully considered and weighed, as hatchery selection pressures may negatively impact genetic integrity and variation in colonizing *O. mykiss* (Quinones et al. 2013).

KLAMATH MOUNTAINS PROVINCE WINTER STEELHEAD

Oncorhynchus mykiss irideus

Moderate Concern. Status Score = 3.3 out of 5.0. Klamath Mountain Province (KMP) winter steelhead are in a state of decline from historical numbers in the basin. These ocean-maturing fish are relatively more widespread than the stream- maturing summer-run fish, yet still face an uncertain future in the Klamath Basin due to reductions in suitable habitat.

Description: Klamath Mountains Province (KMP) winter steelhead are anadromous rainbow trout that return to select freshwater streams in the Klamath Mountains Province, from the Klamath to the Smith rivers in California. Winter steelhead are distinguishable from other steelhead by (1) time of migration (Roelofs 1983), (2) the mature state of gonads at migration (Shapovalov and Taft 1954) (3) location of spawning in mainstem rivers and tributaries and behavioral traits (Everest 1973, Roelofs 1983). More recently, the early maturing life history has been found to have a genetic basis in the Omy5 gene locus (Pearse et al. 2014). Winter steelhead are nearly identical in appearance to the more rare summer steelhead but are more likely to have spawning colors in the lower reaches of rivers (see description under Northern California coastal winter steelhead).

Taxonomic Relationships: All coastal rainbow trout of North America, including coastal steelhead, have been identified in the subspecies *Oncorhynchus mykiss irideus* (Behnke 1992). CDFW recognize distinct life history variations of steelhead in the KMP Distinct Population Segment (DPS) based upon their timing of freshwater entry, reproductive biology and spawning strategy (Busby et al. 1996, Hodge et al. 2013). These life history variations are generally called winter and summer steelhead, with a distinctive variant known as 'half-pounder' that may be derived from any steelhead of any run-timing. Genetic data support the hypothesis that winter and summer steelhead populations are somewhat distinct (Prince et al. 2015). The KMP winter steelhead are treated separately here from summer steelhead that are part of the same ESU because the two runs and they are distinctive in their behavior and reproductive biology. Observations of run timing suggest presence of fish that return during the fall months, especially in the Trinity River, though the origins of this life history require further study (W. Sinnen, CDFW, pers. comm. 2016, Figure 1). For a more thorough discussion of steelhead that return to the UKTR in the fall, see the KMP summer steelhead account.

Steelhead race	KRSIC (1993)	Hopelain (1998)	USFWS (1979)	Busby et al (1996)	Moyle (2002)
Spring/Summer	May- July	March-June	April-June		April- June
Fall	August- October	July-October	August-November		
Winter	November- February	November-March	November-February		November-April
Stream-maturing				April- October	
Ocean-maturing				September-March	

Figure 1. Classification of different run-timings and reproductive ecotypes of steelhead found in the Klamath River Basin. Modified from Moyle 2002.

The National Marine Fisheries Service (NMFS) does not classify Klamath River basin steelhead as Distinct Population Segments based on run-timing of adults, but instead recognizes two distinct reproductive ecotypes adapted to different sets of environmental conditions. These two ecotypes are stream-maturing (summer) and ocean-maturing (winter) steelhead. Genetic analyses from samples collected between the Klamath River estuary and the confluence of the Trinity River supports these two discrete migrating populations based primarily on timing of freshwater entry and resulting early or late maturation (Papa et al. 2007), correlating with run timing. In the future, KMP winter steelhead should be recognized as a distinct DPS and managed separately from other steelhead based on differing life histories, morphology, and genetics (Hodge et al. 2013, Prince et al. 2015). See the Northern California coastal steelhead account for a full discussion.

KMP winter steelhead are more closely related to KMP summer steelhead than to winter steelhead elsewhere (Pearse et al. 2007, 2016; Prince et al. 2015). Pearse et al. (2007) analyzed genetic samples collected from 30 sites throughout the Klamath River watershed and three Trinity River sites, and found that geographically proximate populations were most similar genetically, even when taking into account the genetic contributions from Iron Gate and Trinity River hatcheries. Winter steelhead appear to contain two genetically distinct populations in the KMP (Papa et al. 2007). Steelhead sampled from the mouth of the Klamath were most similar to other nearby coastal streams (Smith River, Wilson Creek), while fish from the Shasta and Scott rivers clustered closely to steelhead from Iron Gate Hatchery, suggesting that influence of hatchery gene flow (possibly from straying) to these nearby tributaries has occurred over time (Pearse et al. 2007). In addition, these more coastal steelhead expressed limited gene flow with steelhead sampled upstream of the Trinity River confluence, suggesting some population differentiation over time.

Life History: Steelhead are coastal rainbow trout that undergo physiological changes as juveniles, become anadromous, and migrate from the ocean to return to spawn in fresh water. Steelhead/rainbow trout are incredibly plastic species, capable of adapting to and utilizing a wide range of habitats through different life history strategies. Unlike salmon, steelhead can spawn several times throughout their lives (iteroparity). Winter steelhead enter freshwater with fully developed gonads and spawn soon after arriving on spawning grounds. The diversity of habitats within the Klamath Basin yields, for salmonids, perhaps the greatest diversity of life history characteristics and of spatial and temporal habitat usage of any steelhead-bearing river. A total of 38 distinct steelhead life history strategies alone have been identified, including non-anadromous and anadromous forms (Coulter et al. 2013, Hodge et al. 2016). Within this portfolio of life history strategies, three primary pathways emerge: 1) a fast growing group that smolts at age one to two) a non-anadromous, intermediate growth group; and 3) a slow-maturing anadromous group that smolts at age two or three (Hodge et al. 2015). The cues for early migration and smoltification in steelhead that determine when they arrive on spawning grounds have been linked to a specific portion of the fishes' genome known as Omy5 (Pearse et al. 2016). This finding indicates that the basic life history diversity expressed in a given run of steelhead has its basis in a common ancestor and can be passed on to offspring. Nearly three-quarters of steelhead juveniles adopt the life history strategy of their mothers, and that only about 10% of the total population became non-anadromous (Hodge et al. 2010).

Ocean-maturing steelhead enter fresh water between November and April and are generally referred to as winter steelhead. Winter steelhead spawn in mainstem reaches of rivers and tributaries that are mostly passed over by summer steelhead (Roelofs 1983). These fish enter the river as sexually mature adults and spawn shortly after reaching spawning grounds (Busby et al., 1996). After spawning is complete, many winter steelhead migrate back to sea by March or April (Hodge et al. 2015). In the Klamath drainage, 40-64% of the total spawning population are repeat spawners (Hopelain 1998), though post-spawn survival rates differ by tributary. In the Salmon River, for example, one-third of returning adult steelhead are repeat spawners, compared to about 15% in other tributaries (Hodge et al. 2015). Maximum-recorded age of steelhead in recent studies is seven years, and female fecundity has been estimated at 2,000-3,000 eggs per fish (Hodge et al. 2015).

The early life history of winter steelhead in the Klamath and Trinity River basins is fairly well understood. Fry in the Trinity River emerge in April and begin downstream emigration in May, reaching a peak in June and July (Moffett and Smith 1950). Newly emerged steelhead initially move into shallow, protected margins of streams (Moyle 2002). Juveniles aggressively defend territories (Shapovalov and Taft 1954) in or below riffles, where food production is greatest. Moffett and Smith (1950) found steelhead fry favored tributary streams with a peak in downstream movement during the early summer on the Trinity River. When higher flows and lower water temperatures returned to the mainstem during late fall and winter, downstream movement increased. This pattern is assumed to be largely intact, but could be aided by replication of a more natural flow regime below Lewiston dam. Steelhead parr showed the greatest freshwater movement towards the end of their first year and spend their second year inhabiting mainstem rivers. At the Big Bar rotary screw trap downstream of Orleans, a fairly equal proportion of young of year (34%), 1+ (37%) and 2+ (27%) steelhead were captured emigrating downstream over a three-year period (USFWS 2001). The large majority of returning steelhead (86%) in the Klamath River basin apparently spend two years in fresh water before undergoing smoltification and migrating to sea (Hopelain 1998). Average lengths for steelhead smolts entering the ocean ranges from 200 mm (Trinity River) to 270 mm (Shasta River) (Hodge et al. 2015). Klamath River basin steelhead remain mostly in the North Pacific Ocean for one to three years feeding and growing larger before returning to spawn (Hayes et al. 2016).

Half-Pounder Steelhead: In addition to the different run timings of mature steelhead, the KMP is also home a population of smaller, immature fish referred to as 'half-pounders' (Snyder 1925). Half-pounders are small, sub adult fish (25-35 cm) that despite their moniker actually tend to weigh closer to 0.4 kg (Lee 2016). They spend two to four months in the Klamath estuary or nearshore ocean environment before returning to the river in late summer and early fall (between late August and early October) to over-winter and forage in the lower and mid-Klamath river reaches (Kesner and Barnhart 1972, Lee 2016). To confuse things further, some steelhead juveniles may 'double-smolt,' which is a variant of the half-pounder strategy that allows fish to adjust their strategy before they enter the ocean: a fish may undergo smoltification in an estuary but then migrate back upstream in search of better growth and survival conditions rather than are found in the ocean. The following year, these fish smolt again at a larger size and emigrate to the sea to feed and grow before returning to spawn. In the KMP, half-pounders are most common in mainstem and some tributary habitats downstream of Seiad Valley.

The proportion of half-pounders in the total KMP steelhead population likely changes over time as conditions change. During the 1980s, the number of half-pounders in the population at Iron Gate Hatchery was high, resulting in an increase in average size at release of smolts since the 1980s. This practice may have decreased expression of half-pounder life history over time (Peterson and Hankin 2014). During the 1990s, half-pounders in spawning winter steelhead were most common from the mid-Klamath region tributaries (86-100%) when compared to the Trinity River (32-80%) or Lower Klamath River (17%); the lower Klamath fish also demonstrated the greatest first-year growth rates (Hopelain 1998). In a more recent study on this unique life history, less than 10% of these smolts matured precociously to spawn, and adult fish that exhibited the half-pounder life history earlier in life tended to be smaller and less fecund than their ocean-maturing counterparts (Hodge et al. 2014). One important tradeoff is the fact that half-pounders are much more likely to spawn repeatedly over their lifetime than are other ocean-maturing fish. Therefore, the half-pounder life history continues to be viable because the fish enjoy increased survival and spawn more frequently than larger-bodied individuals (Hodge et al. 2014). It is hypothesized that a patch of relatively cool ocean water off the Klamath/Trinidad region of California has contributed to the rise and persistence of this life history, because ocean conditions result in ample feeding habitat for only a short time. Generally, September brings a warm period such that a critical thermal migration corridor to the North Pacific feeding grounds may become closed, so some immature steelhead retreat into the rivers to spend the rest of the year in fresh water (Hayes et al. 2015). The half-pounder life strategy of steelhead in the KMP requires further study.

Habitat Requirements: The KMP encompasses a variety of habitats: from cool fog-belt redwood forests near the coast which experience the highest precipitation in the state, to extreme temperatures in the inland valleys of the Scott and Shasta rivers (Siskiyou Co., SWAP 2015). In the upper portions of KMP watersheds, alluvial valleys that historically supported freshwater marshes and grasslands have been converted to agriculture. In middle and lower reaches, KMP rivers flow through steep mountain slopes in deeply incised canyons with bedrock channels and support fairly narrow riparian habitats (SWAP 2015). While flows in rivers such as the Smith and Klamath are largely precipitation-driven, the Shasta River is spring-fed and some Northern tributaries to the Trinity River rely on snowmelt from the Trinity Alps. This diversity of habitats is the reasons for the diversity of life histories and habitat use strategies that steelhead exhibit in this area.

Habitat requirements of KMP winter steelhead are basically the same as Northern California Coastal winter steelhead (see Northern California winter steelhead account). The majority of a steelhead's life is spent at sea, though relatively little is known about that aspect of their life history, or even where they spend their time in the North Pacific, despite their wide distribution. Many age-0 California steelhead juveniles spend a year feeding in the California Current off the Klamath-Trinidad region, then move northwest to cooler waters offshore in the North Pacific. Recent trawl surveys by NOAA Fisheries indicate that most steelhead from Oregon and California populations mix once reaching salt water and move through surface waters of the Northeast Pacific feeding on pelagic organisms. Recent surveys show that steelhead feed on krill, fish, and amphipods in surface waters far offshore of British Columbia (Hayes et al. 2016). After feeding for several years in a narrow range of sea surface temperatures (apparently 8-14°C), they return to their natal rivers for spawning (Harding 2015, Hayes et al. 2016). Winter steelhead typically enter fresh water from September through March, spawning shortly after migrating to suitable spawning areas (Busby et al. 1996). Due to their migration and spawning period coinciding with periods of high flows, winter steelhead often ascend into tributaries not accessible during low-flow periods. These include streams in medium-sized watersheds that are often not accessible early in the fall because of sediment barriers at the mouths. Spawning habitat for winter steelhead is variable in space and time, resulting in low levels of genetic differentiation among populations in the same watershed (Barnhart 1986, Papa 2007, Prince et al. 2015). Spawning peaks before March throughout the KMP. Little is known about the spawning distribution of winter steelhead throughout the KMP because high, turbid flows make observations difficult.

Juvenile habitat requirements of winter steelhead seem to be similar to those of winter steelhead throughout Northern California; they emigrate into perennial streams soon after hatching (Everest 1973). Roelofs (1983) suggested that use of small streams reduces juvenile mortality because the embryos and fry are less susceptible to predation by larger fish that cannot survive in small streams.

Distribution: The KMP winter steelhead range includes all coastal rivers and creeks throughout the Klamath and Trinity basins and streams north to the Elk River near Port Orford, Oregon. Their range encompasses the Smith River in California and the Rogue and Applegate rivers in Oregon. In the Klamath River, they historically ascended all major rivers and tributaries, and likely spawned in tributaries to Upper Klamath Lake before passage was blocked by a chain of dams on the river (Hamilton et al. 2005). Dam removal of the four lower Klamath dams (Copco 1 and 2, J.C. Boyle, and Iron Gate) is slated to begin in 2020, which will presumably restore access to over 480km of spawning and rearing habitat for steelhead. In the Trinity River, Lewiston Dam blocks upstream access (Moffett and Smith 1950).
At sea, NOAA Fisheries' Southwest Fisheries Science Center salmon trawling surveys have indicated that steelhead were most abundant off the Klamath-Trinidad coast; catch for both juvenile and adult steelhead increased with distance offshore compared to both Coho and Chinook salmon (Harding 2015). It is hypothesized that many of California and Oregon's steelhead feed offshore of the KMP region for some time before migrating northward along thermal corridors to rich feeding grounds in the North Pacific off British Columbia and Alaska, but this is far from certain (Hayes et al. 2015).

Trends in Abundance: Population data are sparse for KMP winter steelhead due to their run-timing. Turbidity, survey repeatability, funding personnel availability, safety concerns and other problems, make it nearly impossible to do monitoring using traditional weirs or spawner surveys during winter months. What information does exist comes from weirs, video cameras, or DIDSON sonar in the fall/early winter. Such information as exists, however, indicates that adult steelhead returns to the KMP have declined from historical numbers and continue to follow a general downward trajectory. Busby et al. (1994) estimated steelhead runs in the basin to average 222,000 adults during the 1960s, but it is unclear to which run these fish belonged. Based on creel and gill net harvest data (Hopelain 2001), the winter steelhead population was estimated at 10,000-30,000 adults per year in the early 1980s in the Klamath River. Returns to the Iron Gate hatchery are highly variable and have significantly declined in recent years despite increases in yearling releases (Figure 2). In general, early hatchery practices saw early-returning fish get selected for spawning over time; it is not clear what impacts this unnatural selection had on the life histories, which have been shown to have a genetic and thus heritable basis (Kendall et al. 2015, Williams et al. 2016, Pearse et al. *In review*), of returning future fish over time. However, no steelhead have been released from Iron Gate Hatchery since 2013, and the program remains in transition until restoration plans have been developed for post-dam removal and a fisheries reintroduction plan and Hatchery Genetics Management Plan have been completed (W. Sinnen, CDFW, pers. comm. 2016). CDFW also used video technology at weirs meant to count salmon and counted 180 adults in the Shasta River (CDFW 2013) and 251 adults on the Scott River (CDFG 2012) for the 2010-2011 and 2011-12 seasons, respectively. However, the cameras did not operate for the entire steelhead migration period (CDFW 2013).

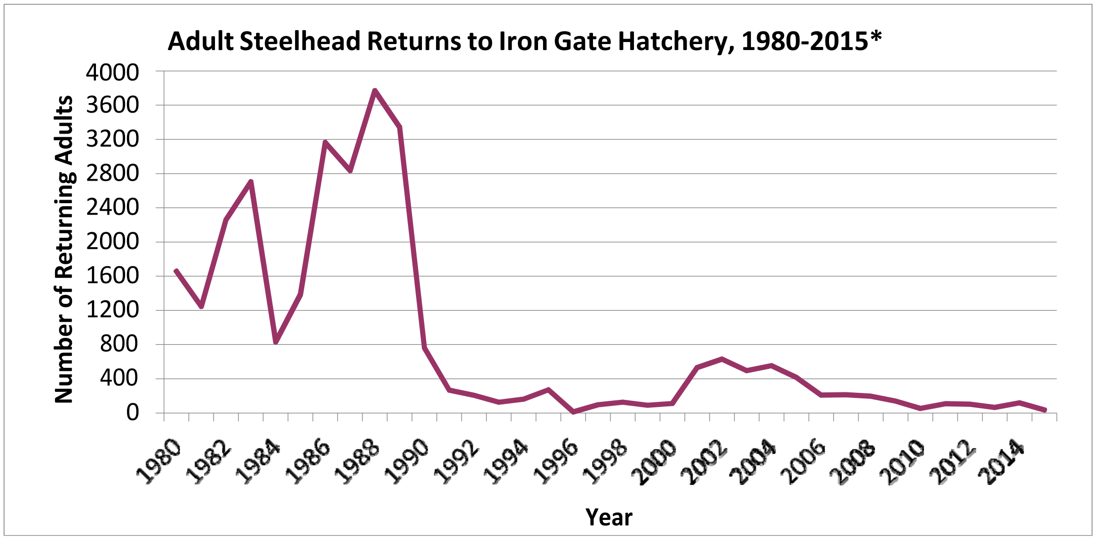


Figure 2. Adult Steelhead Returns to Iron Gate Hatchery. Data from W. Sinnen, CDFW, pers. comm. 2016. *From 1980-89, an average of 200,000 steelhead yearlings were released; from 1990-99, the average was 77,000; from 2000-09, the average increased to 90,000; from 2010-2012, the average was only 32,000. No juvenile steelhead have been released since 2013.

The average historical Trinity River steelhead run was estimated to be in the same range as that of the Klamath River, although more variable, and ranged from 7,800 to 37,000 adults during the 1980s. However, it is unclear which run-timing these totals apply to. Recent estimates of winter-run steelhead are not available due to the difficulty associated with obtaining reliable estimates. They are likely somewhat more abundant than summer-run fish in the UKTR basins (J. Nelson, CDFW, pers. comm. 2016).

In the Smith River, spawning escapement was estimated to be approximately 30,000 adult steelhead during the 1960s. More recently, DIDSON sonar counts were used to estimate 16,000 and 15,000 winter steelhead in the Smith River for the 2010-2011 and 2011-2012 seasons, respectively (Larson 2013).

Factors Affecting Status: Populations of KMP winter steelhead are large enough to support sport fisheries but appear to be in long-term decline in most rivers. The runs in the Klamath-Trinity populations are increasingly supported by hatcheries, while the Smith River remains a stronghold for wild steelhead. The general decline of winter steelhead likely has multiple causes (see Northern California coastal winter steelhead account), including: degradation of river systems by decreased flows from agricultural diversions in the inland portion of the Basin; forestry and other land use practices; inability to pass migration barriers en route to spawning grounds; and human-influenced variation in oceanic and atmospheric conditions related to climate change, especially temperatures (SWAP 2015). The main factors impacting winter steelhead include 1) dams, 2) diversions, 3) logging, 4) agriculture, 4) hatcheries, and 5) harvest.
Dams. Like many rivers in California, the Klamath and Trinity rivers have been dammed but the Smith remains undammed. Three dams that directly affect KMP steelhead in the Klamath Mountains Province are Iron Gate and upstream dams (Klamath), Dwinnell (Shasta), and Lewiston (Trinity) dams. These dams have blocked access to large portions of formerly utilized KMP steelhead habitats, especially important spawning and rearing grounds in the upper portions of both systems. They have altered the main stem by regulating flows, increasing water

diversions (Lewis et al. 2004) and degrading water quality (Hamilton et al. 2011). Iron Gate Dam does not allow fish passage and for now completely blocks access to historical upstream spawning and rearing habitats. Dam operations have decreased the variability, magnitude, duration, and altered timing of flows in the Klamath River. Peak flow timing has also shifted to at least a month earlier than prior to dam construction (Hamilton et al. 2011). Lower flows are of particular concern in the summer because daytime water temperatures can reach 24-26°C across large portions of the Klamath system, reducing available rearing habitat and creating conditions favorable to toxic algae blooms (Hamilton et al. 2011). Juvenile steelhead are likely to persist in the main stem because of abundant food resources and the presence of natural and anthropogenic thermal refuges due to dam operations. Warm temperatures stress steelhead by altering movement, feeding, or growth patterns. Removal of Iron Gate and other upstream dams under the Klamath Basin Agreements will open up over 480 kilometers of potential steelhead habitat in the future; the dam removal project is slated to begin by 2020.Dwinnell Dam has blocked access to greater than 30 km of habitat in the upper Shasta River since its construction in 1928. The dam and multiple diversions have decreased the quality and quantity of habitat by reducing flows and disrupting the natural hydrograph, eliminating peak flows that could improve habitat conditions for steelhead and other salmonids (Lewis et al. 2004). Minimum daytime water temperatures in summer below the dam are usually higher than 20°C, peaking above 22-24°C, which are stressful to steelhead.

Lewiston Dam has blocked access to >170 km of habitat on the Trinity River since 1963. Unlike the precipitation runoff-fed Klamath and Scott rivers or the spring-fed Shasta River, the headwaters of the Trinity River in the Trinity Alps and the Scott River are fed by snowmelt. Along with Trinity Dam, located just upstream, the dams have greatly reduced flows, altered the temperature regime, and disrupted the natural hydrograph of the mainstem Trinity River. In 2000, a Record of Decision was filed, which determined that 48% of Trinity River flows were to be sent downstream to support fisheries and wildlife habitat instead of being diverted at Lewiston Dam via pipe to the Sacramento River. The Trinity River Restoration Program is tasked with allocating Record of Decision flows seasonally and restoring fish habitat in the upper portion of the river. In 2003, a flow regime with lower spring and higher summer and early fall flows than were observed historically was initiated with comprehensive physical/mechanical restoration to restore the riparian corridor and fisheries of the Trinity River. Significant projects have been completed to permit greater flows, reconnect the floodplain with the river channel to improve juvenile rearing habitat, and place spawning gravel in the channel to restore spawning areas.

Diversions. Flows in many KMP streams have been reduced by domestic and agricultural diversions and pumping from adjacent wells. In the Scott and Shasta rivers, diversions may be the single greatest factor affecting steelhead numbers by reducing flows, habitat availability and quality through increases in water temperatures and nutrients from returning 'excess' water to the river (Lewis et al. 2004). Return water is warmed by its passage through ditches and fields and is often polluted with nutrients from animal waste as well. Many of the diversions in the Scott and Shasta valleys are screened to prevent loss of juvenile salmonids in the diversions, but their effectiveness has not been adequately evaluated.

Logging. Both private and public forest lands in the Klamath Basin have been heavily logged in the past century. In the Smith River basin and other protected coastal streams in the KMP, current logging practices are well managed but legacy effects from past, unregulated, timber harvest may continue to limit steelhead production in some areas. Contemporary logging, along with associated roads and widespread legacy effects from extensive historical timber harvest, has increased erosion rates of steep hillsides that are prone to landslides and mass wasting in this region, greatly increasing sediment loads in KMP streams (Lewis et al. 2004). Logging with its associated roads and legacy effects (see SONCC Coho salmon account) has increased erosion on steep hillsides, greatly increasing sediment loads in the rivers. High sediment loads fill deep pools with gravel, embed spawning gravels in fine materials, and create shallower runs and riffles. All this decreases the amount of adult holding habitat and increases the vulnerability of the fish to poachers and predators. Such practices, by increasing the rate of run-off, may also decrease summer flows, raising water temperatures to levels that may be stressful or even lethal. One indirect effect of habitat loss is increased vulnerability of remaining adult fish to predation, both from humans and natural predators such as otters (A. Naylor, CDFW, pers. comm. 1995). The effects of logging are especially severe in tributaries where steelhead concentrate for spawning and rearing. For example, increased sedimentation of spawning grounds leads to reduction of embryo survival and alevin emergence rates in the Shasta and South Fork Trinity rivers (Burns 1972).

Poor watershed conditions caused by logging (and mining) were exacerbated by the effects of the 1964 floods in almost all summer steelhead drainages throughout Northern California. These floods deposited enormous amounts of gravel that originated from landslides and mass wasting, especially from areas with steep slopes. The flood filled in pools, widened streambeds, and eliminated riparian vegetation that served as cover and kept streams cooler. The gravel is gradually being scoured out of pools, but much of it remains. Potential for further mass wasting along the Trinity, Salmon, and Klamath rivers is high because logging still occurs on steep slopes and recent forest fires may be contributing to soil instability through increased by road building. In addition, in many streams, improperly constructed culverts under logging roads are barriers to upstream spawning and rearing areas. Accumulation of gravel in streambeds in recent years has reduced the amount of suitable habitat for summer steelhead by reducing available pool habitat and cover. The shallower, more braided streams also may be warmer, potentially reaching lethal temperature levels. During low flow years, emigrating juveniles can suffer heavy mortality when moving downstream, especially if they become trapped in areas with poor water quality and insufficient flows. In the Smith River and small coastal streams, impacts of logging are less pronounced, especially where watersheds are protected, but legacy effects of past logging still limit the ability of habitat to produce steelhead.

Mining. Legacy effects of 19th century hydraulic mining still negatively affect KMP steelhead habitats in many areas. Historical mining was widespread and intensive in this region and, in combination with logging, devastated many watersheds. Legacy effects of mining may be difficult to distinguish from contemporary impacts from logging, rural development, and other land uses that require road building, vegetation removal, or other landscape alterations that contribute to destabilization of the steep slopes of the Klamath Province and increased sediment loads in rivers and streams. The upper Klamath, Salmon, Scott, and Trinity rivers bear evidence of mining impacts (e.g., extensive tailing piles, active mining claims and associated equipment or refuse piles, cable crossings, etc.), indicating that mining may still affect KMP steelhead habitats by removing spawning gravels, simplifying and channelizing stream reaches, and reducing rearing habitat (USFS and Salmon River Restoration Council 2002, Cramer et al. 2010). As a result, the Scott River has been listed as impaired due to excessive sediment and decreased water quality for nearly two decades, although historical impacts were almost certainly greater than they are today. A ban on suction dredge mining has been upheld in the California Supreme Court, marking a decisive victory for salmonids throughout the state (CDFW 2016).

Fire. The lower KMP tributaries are within the marine fog belt, with cooler temperatures and higher fuel moisture levels that inhibit wildfires. However, inland portions of KMP watersheds such as the Salmon, Scott, and Trinity rivers are subject to frequent and intense fires (e.g., Forks, Salmon, and Corral complex fires, 2013) that are predicted to increase in frequency and intensity under climate change scenarios. Fires can increase water temperatures in holding and rearing headwater streams, cause landslides, increase sediment loading, and remove shading canopy cover, all to the detriment of steelhead. Large rainfall events can quickly mobilize the debris from steep slopes and bury spawning and rearing habitats in headwater reaches.

Recreation. Recreational activities in KMP steelhead streams include: angling, boating, gold panning, swimming, hiking, and other outdoor activities. The impacts from recreation upon steelhead, especially at the population level, are likely minimal. Intensive motorized boating (e.g., lower Klamath River) may disrupt movement patterns and, potentially, habitat utilization, but this has not been substantiated.

Harvest. Current fishing regulations prohibit the take of wild steelhead and only hatchery (adipose fin-clipped) steelhead may be harvested and the influence of recreational angling on steelhead abundance is assumed to be minimal but not known with certainty. Angling pressure on steelhead in the Lower Klamath and the Trinity River near Lewiston can be very high, and likely contributes to some mortality through improper handling and stress during legal catch and release fishing. Tribal net fisheries do target steelhead seasonally; however, the effects of this harvest on both wild and hatchery steelhead is largely unknown (W. Duffy, HSU, pers. comm. 2017). The impact of marine (commercial and recreational) fisheries on steelhead is poorly known. Steelhead are rarely documented in ocean fisheries (Hayes et al. 2016); however, these activities may account for some mortality.

Agriculture. Agriculture, especially irrigated pasture and alfalfa for livestock grazing in the inland portions of the Province, impacts streams throughout the Klamath and Trinity basins through both runoff and sedimentation. The Shasta and Scott valleys have been identified as two regions where improved agricultural practices and corresponding diversions and groundwater pumping could dramatically increase salmon and steelhead populations in the Klamath Basin (Lewis et al. 2004).

Large-scale marijuana cultivation on public lands in the KMP, which is one of the most heavily used areas of the state for illegal cultivation, may be significantly impacting riparian and aquatic habitats through water diversion, pollution from sediment, fertilizer, pesticide inputs, and trash dumps or abandoned growing supplies. Cannabis water usage estimates vary widely, but demands for water during the hot summer growing season contribute directly to reduced surface flows, groundwater flows and recharge, and reduction of habitat availability and quality in small headwater streams. In the neighboring Eel River and Redwood Creek watersheds, this cultivation and illegal withdrawal of water was estimated to reduce low flows by nearly a quarter compared to the seven day average for the summer months. This issue requires further investigation and is confounded by safety risks and subject to insufficient law enforcement involvement, limiting the opportunities to document and reduce impacts from this widespread activity (Bauer et al. 2015). With marijuana cultivation legalization likely in the near future in California, more effort must be placed on understanding, quantifying, and reducing the extent and magnitude of impacts on steelhead habitat in the Klamath Mountains Province.

Grazing. Livestock grazing is common throughout KMP watersheds and, in the Scott and Shasta valleys contributes to degradation of aquatic and riparian habitats. Stream bank trampling and removal of riparian vegetation by livestock can cause bank sloughing, stream channel lie-back and head-cutting in meadows, leading to increased sediment loads and higher water temperatures in streams (Spence et al. 1996). Impacts may also include reduction in canopy cover (shading) over stream channels, siltation of pools necessary for juvenile rearing (Moyle 2002), or sedimentation of spawning gravels. In areas grazed by large herds or where grazing occurs for extended periods without allotment rotation or exclusion fencing, fecal matter from livestock can also impair water quality and increase nutrient loading, leading to eutrophication, as in the Smith River estuary (J. Garwood, CDFW, pers. comm. 2016). In addition, feral cattle near Blue Creek in the lower Klamath may trample riparian habitats and degrade water quality, though these impacts are highly localized (Beesley and Fiori 2008).

Transportation. Most KMP steelhead streams are paralleled or crossed by roads, often in many locations. Unsurfaced and unimproved roads (mining, logging, and rural residential access) are abundant in the Klamath and Trinity basins and culverts associated with road crossings block access to habitat in many streams, while runoff of fine sediments and pollutants associated with roads can degrade water and habitat quality.

Hatcheries. Two hatcheries currently operated by the California Department of Fish and Wildlife operate in the KMP as mitigation for lost habitat above Iron Gate and Lewiston Dams. While hatchery production on the Trinity River has primarily relied upon native brood stock to boost production, there have been numerous documented transfers of fish from outside the respective watersheds, leading to potential hybridization and selection against natural gene flow among winter-run fish (Prince et al. 2015). The hundreds of thousands of smolts released per year over time from Iron Gate and Trinity River Hatchery have been supporting recreational fisheries but not contributing to stability of the overall population. In fact, before Iron Gate Hatchery stopped releasing winter-run steelhead juveniles in 2013, a significant percentage of returning fish were resident *O. mykiss* that never migrated to the ocean and remained in the Klamath River (Jong, 1994). It is assumed that poor location of the hatchery itself (J. Nelson, CDFW, pers. comm. 2016) and poor water quality in the mainstem Klamath River below Iron Gate Dam could have selected for resident life history over anadromous life history of Iron Gate Hatchery steelhead and contributing to their precipitous decline (USFWS 1999).

The behavioral (competition and predation) and genetic interactions of juvenile hatchery steelhead with wild steelhead on the Klamath and Trinity Rivers have not been evaluated, but are recognized as issues requiring attention, as are adult competitive interactions (CDFG 2001). Abadia-Cardoso et al. (2013) found that hatchery operations in Northern California selected for fish that matured more quickly and returned to spawn at age-2 rather than at age-3 as their natural-origin counterparts in the wild. Offspring of hatchery fish were also less likely to return to spawn multiple times over their lifetime. These changes all occurred in a relatively short timeframe, on the order of only a few generations. Future investigations such as these will guide adaptation of Trinity Hatchery operations and drive completion of a Hatchery Genetics Management Plan and fisheries reintroduction after the four lower Klamath dams are removed.

In addition to these state-managed hatcheries, a small private hatchery called Rowdy Creek Hatchery operates near the mouth of Rowdy Creek (Smith River, Del Norte County). This hatchery is operated by the local Kiwanis Club and runs on donations. They raised over 100,000 yearling steelhead a year from returning wild and fin-clipped hatchery steelhead for several decades, with the last steelhead returning to be spawned in 2006 (http://www.rowdycreek.com/).

In 2005 and 2006, 564 and 2,231 steelhead adults were spawned, respectively. The impacts of this relatively small hatchery operation are likely small but unknown.

Alien species. Naman and Sharpe (2012) could not quantify predation by invasive Brown trout (*Salmo trutta*) on juvenile Chinook, Coho, or steelhead in the Klamath-Trinity Basin, but the impacts are likely non-negligible. In addition to direct predation, Brown trout compete with other native salmonids at all life stages for food, rearing and spawning habitat (NMFS 2014). A small but persistent run of American shad (*Alosa sapidissima*) also inhabit the Klamath River, and could potentially prey on juvenile steelhead (W. Duffy, HSU, pers. comm. 2017).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of KMP winter steelhead. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	Medium	Major dams block access to habitat on the Klamath, Trinity, and Shasta rivers, reducing habitat availability and quality.
Agriculture	Medium	Agriculture and diversions in the KMP, especially for illegal marijuana cultivation, reduce flows and degrade water quality.
Grazing	Medium	Cattle/livestock grazing may have substantial but localized impacts, especially in the Shasta and Scott river valleys.
Rural/ residential development	Low	Rural development is widely dispersed but increasing.
Urbanization	Low	Minimal urban development within the KMP.
Instream mining	Low	Suction dredging is now banned throughout California, but KMP watersheds still suffer legacy effects of past gold mining.
Mining	Medium	Impacts from hardrock mines and their effluents appear to be low, but legacy impacts from hydraulic mining and dredging in portions of the Klamath, Scott, Salmon, and Trinity rivers have fundamentally changed productivity and suitability for spawning.
Transportation	Medium	Most primary streams have roads along their length and many crossings, degrading water quality and simplifying habitats.
Logging	Medium	Logging is pervasive in KMP watersheds and continues to degrade habitats; legacy effects limit steelhead production.
Fire	Medium	Wildfires are common in KMP watersheds and can result in sedimentation, exacerbating other habitat alteration stressors.
Estuary alteration	Medium	The Smith River Estuary has lost ~50% of its historical rearing habitat (Quiñones and Mulligan 2005).
Recreation	Low	Recreational impacts are low.
Harvest	Low	It is illegal to take wild steelhead, though poaching may be a limiting factor in some areas. Tribal harvest poorly understood.

Hatcheries	Medium	The Trinity River hatchery produces hundreds of thousands of steelhead a year; interactions between wild and hatchery steelhead are likely detrimental to the health of wild stocks.
Alien species	Low	Brown trout are present in the Trinity River and likely compete with and prey on other salmonids, though impacts not quantified.

Effects of Climate Change: Streams in the Klamath Basin downstream of Iron Gate Dam are projected to be warmer and drier during the summer and fall months due to reduction in total snowpack and seasonal retention of snow as the climate continues to change (Hamlet et al. 2005, Stewart et al. 2005). Snow pack water content in the last 50 years has already significantly declined at several monitoring stations in the Klamath Basin (Van Kirk and Naman 2008), leading to lower flows and increasing water temperatures (Allan and Castillo 2007). Climate change may also alter streamflow patterns by increasing winter runoff of rain rather than snow, likely decreasing spring and summer stream flows, and increasing the occurrence of winter floods and summer droughts (Field et al. 1999). The timing of peak flows in the basin has already shifted nearly a month earlier than existed historically (Cayan et al. 2001, Stewart et al. 2005). Flows in snowmelt-fed rivers (e.g., Salmon River, some tributaries to the upper Trinity River) in the Klamath Basin usually peak in winter with a second, smaller, peak in spring and then gradually decrease to their lowest levels in summer. If changes in flow regimes continue at the current rate, then streamflows in the Klamath River Basin are expected to decrease by 10%-50% in the spring and summer, while the frequency of extreme high and low flows are predicted to increase by 15-20% (Leung et al. 2004, Kim 2005).

Increases in water temperatures will strongly affect the physiology and behavior of salmonids at each stage of life and their life history strategies. Changes in movement patterns are likely to be the most obvious response of individual salmonids to climate change, particularly as fish are exposed to increases in water temperature and changes in stream flow patterns (Groot and Margolis 1991). Increased temperatures will hasten developmental and growth rates, and may shift migration patterns of Klamath salmonids to earlier in the year. However, photoperiod at a given site can also influence the initiation of salmonid migrations; thus, migration initiation and timing may become decoupled from water temperature up to a point (Feder et al. 2010). Salmonids may also seek colder waters as a method of thermoregulation under warmer streamflow conditions. Salmonids use cold water pockets as thermal refuges in rivers during juvenile rearing and adult migration when water temperatures exceed 22°C (Strange 2010). In summer, juveniles use thermal refuges to avoid disease (Foott et al. 1999). Climate change influences could diminish or eliminate cold-water pockets as temperatures increase. The reduction of suitable freshwater habitat is also expected to result in a northward and/or higher elevational shift in the range of cold water fishes (Haak et al. 2010). As a result, steelhead in the KMP may experience local extinctions and range contractions, particularly since most higher elevation, headwater streams are inaccessible behind large dams or due to lower summer and fall flows. Once the four lowermost Klamath dams are removed, access to cooler water in historical northern tributaries will be restored. In addition, fall water temperatures are forecast to decrease several weeks earlier than they currently do after dam removal, with important implications for habitat suitability and run-timing of salmonids (W. Duffy, HSU, pers. comm. 2017).

Altered flow regimes, due to changes in precipitation patterns, may impair salmonid embryo development and juvenile survival. Extreme high flows can scour redds, flush juveniles into suboptimal habitats before they reach critical size, and alter juvenile outmigration timing to miss the spring oceanic phytoplankton bloom (Mote et al. 2003). Fine (< 4 mm) sediment introduced by intense storm events and associated runoff can smother redds, preventing oxygen from reaching developing embryos or acting as a physical barrier to fry emergence (Furniss et al. 1991). Decreases in baseflows may increase juvenile mortality through stranding and changes in the timing of peak spring and base flows may reduce survival of juveniles emigrating from rivers into the ocean (Lawson et al. 2004). Increases in winter flows may decrease adult survival or reproductive success due to the higher metabolic cost of upstream migration at higher flow stages. Due to these factors, Moyle et al. (2013) rated KMP winter steelhead as highly vulnerable to extinction from to climate change, using a systematic procedure, while recognizing at present KMP are not in immediate danger.

Status Score = 3.3 out of 5.0. Moderate Concern. There is no immediate extinction risk for KMP winter steelhead. Some populations will likely decline further or even be extirpated under current management trends. KMP winter steelhead are managed by CDFW for recreational harvest, and recruits from the Trinity River Hatchery prop up populations. In 2001, NMFS found that the DPS as a whole did not warrant listing as a threatened species under the Endangered Species Act. However, Klamath Mountain Province steelhead are listed by the US Forest Service Pacific Southwest Region as a Sensitive Species, and a Species of Special Concern by CDFW (CDFW 2015).

Lack of strong protection for wild stocks, combined with climate change reducing adult holding and juvenile rearing habitat, will continue to pressure KMP winter steelhead. The absence of coordinated province-wide management actions to protect them increases the likelihood of local extirpations. The most recent genetic data indicating that summer and winter steelhead in the KMP are discrete units (Prince et al. 2015) raises the question of whether or not lumping these two runs together for management purposes should be revisited.

Table 2. Metrics for determining the status of KMP winter steelhead, where 1 is poor value and 5 is excellent and 2-4 are intermediate values. Certainty of these judgments is moderate.

Metric	Score	Justification
Area occupied	4	Winter steelhead found throughout KMP watersheds; dams block access to large portion of upstream spawning and rearing habitat.
Estimated adult abundance	3	KMP winter steelhead abundance is relatively unknown, but is probably less than 50,000 fish/year including Smith River run.
Intervention dependence	3	Frequent management actions needed for habitat restoration and protection to prevent continuation of long-term decline. Klamath dam removal and restoration is likely to improve this score.
Tolerance	4	Steelhead are very adaptable and winter fish return during a time when flows are highest and temperatures lowest.
Genetic risk	3	Presumably genetically diverse; however, hybridization risk with hatchery steelhead is a considerable threat.
Climate change	3	All KMP watersheds are projected to see seasonal water temperatures and flows change.
Anthropogenic threats	3	9 Medium factors.
Average	3.3	23/7.
Certainty (1-4)	2	Data are particularly sparse for KMP winter steelhead.

Management Recommendations: KMP winter steelhead appear to be in slow decline, although fish mainly of hatchery origin continue to support a fishery. Climate change will increase stream temperatures, reduce summer flows and otherwise change flow patterns, and will have to be taken into account for all management actions.

Klamath River actions. Wild Klamath Basin steelhead are particularly at risk; present numbers are far below estimates from even two decades ago and the Iron Gate Hatchery has not met production or escapement goals for decades. Returns of anadromous adult steelhead to the Hatchery have been so low over the last decade that release of all steelhead stopped in 2013 (M. Knechtle, CDFW, pers. comm. 2016). It is presumed that hatchery fish overwhelmingly exhibit resident life history strategies in the Klamath Basin due to prevailing environmental conditions (W. Duffy, HSU, pers. comm. 2017). Although currently-accessible Klamath tributaries (e.g., Salmon River and Clear, Dillon, and Elk creeks) provide healthy spawning and nursery areas, water quality and quantity in the mainstem may remain too seasonally poor to provide connectivity between these locations and for rearing habitat of larger juveniles. If coordinated restoration efforts are successful following dam removal on the Klamath River and flows improve in the Trinity River, it is possible to optimistic about the state of KMP steelhead populations in California in the next 15-20 years. Numbers could increase significantly once connections are re-established with Klamath River above Iron Gate and the upper Klamath Basin but there is considerable uncertainty as to how much steelhead will benefit (Hodge et al. 2015). Use of conservation hatcheries to 'jump-start' resident, ocean-maturing, and stream-maturing steelhead populations in the newly-accessible habitat should be carefully considered, recognizing that hatchery selection pressures can negatively impact genetic integrity, behavior, and over all fitness (Quinones et al. 2013). An Iron Gate Hatchery Genetic Management Plan (CDFW 2014) has been drafted for Coho salmon, which may be able to serve as a model for adapting hatchery operations benefit all salmonids, especially after removal of four Klamath dams. An alternative option would be closure of Iron Gate Hatchery for a full decade after dam removal, with monitoring for relative abundances of wild and hatchery fish, while the Trinity River hatchery continues to support the fishery (Quinones et al. 2013). This could offer scientists a chance to better understand the carrying capacity of the habitat left behind on the Klamath River after dam removal, and provide valuable data that could inform recovery of salmonids to levels deemed appropriate for the time.

A particularly promising approach to improving the status of steelhead and other Klamath salmonids is found in Blue Creek, the lowest substantial tributary to the river, has headwaters in the Siskiyou Wilderness Area. The entire lower watershed has been acquired through standard and non-standard means (e.g. carbon credits, New Market tax credits, etc.). The land will eventually pass entirely to the Yurok Tribe for management in perpetuity as a Salmon Sanctuary. The entire project includes adjacent lands as well, so when all negotiations are completed, there will be 73 square miles of land and stream managed by the Yurok as salmon sanctuary, climate preserve, and sustainable community forest. It is important to note that this project assures that cold water from the creek will continue to flow into the Blue Pool in the river at the mouth of the creek; this pool serves as a cool-water refuge for salmon and steelhead moving up the river when temperatures are warm and flows low in fall months (Western Rivers Conservancy 2014).

Trinity River actions. Trinity River Restoration Program actions, such as improved flows, manipulation of shallow edge habitats, and removal of barriers, will benefit steelhead populations. The California Department of Fish and Wildlife, along with partnering agencies and organizations, has implemented the Trinity Record of Decision to supply ~50% of annual inflow to the river. However, several key components have not been implemented, including:

1. Increasing naturally produced steelhead through protection of selected subbasins that protect steelhead distribution and diversity.
2. Completing management plans for each subpopulation of winter steelhead.
3. Restoring favorable instream conditions to benefit desired ecosystem function and the community of fishes, including Coho and Chinook salmon and Coastal Cutthroat trout downstream.
4. Improving hatchery operations for the Trinity River Hatchery to reducing hatchery impacts on wild steelhead populations.

Smith River actions. The Smith River remains relatively undisturbed, with major conservation activities taking place within the watershed, so it is likely to remain a stronghold for wild steelhead regardless of what happens in other watersheds. In 1990, the Smith River National Recreation Area Act provided some degree of additional protection for the important watershed. As in the Klamath Basin, intergovernmental cooperation among tribes, state, and federal agencies, and non-governmental organizations has played an important role in protecting steelhead habitat. The conservation strategy of acquiring large tracts of private lands to protect important watersheds, such as Goose, Mill, and Hurdygurdy creeks is important for conserving steelhead sanctuaries for holding, spawning, and rearing that benefits other salmonids as well. This has since been replicated on Blue Creek, in partnership with the Western Rivers Conservancy and the Yurok Tribe, and should be replicated where possible to ensure strongholds exist for KMP winter steelhead. In 2016, the Red Flat Nickel mine proposal in the Oregon portion of the Smith River watershed was defeated by legislation, setting further precedent of protecting this National Wild and Scenic River into the future.

Research and monitoring. More research on KMP steelhead populations in California is also needed, especially to determine (1) genetic identities of each population, (2) extent of possible limiting holding and rearing areas, (3) distribution of spawning areas and whether they require special protection, (4) habitat requirements of out-migrating smolts, and (5) effects of poaching, illegal marijuana cultivation, and disturbance from recreation on adults. Managers would benefit from a better understanding of the physical and biological cues that lead to their wide variety of migration and habitat utilization patterns. Determination of survival and escapement rates for wild steelhead is essential to understanding viability and persistence of individual populations. Additional information regarding the genetics, ecology, and behavior of KMP steelhead is also needed to help inform management and conservation strategies after the four lower dams on the Klamath River are removed. For example, hooking mortality from legal catch-and-release fishing on wild winter steelhead has unknown consequences to small populations, and new information could better inform updated regulations. Although penalties for fishing violations are high, increased vigilance and enforcement are required to help maintain and recover wild steelhead populations where they remain

Across all KMP winter steelhead populations, there is a need to accurately census distribution and abundance of populations. Monitoring strategies should incorporate approaches from the Steelhead Restoration and Management Plan for California, as well as build upon comprehensive life cycle monitoring as addressed in the Smith River Fishery Management Plan (Draft, 2015). The Coastal Salmonid Monitoring Plan seeks to integrate methods and data across

California coastal watersheds, which can help managers use the best scientific information available to them to manage the resource. More comprehensive monitoring could help adjust fishing regulations to reduce impacts on wild fish.

General actions. Improvement of juvenile steelhead rearing habitat should be a priority for all agencies, because reduction in summer rearing and carryover habitat has been repeatedly identified as a critical limiting factor for steelhead in the Klamath Mountains Province. Land management practices that reduce sedimentation, increase cover, and minimize changes to rearing habitat should be identified and implemented where practicable.

KMP winter steelhead populations would benefit from restoration actions that reduce impacts from logging, mining, and illegal and quasi-legal marijuana cultivation in the basin. Land management strategies that seek to reduce sedimentation, increase cover, and minimize other stressors that negatively affect habitat for adults are critical to recovering populations. Continued funding for upslope restoration on private lands, fencing riparian areas, and improving water conservation will be necessary at a watershed scale, with greater participation by landowners, for there to be a benefit to KMP steelhead in places like the Shasta and Scott Rivers. Removal of migration barriers in tributaries, replanting riparian areas, adding complex woody debris to stream channels, and reducing sediment reaching rivers and streams are also necessary.

Future management measures to bolster KMP steelhead should address: (1) improving water management to mimic natural hydrographs, and especially to reduce undocumented water withdrawals, (2) identifying watershed management approaches that minimize sediment delivery to streams and maintain high water quality, (3) implementing restoration of downstream reaches to favor out-migrating smolts, (4) rebuilding present wild populations through identifying and affording protection to key refuge areas, such as the Smith River, that protect genetic and life history strategy diversity, and (5) creating a basin-wide restoration program involving stakeholders, managers, and policymakers from the upper and lower Klamath Basins to allow rapid implementation of a common vision for the restored Klamath River after dam removal. These strategies should incorporate approaches from the Steelhead Restoration and Management Plan for California, as well as build upon comprehensive life cycle monitoring as addressed in the Smith River Fishery Management Plan (Draft, 2015). The Klamath dam removal and restoration project will be the largest such restoration project ever undertaken in the Klamath Basin. This action provides an exciting opportunity to showcase cooperation and serve as a proving ground for salmonid reintroduction and management to the benefit of all stakeholders and species.

NORTHERN CALIFORNIA SUMMER STEELHEAD
Oncorhynchus mykiss irideus

Critical Concern. Status Score = 1.9 out of 5.0. Northern California (NC) summer steelhead are in long-term decline and this trend will continue without substantial human intervention on a broad scale. Due to their reliance on cold water to over summer during the warmest months in freshwater and critical susceptibility to climate change, NC summer steelhead are vulnerable to extinction by 2050.

Description: Summer steelhead are anadromous rainbow trout which return to freshwater from the ocean as large, silvery, immature trout with numerous black spots radiating outward on their tail, adipose and dorsal fins, and are nearly identical in appearance to winter steelhead. Their backs are an iridescent blue to olive or nearly brown. Their sides and belly appear silver, white, or yellowish with an iridescent pink to red lateral band. The mouth is large, with the maxillary bone usually extending beyond the eyes, which are above pinkish cheeks (opercula). Teeth are well developed on the upper and lower jaws, although basibranchial teeth are absent. The dorsal fin has 10-12 rays; the anal fin, 8-12 rays; the pelvic fin, 9-10 rays; and the pectoral fins 11-17. The scales are small with 110-160 scales along the lateral line, 18-35 scale rows above the lateral line, and 14-29 scale rows below it (Moyle 2002).

The coloration of juveniles is similar to that of adults except they have 5-13 widely spaced, oval parr marks centered on the lateral line with interspaces wider than the parr marks. Juveniles also possess 5-10 dark marks on the back between the head and dorsal fin, which make the fish appear mottled. There are few to no spots on the tail of juveniles and white to orange tips on the dorsal and anal fins. There are spots on the dorsal fin, unlike Chinook and Coho salmon. Resident adult trout may retain the color patterns of parr (Moyle 2002).

The various forms in California show slight morphological differences (Bajjaliya et al. 2014), and are mainly distinguished by genetics, behavior, and life history strategies, although different populations may show some variation in the average size of returning adults. Summer steelhead are distinguishable from other steelhead by (1) time of migration (Roelofs 1983), (2) the immature state of gonads at migration (stream-maturing ecotype) (Shapovalov and Taft 1954, NMFS 2016), (3) location of spawning in higher-gradient habitats and smaller tributaries than other steelhead (Everest 1973, Roelofs 1983), and more recently, genetic variation in the Omy5 gene locus, which helps trigger migration timing in individuals (Pearse et al. 2014).

Taxonomic Relationships: Until the late 1980s, all steelhead were listed as *Salmo gairdneri gairdneri*. However, Smith and Stearley (1989) showed that steelhead are closely related to Pacific salmon (genus *Oncorhynchus*) and are conspecific with Asiatic steelhead, then called *Salmo mykiss*. As a result, the American Fisheries Society recognizes coastal rainbow trout, including steelhead, as *Oncorhynchus mykiss*. All steelhead and nonmigratory Coastal Rainbow trout are usually lumped together as *O. m. gairdneri* or, more recently, as *O. m. irideus* (Behnke 1992). However, in this report, *O. mykiss* upstream of a manmade barrier are discussed in the account with their relevant DPS downstream, and O. mykiss above natural barriers to anadromy and in inland waters are discussed in the coastal rainbow trout account.

Moyle (2002) discusses the complex systematics of California populations of steelhead. The six genetic units (ESUs and DPSs) recognized by NMFS for California have more or less discrete geographic boundaries, with genetic similarities strongest between adjacent populations across ESU boundaries. These units are used as the basis for independent steelhead accounts. Recent genetic studies of summer and winter steelhead demonstrate greater levels of differentiation between spatially isolated reproductive populations (Papa et al. 2007, Pearse et al. 2007, Prince et al. 2015). For a full description of the ESU and DPS management units, see the NC winter steelhead account.

The Northern California summer steelhead is represented by a group of distinct populations (Distinct Population Segment, DPS, Box 1) that is well adapted to persisting in California's northern coastal mountains. The genetics of steelhead along the coast of California have been recently studied with microsatellite DNA, which reveals complex interactions with other coastal population segments and the legacy of hatchery-planted fishes (Bjorkstedt et al. 2005). In general, results indicated that geographically proximate populations were most similar genetically. The northernmost populations of NC summer steelhead show a genetic influence from Klamath Mountains Province steelhead, which are the next DPS to the north. This reflects both their transitional nature with more northern populations and possibly the transfer of hatchery juveniles from the Klamath Mountain Province and Central Coast steelhead DPSs in the 1980s.

Along the Lost Coast, collections of steelhead from the Eel, Mattole, and Bear rivers cluster together, while collections of steelhead along the Mendocino Coast show genetic connectivity among these smaller basins. This may indicate higher levels of dispersal among these numerous streams or be the legacy of past transfers of fish among these basins (Bjorkstedt et al. 2005). Genetic studies on steelhead from the Eel River found that winter and summer populations were more closely related to each other than they were to winter and summer populations from other rivers (Clemento 2006). Recent genetics information indicates that early-migrating steelhead, those that make up fall- and summer-run fish, have a genetic variation that has evolved separately in populations of both steelhead and spring-run Chinook salmon (Pearse et al. 2016). Over time, positive selection and straying likely have caused this favorable mutation to radiate outward and give rise to discrete runs of steelhead and salmon along the West Coast of the United States.

The distribution of non-anadromous individuals in the NC steelhead DPS is poorly documented. It is likely that these trout historically constituted only a small component of the overall population in most coastal basins, given the limited extent of historical barriers in most northern California watersheds. In larger basins where there are more opportunities in headwater areas for non-anadromous life histories to develop in isolation, rates of gene flow between resident and anadromous rainbow trout are likely low enough for the two forms to be considered separate populations (Bjorkstedt et al. 2005). Genetic analyses among juvenile trout in upper Middle Fork Eel River tributaries showed significant genetic differences indicating isolated, small, resident populations (Clemento 2006). Despite natural barriers, such as boulder fields and waterfalls, which segregate some Eel River tributary populations from anadromous adults in the mainstem in most years, headwater populations still retain migratory alleles in their genome, indicating their potential conservation value and potential to express an anadromous life history (Kelson et al. 2016).

Larger watersheds within the DPS also support summer run steelhead in Redwood Creek and the Mad, Mattole, Eel, and Van Duzen rivers. The Northern California steelhead DPS variations have been defined as: winter and summer, with a distinctive variant known as 'half-pounder' that may be derived from any of the three DPSs. NC summer steelhead are treated separately in another account because the two runs are distinctive in their genetic makeup, behavior, and reproductive biology and require different conservation frameworks than winter-run fish (Busby et al. 1996, Prince et al. 2015, Hodge et al. 2016). Genetic analyses support two discrete, separate monophyletic units of migrating populations based primarily on timing of freshwater entry and resulting maturation (Papa et al. 2007), correlating with run timing for the ocean-maturing (winter) and stream-maturing (summer, fall) ecotypes (Prince et al. 2015).

Life History: The continuum of life history strategies and migration timing of steelhead in California is covered in detail in the Klamath Mountains Province winter steelhead account. Summer steelhead are stream-maturing ecotype fish that enter freshwater with undeveloped gonads, and then mature over several months in freshwater. This life history is uncommon compared to ocean-maturing or winter-run fish. These steelhead oversummer in typically deep, bedrock holding pools and remote canyon reaches of streams with some overhead cover and subsurface flow to keep cool until higher flows arrive in winter (Busby et al. 1996).

NC summer steelhead enter estuaries and rivers as immature fish between April and June in the northern portion of the DPS (Redwood National Park 2001). In the Mad River, summer steelhead enter the mouth in early April through July as flows allow (M. Sparkman, CDFW, pers. comm. 2016). Mattole summer steelhead enter the river between March and June (Mattole Salmon Group 2016), and further migrations upstream occur from June on, but timing depends upon rainfall and consequent suitable stream discharge for passage into upper sections of watersheds. Spawning happens primarily in the winter between December and early April in headwater reaches of streams not utilized by winter steelhead (Roelofs 1983, Busby et al. 1997), though favorably wet conditions may lengthen the spawning period into May. Infrequent observations of steelhead spawning in June have also been reported on the Mattole River (Mattole Salmon Group 2016).

Unlike salmon, steelhead can spawn several times throughout their lives (iteroparity). Steelhead utilize this strategy to maximize reproductive success, spread survival risk over cohorts, and to buffer against short-term decline and catastrophic events such as wildfires, earthquakes, or landslides (Ricker 2016). Hopelain (1998) reported that repeat spawning varies considerably among runs and populations, from 18 to 64% of spawners. Females make up the majority of repeat spawners, and potentially the majority of spawners that are successful (Busby et al. 1996). In Freshwater Creek, between 10 and 26% of steelhead are repeat spawners, though the proportion of repeat spawners may be mostly indicative of a strong cohort of first time spawners (Ricker 2003). Females lay between 200 and 12,000 eggs (Moyle 2002). Outmigration of spawned adults can occur as late as June, but typically occurs no later than May in most watersheds (Busby et al. 1997). Shapovalov and Taft (1954) noted that hundreds of spawned-out adults often schooled above Benbow Dam on the South Fork Eel River. Additionally, in years with low spring outflows, steelhead may become stranded in their natal streams for the summer (S. Harris, CDFW, pers. comm. 2007).

Steelhead spawning occurs over a protracted period, and thus fry emergence may also take place over a long period, which influences young-of-the-year redistribution and potentially result in emigration into estuaries (Day 1996). Based on their occupancy of headwater streams with relatively low (< 50 CFS) winter flows (Roelofs 1983), newly emerged fry move out of these smaller natal streams into larger tributaries soon after emerging. Juvenile steelhead school together and seek shallow waters along riffle margins or pool edges, while older juveniles maintain territories in faster and deeper locations in pool and run habitats (M. Sparkman, CDFW, pers. comm. 2016). Where steelhead coexist with larger coho salmon juveniles, they prefer pool habitats for faster growth, although young-of-year steelhead can be competitively displaced to riffle habitats (Smith and Li 1983). Yearling steelhead occasionally emigrate from their natal rivers and recent studies have shown that some one-year-old smolts return as adults (M. Sparkman, CDFW, pers. comm. 2007). Typically successful juveniles rear in streams for two years.

Juvenile steelhead favor areas with cool, clear, fast-flowing riffles, ample riparian cover and undercut banks, and diverse and abundant invertebrate life (Moyle 2002). Growth rates vary with environmental conditions. NC steelhead in Redwood Creek can grow from 0.26 to 0.73 mm/day (M. Sparkman, CDFW, pers. comm. 2007). NC summer steelhead juveniles of all sizes move within their natal streams, and typically individuals leave during higher spring flows with movement peaking during late April or May. Young-of-year steelhead will emigrate to estuaries as late as June or July in Redwood Creek (M. Sparkman, CDFW, pers. comm. 2007). Very small emergent fry have been observed by divers in July and even August in the Mattole River, suggesting either a very late or very early spawning period (Mattole Salmon Group 2016). In Freshwater Creek, out-migrating steelhead averaged 156 mm FL, while the back-calculated ocean entry check for migrating spawners was at 194 mm FL, suggesting that additional rearing takes place in the estuary (Ricker 2003). Minimum growth in the estuary appears to occur when the river mouth is closing and a shift from estuarine to lagoon conditions occurs, typically between mid-August and mid-September (Cannata 1998). In the Mattole lagoon, juveniles display benthic feeding strategies. In the lower lagoon, they primarily eat amphipods (*Corophium* spp.), while in the upper lagoon they eat primarily caddisfly larvae (Zedonis 1990).

Smoltification (the physiological process of adapting to survive in saline ocean conditions) occurs in early spring. Scale studies suggest the majority of juvenile fish from the Middle Fork Eel River become smolts at two years old and return to freshwater at age 3 and 4 (Puckett 1975). Smolts typically emigrate from the river to the estuary or ocean between March and June, but prevailing habitat conditions may prevent exit from the estuary until late fall. A common process in small estuaries supporting NC summer steelhead is the formation of a summer lagoon, where beach sands form a bar across the mouth of the river. Strong salinity stratification in lagoons without sufficient inflow or very strong winds can lead to poor water quality, causing steelhead to seek refuge near the surface, in near-shore waters where more mixing occurs, or upstream beyond the seasonally stratified zone. Lagoon habitats offer juvenile steelhead flexibility in life history strategies to display a "double-smolting" strategy, whereby they enter an estuary or lagoon habitat for a short time before migrating back upstream to freshwater to grow and then before outmigrating to the lagoon and ocean once the lagoon breaches the following winter (FishBio 2016). This life history expression results in tradeoffs between carrying capacity, food availability, and likelihood of survival at different sizes between freshwater, brackish, and saltwater habitats available to them as juveniles. The prevalence and quality of these estuary and lagoon habitats are crucial for expression of the full spectrum of life history diversity of the DPS and the species, and are necessary for population resilience and recovery (FishBio 2016).

Some NC summer steelhead enter the ocean as they begin their third year of life after spending at least one year in the estuary (Cannata 1998). NC steelhead were captured in August during trawl surveys north and south of Cape Blanco (Brodeur, Fisher et al. 2004, Harding 2015), suggesting much of their time in the ocean is spent close to their natal streams. Steelhead grow rapidly at sea, feeding on fish, squid, and crustaceans in surface waters (Barnhart 1986, Harding 2015). Steelhead use their strong homing sense to return to natal areas where they were born to spawn (Moyle 2002). While California steelhead can spend several years in the ocean, many steelhead returning to Freshwater Creek, a small coastal tributary of Humboldt Bay, spend just two years in the sea (e.g., Ricker 2003). In coastal California basins, the most common life history patterns for first time spawners are 2/1 (years in fresh water/ocean), 2/2, and 1/2 (Busby et al. 1996). The majority of returning steelhead in the Mad River were three years old (Zuspan and Sparkman 2002; Sparkman 2003).

In Redwood Creek and the Mad, Eel, and Mattole Rivers, a small number of small, mostly immature "half pounder" steelhead (Snyder, 1925) are observed annually. In the Mad River, these smaller steelhead have been documented following the fall-run Chinook salmon as they migrate upstream (M. Sparkman, CDFW, pers. comm. 2016). Frequently, half-pounders outnumber adult steelhead during these surveys. However, the presence of half-pounders over-summering with adult summer steelhead is not typically characterized in the literature (Kesner and Barnhardt 1972, Hopelain 1998). While half-pounders are stream-maturing fish like summer steelhead, they are not traditionally considered to be part of summer steelhead life history because they do not mature or reproduce while in the river. (Lee 2016). The relative contribution of winter-run and summer-run steelhead to offspring that exhibit the half-pounder life history is unknown, but seems to be more common to summer-run fish (Lee, 2016). Half pounders along the Northern California coast are likely distinct from half pounder steelhead in the Klamath Mountain Province, which are reported to enter and leave the river as immature, subadult fish and spend up to 4 months at sea (Kesner and Barnhart 1972, Lee 2015, FishBio 2016). The NC steelhead half pounders are generally larger (25-35 cm FL or larger) than Klamath fish, but they are not well documented. High phenotypic plasticity in juvenile and adult life histories, demonstrated by NC summer steelhead, warrants further study. For a thorough discussion of the half-pounder life history, see the Klamath Mountains Province winter steelhead account.

Habitat Requirements: Steelhead require distinct habitats for each stage of life. The abundance of summer steelhead in a particular location is influenced by the quantity and quality of suitable coldwater habitat during low flow summer and fall months, food availability, and interactions with other species. Over-summering habitat for adult summer steelhead is critical for survival of this life history. In general, suitable habitats are often distributed farther inland than those for winter steelhead in the same watersheds (Moyle 2002).

Adult steelhead have a body form adapted for holding in faster water than most other salmonids with which they co-occur can tolerate. Within California, Bajjaliya et al. (2014) found important differences in steelhead morphology based on flow regimes and habitats occupied. Northern California steelhead had the largest individuals, on average, than populations of steelhead from elsewhere in the state. In general, coastal steelhead that occupied smaller, slower coastal rivers were deeper bodied, longer, and more robust than steelhead from larger inland rivers with higher velocities. Low flows associated with more inland rivers and tributaries do not facilitate passage of larger bodied adults, and therefore select for smaller, more streamlined fish. Adult summer steelhead require water depths of at least 18 cm for passage (Bjorn and Reiser 1991), however, this may not take into account the deep-bodied, robust physiology of coastal steelhead in the NC steelhead DPS, which would require slightly more flow to allow passage (Bajjaliya et al. 2014). Reiser and Peacock (1985 in Spence et al. 1996) reported the maximum leaping ability of adult steelhead to be 3.4 m. Hawkins and Quinn (1996) found that the critical swimming velocity for juvenile steelhead was 7.7 body lengths/sec compared to juvenile cutthroat trout that moved between 5.6 and 6.7 body lengths/sec. Adult steelhead swimming ability is hindered at water velocities above 3 m/sec (Reiser and Bjornn 1979). Preferred holding velocities are much slower, and range from 0.19 m/sec for juveniles and 0.28 m/sec for adults (Moyle and Baltz 1985). Physical structures such as boulders, large woody debris, and undercut banks create hydraulic heterogeneity that increases availability of preferred habitat in the form of cover from predators, visual separation of juvenile territories, and refuge during high flows.

Steelhead require cool water and holding habitat to withstand the higher temperatures and lower flows of summer and fall while they mature. Important factors influencing summer steelhead habitat use are pool size, low substrate embeddedness (< 35%), presence of riparian habitat shading, and instream cover associated with increased velocity through the occupied pools (Nakamoto 1994, Baigun 2003). Temperatures of 23-24°C can be lethal for the adults (Moyle 2002), which can limit abundance and spatial distribution. Subsurface, or hyporheic, flows can be important to providing cool, flowing water in habitats separated by thermal or other barriers. In August 2015 on the upper Middle Fork Eel River, adult summer steelhead were observed in pools of varying depth, but only with maximum temperatures of less than 23°C (CDFW 2015, Table 1). For a full description of steelhead thermal tolerances, see the Northern California winter steelhead account.

Table 1. Preferred summer steelhead temperatures, adapted from CDFW 2015, Table 3, pg. 9.

Pools with fish	**Mean (°C)**	**Max. (°C)**	**Min. (°C)**
Surface temp.	19.9	22.7	15.5
Bottom temp.	18.7	21.1	15.0
Pools without fish	**Mean**	**Max.**	**Min.**
Surface temp.	20.2	25.0	16.1
Bottom temp.	19.5	25.0	15.5

For spawning, adult steelhead require loose gravels at pool tails for optimal conditions for redd construction. Redds are usually built in water depths of 0.1 to 1.5 m where velocities are between 0.2 and 1.6 m/sec. Steelhead use a smaller substrate size than most other coastal California salmonids (0.6 to 12.7 cm diameter). Spawning habitat for summer steelhead can be variable, but their temporal and spatial isolation from other steelhead runs maintain low levels of genetic differentiation from winter steelhead in the same watershed (Barnhart 1986, Papa 2007, Prince et al. 2015). Summer steelhead can spawn in intermittent streams, from which the juveniles emigrate into perennial streams soon after hatching (Everest 1973). Roelofs (1983) suggested that use of small streams for spawning may reduce egg and juvenile mortality because embryos may be less susceptible to scouring by high flows and predation on juveniles by adults.

After spawning, adult steelhead, called "kelts" at this life stage, are capable of rapidly making their way back out to sea; the entire migration and spawning cycle of an adult fish can be completed in less than ten days (J. Fuller, NMFS, pers. comm. 2016). In contrast, in Redwood Creek, relatively large numbers of kelts migrate downstream through the lower watershed in March (M. Sparkman, CDFW, pers. comm. 2016). Due to the relatively short distances these fish must travel in small coastal watersheds to spawn, their survival rates and incidence of repeat spawning are higher than steelhead in the much larger Eel River, which reach dozens of kilometers inland.

Embryos incubate for 18 to 80 days, depending on water temperatures, which are optimal in the range of 5 to 13° C. Hatchery steelhead take 30 days to hatch at 11°C (Leitritz and Lewis, 1980 in McEwan and Jackson, 1996), and emergence from the gravel occurs after two to six weeks (Moyle 2002; McEwan and Jackson 1996). High levels of sedimentation (> 5% sand and silt) can reduce redd survival and emergence due to decreased permeability of the substrate and dissolved oxygen concentrations available for the incubating eggs (McEwan and Jackson 1996). When fine sediments (< 2.0 mm) compose > 26% of the total volume of substrate, poor embryo survival is observed (Barnhart 1986). Emerging fry can survive at a greater range of temperatures than embryos, but they have difficulty obtaining oxygen from the water at temperatures above 21.1°C (McEwan and Jackson 1996).

During the first couple years of freshwater residence, steelhead fry and parr require cool, clear, fast-flowing water (Moyle 2002). Exposure to higher temperatures increases the energetic costs of living for steelhead and can lead to reduced growth and increased mortality. As temperatures become stressful, juvenile steelhead will move into faster riffles to feed on more abundant prey (Moyle 2002 and bioenergetic box in SONCC coho account) and seek out cool-water refuges associated with cold-water tributary confluences and gravel seeps. In Redwood Creek, young-of-year (YOY) steelhead may travel 46 km downstream during summer months in search of rearing areas (M. Sparkman, CDFW, pers. comm. 2016). In the Mattole River, juvenile steelhead are found over-summering throughout the basin, although water temperatures often restrict their presence in the estuary. Cool water areas, including some restoration sites, provide refuge from temperatures that can rise above 19°C in the Mattole (Mattole Salmon Group 2005). However, juvenile steelhead can live in streams that regularly exceed 24°C for a few hours each day with high food availability and temperatures that drop to more favorable levels at night (Moyle 2002, M. Sparkman, CDFW, pers. comm. 2016).

Juvenile steelhead rear in the estuaries of Redwood and Freshwater creeks, Humboldt Bay, and the Eel, Navarro, Garcia, Gualala rivers. Lagoon habitats are critical for steelhead for rearing, feeding, and growth before and during smoltification (FishBio 2016). Estuary ecotones serve as important transitional habitat for both juvenile and adult salmonids, allowing feeding, resting, and acclimation to changing salinity before migration (Wallace et al. 2015). Juveniles that rear in ponds, sloughs, and other inundated estuary habitat grow more quickly than juveniles rearing in streams or tidally influenced freshwater habitats (CDFW and PSMFC 2014). As freshwater inflows decline during late spring, many of these estuaries become closed with sand bars, forming lagoons. Algal mats may then form, which reduce dissolved oxygen (DO) levels, eliminating much of this productive habitat from use by juvenile steelhead. Dissolved oxygen levels below 4.5 mg/L negatively affect juvenile steelhead trout (Barnhart 1986), although they can survive DO levels as low as 1.5-2.0 mg/L for short periods of time (Moyle 2002).

In saltwater, many age-0 California steelhead juveniles spend a year feeding in the cold California Current off the Klamath-Trinidad region, then move northwest to the North Pacific (Mantua et al. 2015, Hayes et al. 2016). Recent trawl surveys by NOAA Fisheries indicate that steelhead feed on pelagic organisms such as krill, fish, and amphipods in surface waters for several years in a narrow range of sea surface temperatures (apparently 8-14°C), then return to their natal rivers for spawning (Harding 2015, Hayes et al. 2016).

Distribution: Along the eastern Pacific, rainbow trout, including steelhead, are distributed from Southern California north to Alaska and west to Siberia (Sheppard 1972). In California, the NC summer steelhead DPS includes all naturally spawning populations of steelhead in California coastal river basins below upstream barriers to migration from Redwood Creek in the North, to the Mattole River in the South (NMFS 2016). While NC summer steelhead are present wherever streams are accessible to anadromous fishes and there are sufficient flows in the DPS geographic area, (J. Fuller, NMFS, pers. comm. 2016), summer steelhead are only present in a few select watersheds, including Redwood Creek and the Mad, Eel, and Mattole rivers. Summer steelhead were noted in local Humboldt County papers, including the Blue Lake Advocate, since at least 1908. Reports note large steelhead entering the North Fork Mad and Eel rivers in April and May (S. Van Kirk, 2013).

According to the National Marine Fisheries Service (NMFS), NC summer steelhead are divided into two geographic diversity strata: North Mountain Interior and Northern Coastal (Figure 1). Within these diversity strata, there are 10 essential independent populations that comprise the summer run portion of the Distinct Population Segment (DPS) based on environmental and ecological similarities (NMFS 2016). These populations include: Redwood Creek, Mad River, Van Duzen River, North, Middle, and South forks of the Eel River, Larabee Creek, Upper Middle and Upper Mainstem of the Eel River, and the Mattole River.

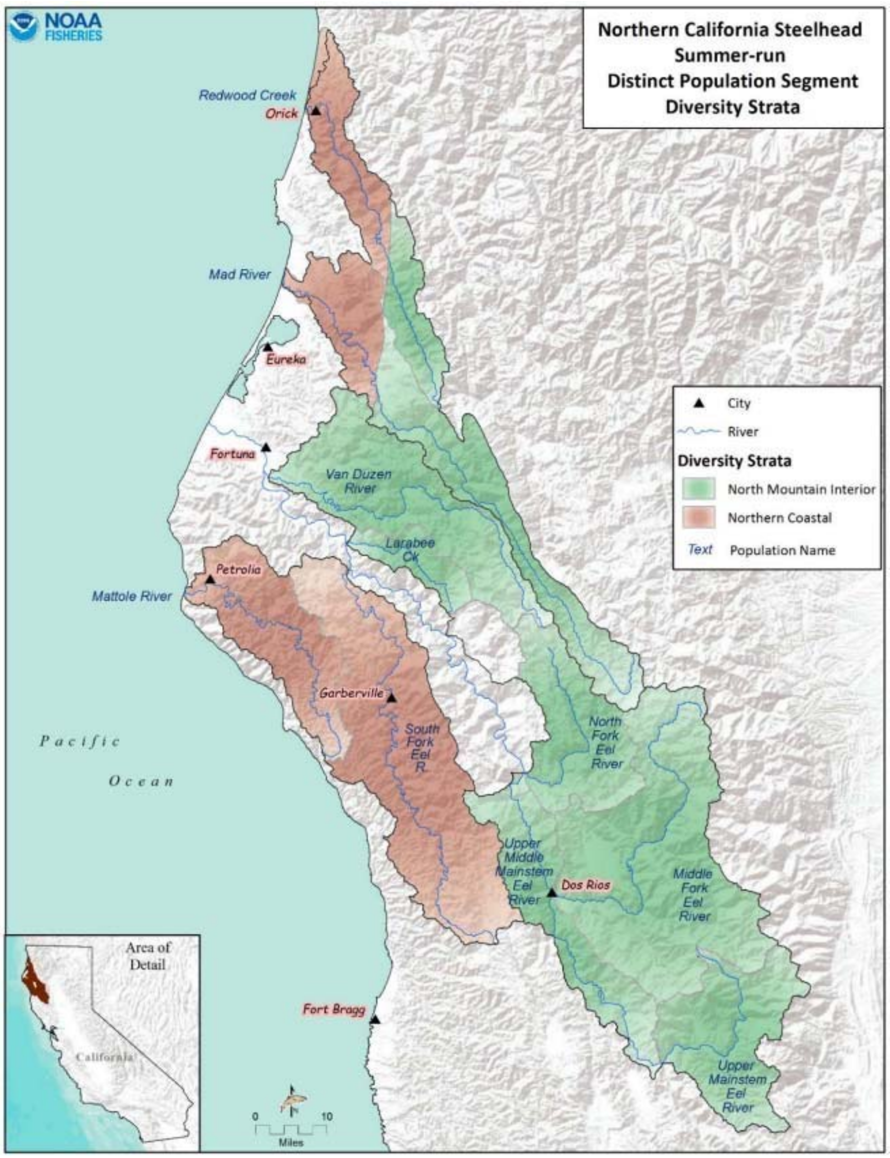


Figure 1. NC summer steelhead diversity strata. From: NMFS 2016, Fig. 2, pg. 4.

In the Mad River, upstream passage for summer steelhead is often blocked by a natural flow and velocity barrier near Humbug Creek (Mad River Alliance et al. 2016). An estimated 36% of potential habitat for summer steelhead over summering, spawning, and rearing is upstream of R.W. Matthews Dam and Ruth Lake. Flows from the dam provide cool water for summer-run adult fish and juveniles (NMFS 2016). However, a potential natural landslide barrier is about 48 river km downstream of the dam, so some flushing flows or restoration of the area may be required to access upstream available habitat.

On the Eel River, summer steelhead are found in the mainstem, upper mainstem, North, Middle, and South forks. Valuable habitat for summer steelhead on the Middle Fork Eel historically existed above Rio Dell, Fly Creek (Mendocino Co., CEMAR 2009). Estimates of the potential spawning and rearing habitat upstream of Scott Dam on the Eel River differ somewhat, but are significant (over 100 km) (Eel River Forum 2016). This population has never had hatchery influence, and was considered one of the most important summer steelhead populations in the state. A 1983 study found Fossil, Morrison, and Red Chert creeks to be important spawning tributaries to the North Fork of the Middle Fork Eel (DFW 1983 in CEMAR 2009). The natural boulder roughs (> 8-12% slope) at the Asa Bean crossing on the Middle Fork Eel River is a known impediment to passage of adult summer steelhead, forming a natural dam. The roughs in this area generally go dry by the end of July, causing adult and hundreds of juvenile steelhead to perish each year (CDFW 2015). Low baseflows during summer months are limiting to summer steelhead on the North Fork Eel and the Van Duzen River (NMFS 2016). Past CDFW reports have indicated that natural falls and the Eaton Roughs, approximately 67 km from the mouth of the Van Duzen, form an upstream barrier to migration, but that significant productive capacity exists could passage be enabled upstream (CEMAR 2009).

On the Mattole, summer steelhead can access the entire length of the river and two major tributaries, making these fish the southernmost population of summer steelhead in a watershed without significant snowmelt during spring and summer (Mattole Salmon Group 2016).

Trends in Abundance: Little historical abundance information exists for naturally spawning populations of NC summer steelhead, but current abundance of this species is likely much less than historical estimates. In general, summer steelhead abundance estimates come from volunteer-organized snorkel surveys in headwater reaches of rivers. These surveys are often difficult to make consistent in their scientific rigor or replicability due to issues with skill level, training, safety concerns, and access to similar reaches year after year. In addition, resident *O. mykiss* can often be mistaken for "half-pounder" subadult steelhead, which can cloud results and total counts. For example, snorkel surveys on the Mad River that have been occurring for decades must occasionally be altered due to issues of access associated with wildfires, changes in private land ownership, and illegal proliferation of marijuana (Mad River Alliance, 2015). However, redd surveys, angler surveys, carcass surveys, and DIDSON sonar and other methods have been used to provide rough estimates of returning adult spawners throughout the DPS. In addition, the Coastal Salmonid Monitoring Protocol has been in place for several years for the North Coastal Diversity Stratum (Redwood Creek, Mad River, Humboldt Bay tributaries, and Eel River) and has helped streamline data collection and synthesis (Williams et al. 2016).
The catastrophic flood of 1964-1965 caused nearly total loss of holding pools and created barriers to migration that have contributed to decline of summer steelhead populations in the Eel River watershed, as in Woodman Creek (CEMAR 2009). Low summer flows, sedimentation, high turbidity, and poor water quality associated with rural development, diversions, forestry

practices, and cattle grazing have significantly reduced summer steelhead holding habitat in the Eel River basin and contributed to their decline (CEMAR 2009).

Redwood Creek, in northern Humboldt County, historically had summer steelhead, but only numbering a few dozen fish per year from 1981-1991 due to poor water quality from land use patterns, diversions, and warm temperatures (Anderson 1961, Figure 2). A recent evaluation (Williams et al. 2016) suggests that Redwood Creek would need to support a population in the hundreds to aid recovery of the DPS. It is currently temperature-impaired under the Clean Water Act, and faces temperature threats to over summering fish (NMFS 2016).

Year	No. of Summer Steelhead	Survey Dates
1981	16	8/10 - 13
1982[a]	2	10/12 & 14
1983	5	8/22 - 25
1984	44+	8/08 - 10
1985	44+	8/20 - 22, 9/4
1986	19+	8/25 - 27
1987	14	7/14 - 16
1988	8	7/26 - 28
1989[b]	0	7/31, 8/01 - 02
1990	14	7/31, 8/01 - 03
1991	15	8/05 - 08
1992	5	8/03 - 06, 10
1993	2	8/02 - 05, 09
1994	5	8/01 - 04
1995	5	7/24 - 27
1996	1	8/05 - 08
1997	6	8/04 - 07
1998	4	7/27 - 30
1999	5	8/2-10
2000	3	8/1 - 09
2001	0	7/31, 8/01-02, 08

[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.

Figure 2. Redwood Creek summer steelhead estimates, 1981-2001. From National Parks Service 2001, Appendix 1.

In the Mad River, snorkel surveys and opportunistic redd surveys are used to provide trend data for summer steelhead populations (Mad River Alliance et al. 2016). From 1994 to 2002, divers observed a mean of 250 adult steelhead per year (Mad River Alliance 2016, Figure 3). In recent drought years, numbers have continued to decline and some upstream reaches were no longer accessible due to low flows. Divers also complained about headaches and rashes, symptoms often associated with exposure to toxic cyanobacteria that are known to multiply in warm waters on the Eel River (Power et al. 2015). If the sampling protocol in use since 2013 can be slightly amended, NOAA could assess viability of Mad River summer steelhead for recovery (CDFW 2016).

Table 1. Mad River summer-run steelhead dive survey results 1980 - 2015

		Adults			Half-Pounders		
Year	Miles Surveyed	Live	Dead	Total	Live	Dead	Total
1980[p]	17.9	0	0	0	0	0	0
1981[p]	17.5	2	0	2	0	0	0
1982[p]	32.4	167	0	167	0	0	0
1983[p]	22.8	31	0	31	0	0	0
1984[p]	14.1	111	0	111	0	0	0
1985[p]	14.8	52	0	52	0	0	0
1986[p]	7.8	10	0	10	0	0	0
1987[p]	20.2	18	0	18	0	0	0
1988[p]	10.6	60	0	60	0	0	0
1989[p]	10.6	20	0	20	0	0	0
1990[p]	10.6	33	0	33	0	0	0
1991[p]	14.7	59	0	59	0	0	0
1992[p]	10.6	34	0	34	0	0	0
1993[p]	10.6	48	0	48	0	0	0
1994[p]	51.6	305	0	305	166	0	166
1995[p]	66.6	541	1	542	10	0	10
1996[p]	60.7	427	1	428	19	0	19
1997[p]	66.6	292	5	297	12	0	12
1998[p]	57.0	191	0	191	20	0	20
1999[p]	46.4	82	0	82	15	0	15
2000[p]	53.5	170	0	170	62	0	62
2001[p]	12.5	194	0	194	583	0	583
2002[p]	19.7	185	0	185	80	0	80
2003[p]	18.7	483	0	483	5	0	5
2004[p]	5.8	209	0	209	9	0	9
2005[p]	5.6	211	0	211	10	0	10
2006	No Survey						
2007	No Survey						
2008[p]	5.1	110	0	110	20	0	20
2009	No Survey						
2010	No Survey						
2011	No Survey						
2012	No Survey						
2013	50.0	280	2	282	28	0	28
2014	61.0	322	0	322	92	0	92
2015	47.1	336	0	336	222	0	222

p = Provisional data

Figure 3. Mad River summer steelhead snorkel survey estimates, 1980-2015. From: Mad River Alliance 2016, pg. 19, Table 1.

The Eel River is the most important steelhead producing river in this DPS and once supported between 100,000 and 150,000 winter and summer steelhead, with the South and Middle forks combining to hold 70% of these spawning fish (NMFS 2016). Annual counts of steelhead in the Eel River were historically made at the Benbow Dam Fishway on the South Fork Eel River and at Van Arsdale Dam on the mainstem Eel River. The North Fork Eel River had populations of summer steelhead in the past, but that run has likely been extirpated (Higgins 1995). The Middle Fork Eel also had summer steelhead arriving as early as April 20th in some years and supported good numbers of fish (DFG 1959). It was once home to what was considered the largest run of summer steelhead left in the basin (DFG 1999). CDFW has conducted snorkel and electrofishing surveys on the Middle Fork since 1966, with survey data showing a downward trend in abundance and relatively low fluctuating numbers of fish over the last five decades (Figure 4). The majority of fish counted this past year were seen upstream of Fly Creek on the Upper Middle Fork Eel, with the majority of fish observed in pools between 3 and 6 meters in depth (S. Harris, CDFW, pers. comm. 2016). In addition, juvenile steelhead sampled near Osborn, fairly low in the Middle Fork Eel watershed, turned up abundance estimates about half of what they had been the previous two years in the area (CDFW 2015). The South Fork Eel River never supported large runs of summer steelhead, but occasionally reports of adult fish came out of this area (DFG 1992).

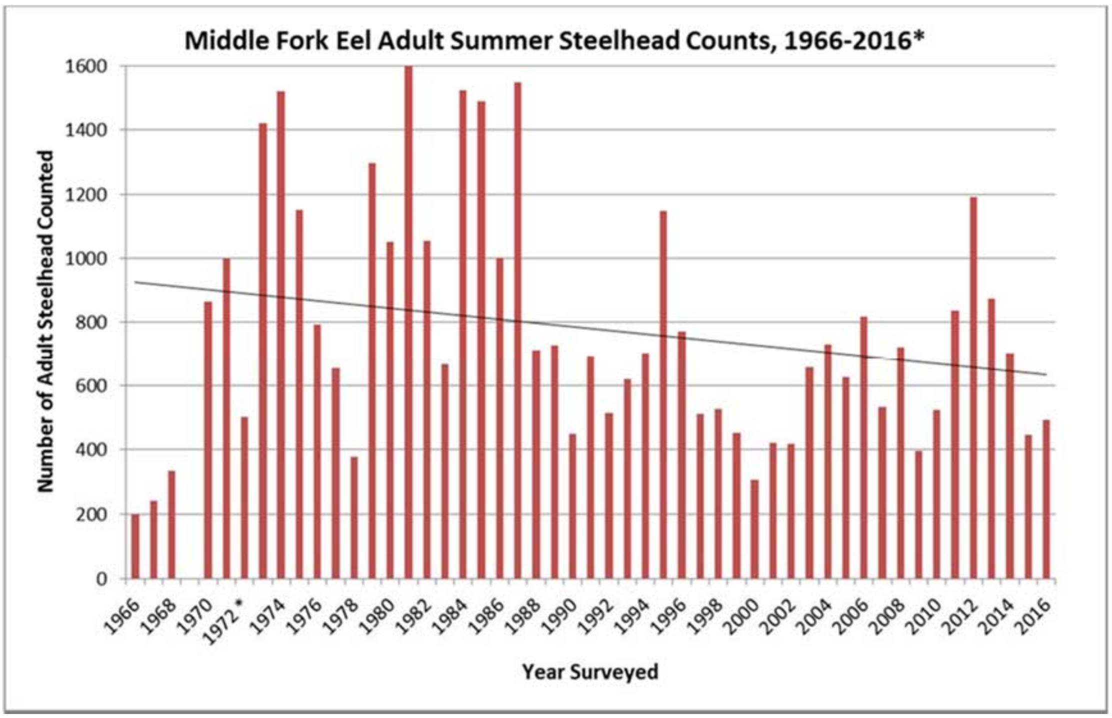


Figure 4. Middle Fork Eel River summer steelhead snorkel survey data, 1966-2016. Data from S. Harris, CDFW, pers. comm. 2016. *1972 surveys incomplete.

In the Van Duzen River, summer steelhead were likely never abundant, and were designated by the American Fisheries Society as having a high risk of extinction due to populations less than 200 adults per year (Higgins 1992). In 1995, a local biologist estimated that less than 100 adults were still present (Higgins 1995). The Little Van Duzen (South Fork Van Duzen) used to have around 100 summer steelhead adult per year in the early 1960s, but more recent surveys only found a single adult summer steelhead in 1997 (Preston 1997).
The Mattole River supports a small population of summer steelhead (Williams et al. 2016, Figure 5) at the southernmost extent of the DPS, and is not fed by snowmelt, making it even more susceptible to changes in flow and temperature (Mattole Salmon Group 2015). To sustain this population, CDFW and the Mattole Salmon Group captured and relocated as many coho and steelhead as possible from Baker Creek to Thompson Creek in August and September, 2014 as pools dried (Mattole Salmon Group 2014).

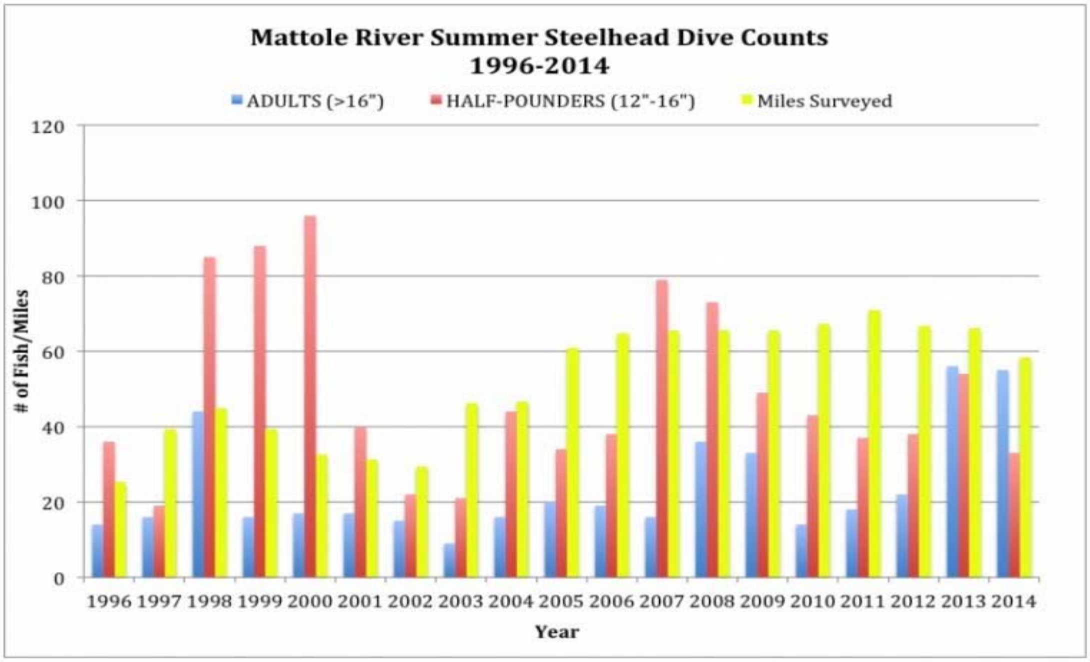


Figure 5. Mattole River summer steelhead snorkel surveys, 1996-2014. From: Mattole Salmon Group, 2015. Fig. 1, pg. 4.

Factors Affecting Status: Over the recent, historic drought (2012-2016), increasingly severe anthropogenic pressures have compounded naturally stressful conditions for steelhead in California (floods, drought, fires, poor ocean conditions, etc.), causing depleted populations to decline further. The Northern Diversity Stratum of summer steelhead lack adequate shelter, staging pools, gravel quantity, and sufficiently cool mainstem water temperatures. Reduced floodplain connectivity, low passage flows, physical barriers to migration, and low abundance are limiting recovery of the DPS (NMFS 2016). Expression of the full suite of steelhead life history diversity is at risk due to low population abundance, fishing pressure during summer months, poor water quantity and quality, and lack of complex over summering pool habitat (NMFS 2016).

Dams. Both the Eel and Mad rivers have dams that prevent access to considerable steelhead rearing and spawning habitat. Approximately 36% of potential steelhead habitat in the Mad River lies above Matthews Dam, while in the upper mainstem Eel River more than 99% of available spawning habitat upstream of Soda Creek is blocked by Scott Dam, exemplified by Gravelly Valley (NMFS 2016). Recent study (Cooper et al. *In progress*) suggests that potential steelhead habitat upstream of Scott Dam ranges from 291-463 km. In 1981, the Eel and its major tributary, the Van Duzen, were both designated as Wild and Scenic Rivers, protecting them from future dams, but not from diversions (Power et al. 2015). These barriers represent a major limiting factor on NC summer steelhead abundance: reduced gravel quantity and quality necessary for successful spawning and egg hatching in the Eel River (NMFW 2016). Blocked high gradient, small tributaries are important to summer steelhead, because these fish probably ascended higher in each watershed than any other salmonid based on their morphological adaptations to hold in lower, faster water and leap higher than other steelhead or Chinook salmon (Hodge et al. 2011). Scott Dam also reduces flows into the mainstem Eel River by delivering about 3% of its water into the neighboring Russian River watershed (NMFS 2016). This flow reduction negatively impacts mainstem water quality, especially during summer and fall, reduces stream complexity, and constricts the period of outmigration by juvenile steelhead during the spring and summer. Barrier inventories have been completed in NC summer steelhead counties, but most are still in place because considerable effort is required to eliminate even the priority barriers. Recent creation of the Eel River Action Plan may help prioritize and catalyze action to remove these barriers.

Flow releases in the reach between Scott Dam and Cape Horn Dam have improved summer temperatures. As a result, juvenile steelhead grow faster than those rearing in tributaries; some may reach over 19 cm in a single year of growth, a size which is suitable for smolting and migrating out to sea (SEC 1998). Unfortunately, the smolts leaving the interdam reach tend to migrate several weeks later than those from the tributaries, exposing them to less favorable conditions (higher temperatures, lower flows) than fish that migrate earlier (SEC 1998).

Logging. A significant proportion of the NC summer steelhead landscape is industrial timberlands, both private and public, which have already undergone intense logging in the 19th century. The cumulative, synergistic effects of these operations is difficult to grasp, though direct impacts to steelhead from logging include increased sedimentation and stream temperatures, reduced canopy cover, destruction of instream habitat, lack of large woody debris, and altered flow timing and volume. The channel of the Eel River and its tributaries have become shallower, braided, and less defined (Lisle 1982). These changes have reduced the ability of adults to reproduce, juveniles to forage, and migrants to safely pass to the ocean, as well as reducing productivity of aquatic invertebrates that are the principal food for fish. Excessive fine sediments from legacy impacts of logging still clog channels in the Eel today (Power et al. 2015).

Areas subjected to logging in many steelhead watersheds also suffer from increased effects of fire, a natural phenomenon in most coastal landscapes, especially outside the coastal fog belt. The history of timber management, combined with natural variability in conditions, create a complex potential fire regime (Noss et al. 2006), but in many areas both the frequency and intensity of fires has been increased by a long history of forest management focused on tree production. An additional problem has been "salvage logging" where large dead trees are removed after a fire, enhancing the erosion following a fire by increased road building and reducing availability of trees to fall into streams and create steelhead habitat.

Logging and its associated roads and legacy effects (see Coho Salmon accounts) have increased erosion on steep hillsides, greatly increasing sediment loads in the rivers. High sediment loads cause deep holding pools to fill with gravel, embed spawning gravels in fine materials, and create shallower runs and riffles. All this decreases the amount of usable habitat and increases the vulnerability of fish to poachers and predators. Such practices may also decrease summer flows, raising water temperatures to levels that may be stressful or even lethal to summer steelhead (Lewis et al. 2004).

Agriculture. Agricultural and cattle ranching land use practices in the DPS can negatively impact adjacent streams containing steelhead and other anadromous fish. The trampling and removal of riparian vegetation by grazing livestock destabilizes and denudes stream banks, increasing sediment and stream temperatures (Spence et al. 1996). These activities can reduce canopy over stream channels and increase siltation of pools necessary for juvenile rearing (Moyle 2002). Other impacts of agriculture include stream channelization, large woody debris removal, and armoring of banks to prevent flooding of fields (Spence et al. 1996). These types of activities remain "best management practices" for agriculture, vineyards, and ranching in some parts of the NC summer steelhead range. All of these activities, in combination with diversions for irrigation, degrade aquatic habitat quality, reducing its suitability for steelhead or other native fishes, while enhancing its suitability for non-native fishes, such as Sacramento pikeminnow (*Ptychocheilus grandis*) (Harvey, White et al. 2002).

These land uses have altered floodplain hydrology, decreased bank stability, increased sediment delivery and transport of pollutants. Within river channels, these activities disrupt substrate composition, divert flows, reduce water quality, and inhibit natural processes of temperature regulation. In addition, lagoon and estuary habitats often store excess sediments, have reduced habitat complexity, and are impaired by temperature increases.

In the past few decades, illegal water diversions and subsequent habitat degradation of remote headwater streams as part of the "Green Rush" for marijuana cultivation has become the limiting factor for salmonid survival in first- and second-order streams in the DPS. The unregulated pesticide, damming, and habitat destruction that typically accompany illegal grow operations pose a serious threat to the long term persistence of steelhead in the DPS (NMFS 2016). Specifically, Mendocino and Humboldt Counties, with their sparse, rural populations and heavily forested landscape, serve as epicenters of these illegal activities. A recent CDFW-funded study (Bauer et al. 2015) found that in the headwaters of Redwood Creek, home to a key independent population of NC summer steelhead, illegal and unregulated diversions for marijuana cultivation could consume over 20% of the available water in a year. These diversions have significant consequences on habitat quantity and quality for salmonids, such as elevated temperature and sediment, increased competition, predation and disease risks, increased stranding and delayed migration, lower growth rates, and reduced survival, and are likely occurring in remote streams throughout the DPS.

Water diversions and impoundments are threats rated as "very high" by NMFS because they lead to decreased flows and increased temperatures that are often lethal to over summering fish (NMFS 2016). Water temperature is a crucial limiting factor, and can be addressed with riparian vegetation restoration, increased large woody debris in streams, and fewer water diversions that allow more groundwater recharge within watersheds (NMFS 2016). Irrigated agriculture, subdivisions of parcels in rural areas, and illegal marijuana diversions all stress water supply, especially during drought. Estuary hydrodynamics become altered with less freshwater inflow, causing hypersaline conditions to prevail. A voluntary Sanctuary Forest water forbearance program is attempting to use water tanks in place of diversion during critical months to support oversummering fish.

Degraded water quality and supply have perhaps the greatest impact on adult summer steelhead in California. Many rivers and streams in the DPS are impacted by high temperatures, low dissolved oxygen, low flows, or all of the above during summer and early fall months due to diversions and land use practices. Summer discharge in the lower Eel is often insufficient to connect pools with surface or even hyporheic flow, leading to harmful cyanobacteria blooms that disrupt food chains and kill salmonids (Power et al. 2015). In 2013, a very poor water year led to the formation of the largest cyanobacteria algal bloom in the South Fork Eel River since 1988. In the large Eel watershed, summer temperatures peak between 20-22 degrees Celsius in headwaters, 26 degrees Celsius in upper mainstem tributaries, and reach 30 degrees Celsius at sites in the lower drainages that are not protected by cooling fog. High summer baseflows fueled by groundwater recharge are more likely to provide temperatures, flows, connectivity, and support a food web favorable for juvenile fish survival during summer (Grantham et al. 2012). Flows are especially crucial for sustaining adult summer steelhead until higher flows return in the fall and winter for spawning.

Estuarine alteration. According to NMFS (2016), estuary/lagoon quality and extent of available habitat is a limiting factor on NC summer steelhead. What suitable estuary habitat remains is subject to high turbidity, poor water quality from runoff and sedimentation, and onset of hypersaline conditions by mid-summer that cause juvenile steelhead to leave these valuable rearing and feeding habitats (CDFW 2014). The estuaries of the Eel and Mad rivers and Redwood Creek have been leveed, subjected to armoring with hard structures, drained, altered by tidegates, and converted to support agricultural and rural development, robbing juvenile salmonids of valuable habitat. The essential water purification and sediment delivery functions of the Eel delta, which delivers more sediment per watershed area than any river in America, have been significantly hampered by human development (Taylor 2015).

Recreation. While sport fishing regulations require a zero take for naturally produced NC steelhead, fishing for steelhead and "trout" continues in large portions of the two largest systems, the Mad and Eel rivers. Angling is allowed on the Mad River for ten months. The fishery is directed towards hatchery winter steelhead, which are marked, and supports an angler success rate that is normally higher than other North Coast rivers (Sparkman 2003). Natural steelhead populations in the Mad River are at very low levels, reflected in the low harvest of natural produced fish (Sparkman 2003). However, recent counts using sonar are showing increases in the returns of natural adult steelhead (M. Sparkman, CDFW, pers. comm. 2016). The mainstem Eel River and its forks support catch-and-release fisheries, which are monitored through the Steelhead Report Card Program. Summer steelhead are especially vulnerable to illegal fishing and poaching, as they are often constrained to a single pool that is disconnected from other usable habitat and enforcement is inadequate (NMFS 2016). While the bag limit for hatchery-reared steelhead is 2 fish per person per day, even catch and release fishing can have an impact at times when conditions are naturally stressful to wild steelhead (NFMS 2016).

Hatcheries. Mad River Hatchery releases about 150,000 winter steelhead smolts per year since 2009 (CDFW 2016 Mad River Hatchery Plan). Competition among winter adults of hatchery origin on natural origin summer steelhead remain unknown, but broodstock for the hatchery fish originate in the basin now to reduce genetic impacts to natural populations such as outbreeding depression. Large numbers of hatchery origin fish do spawn downstream of the Mad River Hatchery each year, which increases competition for redds with natural origin fish and redd superimposition (CDFW 2016). Hatchery steelhead have also been documented to displace a large percentage of wild steelhead in some streams (McMichael et al. 1999) and they may directly prey upon smaller young-of-year wild steelhead. Other risks from hatcheries include disease transmission, alterations of migration behavior in wild fish, and genetic changes that affect subsequent fitness in wild populations such as reduced fitness and productivity of natural stocks (Waples 1991). Effects on summer steelhead should be explored.

Mining. Instream mining, especially in the leveed reach of lower Redwood Creek, has become necessary to maintain flood control conveyance as a result of inadequate levee design and increased sediment delivery to the stream. Humboldt County has mined gravel from this reach every year from 2004-2010, causing simplification of the channel, removal of pools and aquatic vegetation, and important shelter habitat (NMFS 2016). This threat was rated "high" for the most recent status review update.

Alien species. Non-native species are present in NC summer steelhead watersheds, and invasion of the Eel River system by Sacramento pikeminnow pose a threat to juvenile steelhead (Brown and Moyle 1997). Pikeminnow prey directly on and displace juvenile steelhead, pushing them from pool habitat into less desirable riffle habitat, resulting in reduced growth and survival.

In addition, reduced habitat connectivity and low water levels may increase susceptibility of summer steelhead to diseases (Power 2015) and predation, especially from river otters (*Lontra canadensis*, M. Sparkman, CDFW, pers. comm. 2016).

Table 2. Major anthropogenic factors limiting, or potentially limiting, viability of Northern California summer steelhead populations. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result. Certainty of these judgments is moderate based on peer reviewed and gray literature, direct observation, expert judgment, and anecdotal information. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Major dams present on the Mad and Eel Rivers, which block access to significant important spawning and rearing habitat.
Agriculture	High	Conversion of estuarine wetlands to agricultural lands, diversions, influx of fertilizers and other pollutants into estuaries, especially for illegal marijuana cultivation.
Grazing	Medium	Some impacts in lowland areas, especially where estuary marshes have been converted to pasture.
Rural/ residential development	Medium	Effects localized, but increasingly an issue in Humboldt Bay tributaries.
Urbanization	Low	Increasingly an issue in the DPS.
Instream mining	Low	No known impact but occurs in some streams.
Mining	n/a	
Transportation	Medium	Roads are an ongoing source of sediment input, habitat fragmentation, and channel alteration.
Logging	Medium	Major activity, with dramatic historical impacts in many areas.
Fire	Low	Increased stream temperatures and sediment input may be a factor in some inland watersheds.
Estuarine alteration	High	Estuaries are vitally important rearing, feeding, and migrating habitat and have been significantly altered in most watersheds.
Recreation	Low	Mortality and sublethal impacts are likely through regulated, legal catch and release fishing.
Harvest	Low	Prohibited for all summer steelhead.
Hatcheries	Medium	Hybridization or competition with hatchery steelhead is possible but not well-studied recently.
Alien species	Low	Sacramento pikeminnow may play an increasing role in predation on juvenile salmonids in the Eel River watershed.

Effects of Climate Change: Climate change is a major threat to the continued persistence of NC summer steelhead. In general, climate change will impact the freshwater habitat of steelhead in several important ways:

1. Increased runoff and flooding, scouring redds
2. Higher stream temperatures reducing habitat quality and survival
3. Lower stream flows reducing habitat quantity and accessibility
4. Earlier spring snowmelt reducing juvenile outmigration success
5. Altered ocean circulation and productivity reducing sub-adult growth and survival in the marine environment (decrease in smolt to adult survival)
6. Higher stream temperatures and flows creating thermal and velocity migration barriers to juveniles and adults in both marine and freshwater
7. Increased frequency and intensity of catastrophic wildfires, threatening salmonid survival with attendant erosion, mass wasting, etc.
8. Altered woody debris availability and characteristics reducing holding areas for juvenile salmonids
9. Higher temperatures shifting range of suitable habitat northward in ocean and freshwater habitats
10. Increased eutrophication of estuaries that serve as important nurseries and foraging habitat for juvenile and sub-adult salmonids

To summarize the recent NMFS findings on climate-related impacts to NC steelhead, the primary concerns focus on altered streamflows and warmer temperatures, which reduce survival and passage through reductions in suitable holding, spawning, and rearing habitat. These impacts can reduce life history diversity, further stressing low populations of summer steelhead (NMFS 2016). NMFS considered summer-run steelhead in the DPS separately from winter-run fish, due to their increased susceptibility to redd scour due to timing of spawning and necessary holding in mainstem rivers during the warmest months of the year (NMFS 2016). Summer steelhead were found to be more vulnerable to these impacts than winter fish in "most (if not nearly all) cases" (NMFS 2016, Appendix B, pg. 21). Using a threat vulnerability analysis, NOAA Fisheries forecast that NC summer steelhead populations in the Redwood Creek, Van Duzen River, North and South Fork Eel, and Mattole are all highly susceptible to climate change impacts in the near future (NMFS 2016). These impacts are already being seen throughout the DPS range, and are limiting suitable upper watershed habitat for summer steelhead. Persistence of these populations is likely only with increased protection and restoration to improve stream flows, allow accessibility to prime holding and spawning habitat, and maintain cool temperatures in headwater tributaries for both spring Chinook salmon and summer steelhead.

Modeling of high greenhouse gas emissions scenarios have forecast increasing frequency and duration of critical drought, which exacerbates and compounds these impacts by reducing overall streamflow and increasing the variability in timing of precipitation events in California (NMFS 2016). As a result, Northern California summer steelhead may experience local extinctions and range contractions since higher gradient or elevation headwater streams are inaccessible behind falls, boulder fields, or dams in the DPS. Ongoing drought in California has likely contributed to a dip in populations of summer steelhead in the DPS, as lower flows and warmer summer water temperatures likely caused increased mortality before spawning. Persistent drought is likely to exacerbate already acute problems associated with depletion of summer baseflows, reduction of coldwater refugia, or even stream dewatering during the late summer and early fall months by reducing spawning, rearing, and migration habitat. More frequent and severe droughts are likely to contribute to higher occurrences of low summer baseflows that fuel toxic cyanobacteria blooms and degrade food webs that oversummering adult steelhead and juveniles depend on (Power et al. 2015). If summer temperatures increase during summer and early fall month and precipitation and prevalence of fog decrease, as has been observed in Northern California over the last fifty years, stream temperatures will rise and further stress summer-rearing salmonids and summer steelhead holding in pools (Madej 2011).

Drought and poor ocean conditions tied to climate change and El Nino conditions likely caused some decline in salmonid populations across the state by reducing coldwater upwelling and food availability (Daly et al. 2013, Williams et al. 2016). Changes in precipitation patterns could lead to flooding, contributing sediments from highly erodible terrain that smothers valuable gravel and fills in pool habitat. As populations continue to decline and become more fragmented, stochastic events such as increased catastrophic fire may change genetic structure, breeding, and population dynamics in ways that are unrecoverable.

Status Score = 1.9 out of 5.0. NC summer steelhead have a high risk of extinction in the next 50 years without significant restoration and intervention (Table 2). The entire DPS, which includes winter steelhead, was listed as threatened under the Federal Endangered Species Act on June 7, 2000 (NMFS 2000), a status that was reaffirmed on January 5, 2006 (NMFS 2006). They are considered a Sensitive Species by the U.S. Forest Service. This status could deteriorate rapidly if restoration and protection efforts are not put into effect. NC summer steelhead currently have no special conservation status within the state of California, but should be officially recognized as threatened under the California Endangered Species Act by the Fish and Game Commission or at least declared a state Species of Special Concern.

Table 3. Metrics to determine the status of Northern California summer steelhead, where 1 is poor value and 5 is excellent and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Much diminished from historical distribution.
Estimated adult abundance	2	Likely fewer than 1,000 adults across the DPS in a given year.
Intervention dependence	3	Require continuous monitoring and significant improvement of habitat and accessibility for recovery.
Environmental tolerance	2	Adults require coldwater refuges and pool habitat with cover that is free from human intervention.
Genetic risk	2	Spatial and temporal segregation between summer and winter fish make this life history susceptible to extinction.
Climate change	1	Highly vulnerable; temperatures and flows already marginal in many areas and summer steelhead require cold water in the warmest months to survive to spawn.
Anthropogenic threats	1	3 High and 5 Medium threats. Sufficient flows and temperatures are rapidly disappearing in the DPS.
Average	1.9	13/7.
Certainty	2-3	Actual numbers of fish poorly known.

Management Recommendations: Northern California summer steelhead are trending downward over time, and require significant action to recover from legacy impacts of road building, logging, forest fires, poor water quality, and disjointed land use throughout their range. Increasing rural development and illegal diversions and withdrawals for illegal marijuana cultivation throughout the DPS range, coupled with five years of ongoing historic drought, have significantly stressed summer steelhead populations and have driven their decline. Other threats across diversity strata include dearth of large woody debris and cover for rearing fish, abundance of roads and railroads adjacent to sensitive watersheds and associated sedimentation/erosion, illegal diversion and degradation, presence of barriers to migration, and lack of sufficient high quality spawning and rearing habitat due to uncoordinated land use practices (NMFS 2016).

To ameliorate these threats, the NMFS Coastal Multispecies Recovery Plan for the NC steelhead DPS lays out a full suite of necessary recovery actions and essential partners (NMFS 2016). CDFW is currently revising a steelhead restoration and management plan, which will help compile threats and identify specific actions to restore and manage steelhead in California (Nelson 2016). However, lack of coordination and prioritization of specific actions to protect summer-run life history steelhead in California represents a major challenge. Although designation of ESUs and DPSs are based upon distinctiveness of life-history traits and distinguishing genetic characteristics, such distinctions are not guiding conservation of steelhead life history diversity at the watershed scale, which is essential for maintaining populations of summer steelhead in the future.

CDFW and NMFS have been developing a statewide coastal salmonid monitoring program on the North Coast. Developing consistent monitoring protocols, with comprehensive abundance and trend data for all salmonids is essential for assessing the viability and recovery of NC summer steelhead. In addition, California has made significant investments in North Coast watersheds by matching federal funds from the Pacific Coast Salmon Recovery Fund to provide annual grants for restoration activities through the Fisheries Grant Restoration Program. Coordination with the State Coastal Conservancy grant programs has also leveraged funds to meet identified habitat restoration needs. To that end, a major timberlands owner along California's coast, Mendocino Redwood Company, is implementing its Habitat Conservation Plan (HCP), which will cover 6 high priority watersheds for NC summer steelhead, as well as Chinook and Coho Salmon (Williams et al. 2016). While these efforts are important, there is a need to better integrate HCPs with other watershed-based management actions and restoration activities across basins.

California's historic drought has sparked emergency drought actions to help stem declining trends in fish abundance. Law enforcement has cracked down on illegal diversions for marijuana cultivation, violations of fishing closures and poaching, and other activities that negatively impact salmonid populations, especially summer steelhead (J. Fuller, NMFS, pers. comm. 2016). Partnerships between NGOs, local landowners, and municipalities have increased utilization of real-time sensors and helped reduce water diversions voluntarily from juvenile rearing habitat during critical summer periods (Lehr 2016). In the future, California can develop innovative approaches to conservation by utilizing available legal provisions, such as AB 2121, to maintain instream flows to protect fisheries resources downstream of water diversions (Williams et al. 2016). In 2015, Governor Brown allocated over $35M in emergency funding for CDFW to monitor water quality, conduct baseline population and stressor monitoring, and carry out juvenile fish rescues across the state during the drought. CDFW teamed up with the Mattole Salmon Group to relocate juvenile steelhead and coho salmon from isolated pools on Baker Creek, near Whitethorn, (Humboldt Co.) to another tributary of the Mattole River with sufficient flow in 2014 and 2015 (Fisheries and Aquatics 2016). In 2015, over 200 juvenile SONCC Coho Salmon and 300 juvenile steelhead were successfully relocated (Fisheries and Aquatics 2016).

Emergency fishing closures under CDFW Code of Regulations Title 14 were enacted in the last three years throughout watersheds in the DPS range due to concerns over low flows and dissolved oxygen levels, high water temperatures, reduced fish passage, and increasing rates of disease and infection among already-stressed steelhead populations (Nelson 2016). Through the dedicated work of NOAA Fisheries, CDFW, the Native Fish Society, CDFW, and others, the low-flow closures for the Northern California steelhead DPS were amended by the California Fish and Game Commission to better reflect real-time changes in streamflow in the flashy systems most common in the geographic range. Now, the gages on the more representative Noyo and Navarro Rivers are used as triggers for low flow closures instead of utilizing the managed flows of the Russian River below Lake Sonoma (D. DeRoy, Native Fish Society, pers. comm. 2016). These actions have reduced poaching and targeted fishing over stressed individuals, but are not sufficient to allow these fish to recover or persist long term (J. Fuller, NMFS, pers. comm. 2016).

Perhaps the single greatest opportunity for increasing robust runs of wild salmon and steelhead in California remains large scale restoration and recovery of the Eel River through implementation of action items prioritized by the Eel River Forum. This ecosystem scale approach to managing salmonid and nonnative fish in the Eel River is essential to maintain the steelhead population in the tributaries and forks of this basin in the long term. The Eel River Forum, a partnership of resource users, landowners, managers, residents, and others, recently released its Eel River Action Plan, which will "coordinate and integrate conservation and recovery efforts in the Eel River watershed to conserve its ecological resilience, restore its native fish populations, and protect other watershed beneficial uses" (Eel River Forum 2016). Key action items identified in by the Forum include:

1. Evaluate flow releases from Potter Valley project
2. Prioritize block flow releases below the project to assist out-migrating salmon and steelhead in the spring
3. Evaluate extent of pikeminnow invasion and impacts on salmonids
4. Explore and document salmonid habitat upstream of Scott Dam
5. Determine water dynamics and quality for Lake Pillsbury
6. Evaluate potential use of PG&E lands for salmon and steelhead habitat restoration implementation
7. Understand past and future potential carrying capacity of the Eel watershed for salmonids and how proposed projects would impact it
8. Consider potential changes to operations during FERC relicensing of the Potter Valley Project, slated to begin in April 2017

Hatchery operation can also play a role in conservation of Northern California summer steelhead on the Mad River. Further monitoring, including implementation of the hatchery genetic management plan, should be undertaken to minimize the risks associated with the operation of hatcheries on naturally-produced NC summer steelhead. Implementation of the Mad River Hatchery Genetic Management Plan (2016) is essential for recovering wild winter steelhead in the DPS. Use of broodstock of Mad River basin fish and reduced numbers of fish should reduce interactions between natural origin fish and hatchery origin fish, giving wild steelhead a foothold to recover in the watershed.

Innovative, large-scale restoration activities that seek to regulate land use, manage sediment transport and input into streams, restore floodplain and estuary function as rearing habitat, and re-introduce large woody debris instream have occurred throughout the DPS range in the last several years. For example, a decades-long, multi-stakeholder project on the Salt River, in the Eel River estuary, continues to create valuable habitat for rearing fish on floodplain habitat purchased from a willing cattle rancher. Restoration of the stream-estuary ecotone of the Elk River, seasonal flooding of marginal agricultural lands, and construction of off-channel habitats on Salmon and Jacoby creeks, have created habitat that is used seasonally by over 17 species of fish (including federally threatened species like Coho Salmon and tidewater goby) and helps support a relatively intact native fish assemblage (Taylor 2015, Scheiff et al. 2016). In September 2013, over 100 whole large conifers and their intact root wads were planted strategically instream to provide scour pools, velocity refuge, and foraging and rearing cover for juvenile salmonids. By allowing large woody debris to interact with flows, the river itself can generate habitat complexity in the form of side-channels, scour pools, and meanders to provide rearing habitat for juveniles and resting areas for migrating adults (Mattole Salmon Group 2015). These types of cutting-edge projects are scaling from proof of concept to become replicable across watersheds, and are essential to recovery of NC summer steelhead. While such projects are important to test ideas and strategies, more coordinated and extensive restoration is required to bolster wild salmonid populations.

NORTHERN CALIFORNIA WINTER STEELHEAD
Oncorhynchus mykiss irideus

Moderate Concern. Status Score = 3.3 out of 5.0. Northern California (NC) winter steelhead are in long-term decline and face extirpation over much of their range.

Description: Steelhead are anadromous rainbow trout which return from the ocean as large silvery trout with numerous black spots radiating outward on their tail, adipose and dorsal fins. Their backs are an iridescent blue to nearly brown or olive. Their sides and belly appear silver, white, or yellowish, with an iridescent pink to red lateral band appearing when they are nearly mature and ready to spawn. The mouth is large, with the maxillary bone usually extending behind the eyes, which are above pinkish cheeks (opercula). Spawning males have a hooked lower jaw, called a kype. Teeth are well developed on the upper and lower jaws, although basibranchial teeth are absent. The dorsal fin has 10-12 rays; the anal fin, 8-12 rays; the pelvic fin, 9-10 rays; and the pectoral fins 11-17. The scales are small with 110-160 scales along the lateral line, 18-35 scale rows above the lateral line, and 14-29 scale rows below it (Moyle 2002).

The coloration of juveniles is similar to that of adults except they have 5-13 widely spaced, oval parr marks centered on the lateral line with interspaces wider than the parr marks. Juveniles also have spotting on the dorsal fin, unlike juvenile salmon. Juveniles possess 5-10 dark marks on the back between the head and dorsal fin, which make the fish appear mottled. There are few to no spots on the tail of juveniles and white to orange tips on the dorsal and anal fins. Resident adult trout may retain the color patterns of parr (Moyle 2002).

The various forms in California are nearly identical morphologically (Bajjaliya et al. 2014) although different populations may show some variation in average size of returning adults. They are distinguished by genetics, behavior, and life history strategies,

Taxonomic Relationships: Until the late 1980s, all steelhead were listed as *Salmo gairdneri gairdneri*. However, Smith and Stearley (1989) showed that steelhead are closely related to Pacific salmon (genus *Oncorhynchus*) and are conspecific with Asiatic steelhead, then called *Salmo mykiss*. As a result, the rainbow trout, including steelhead, is officially recognized by the American Fisheries Society as *Oncorhynchus mykiss*. All steelhead and nonmigratory coastal rainbow trout are usually lumped together as *O. m. gairdneri* or, more recently, as *O. m. irideus* (Behnke 1992).

Moyle (2002) discusses the complex systematics of California populations of steelhead. The six Distinct Population Segments (DPSs, Box 1) recognized by NMFS for California have more or less discrete geographic boundaries, with genetic similarities strongest between adjacent populations across DPS boundaries. The DPSs are: California Central Valley, Central California Coast, Klamath Mountains Province, Northern California, South Central California, and Southern California. Within each DPS, NMFS has identified Diversity Strata based on ecological similarities and life history differences between winter- and summer-run steelhead. These units are used as the basis for naming independent steelhead accounts in this report.

The NC winter-run steelhead is a well-supported, easily identifiable group of populations (Box 1) that is well adapted to conditions in California's northern coastal mountains. The genetics of steelhead along the coast of California have been recently studied with microsatellite DNA, which reveals complex interactions with other coastal population segments and the legacy of hatchery-planted fishes (Bjorkstedt et al. 2005). The northernmost populations of NC steelhead show some genetic influence from Klamath Mountains Province steelhead, which are the next DPS to the north. Genetically, fish along this portion of the coast, including the Mad River and Humboldt Bay tributaries, do not cluster tightly with NC steelhead populations from the Eel River or more southerly steelhead watersheds (Bjorkstedt et al. 2005). This reflects both their transitional nature with more northern populations and possibly the transfer of hatchery juveniles from the Klamath Mountain Province and Central Coast steelhead DPSs in the 1980s. Some NC winter steelhead populations in the Mad River and Redwood Creek cluster with steelhead populations from other NC steelhead streams, which either reflects ecotypes adapted to local conditions or the intra-DPS transfer of NC steelhead from different origins between basins (Busby et al 1996).

Within the Eel River, Clemento (2006) detected significant genetic differences between winter steelhead from the Middle Fork Eel River and those from the South Fork Eel River, Lawrence Creek (Van Duzen River tributary), and Willits Creek (upper Eel River tributary). Along the Lost Coast, from about Fort Bragg to Mendocino (Mendocino County), collections of steelhead from the Eel, Mattole, and Bear rivers cluster together, while collections of steelhead along the Mendocino Coast show genetic connectivity among these smaller basins. This may indicate higher levels of dispersal among these numerous streams or be the legacy of past transfers of fish among these basins (Bjorkstedt et al. 2005).

The distribution of non-anadromous individuals in the NC steelhead DPS is unknown. It is likely that these trout historically constituted only a small component of the overall population in most coastal basins, given the limited extent of historical barriers in most northern California watersheds. In larger basins where there are more opportunities in headwater areas for non-anadromous life histories to develop in isolation, rates of gene flow between resident and anadromous rainbow trout are likely low enough for the two forms to be considered to be separate populations (Bjorkstedt et al. 2005). Genetic analyses among juvenile trout in upper Middle Fork Eel River tributaries showed significant genetic differences indicating isolated, small, resident populations (Clemento 2006). Natural barriers such as boulder fields and waterfalls segregate some resident populations on Eel River tributaries such as Elder Creek from anadromous adults in most years. Headwater populations above barriers retain migratory alleles in their genome, indicating their potential to express an anadromous life history and retain conservation value (Kelson et al. 2016).

Larger watersheds within the DPS also support summer run steelhead in Redwood Creek and the Mad, Mattole, Eel, and Van Duzen rivers. Genetic data support the hypothesis that winter and summer steelhead populations are somewhat distinct (Prince et al. 2015). The NC summer steelhead are treated separately in another account because the two runs are distinctive in their genetic makeup, behavior, and reproductive biology and require different conservation frameworks than winter-run fish.

Box 1: NMFS, ESUs and DPSs

The Endangered Species Act of the 1973 (ESA) has an expansive definition of species, that includes distinct population segments, which was not further defined. This resulted in The National Marine Fisheries Service (NMFS) developing the concept of the Evolutionary Significant Unit (ESU) for salmon. An ESU was basically a cluster of populations that had a common evolutionary history and trajectory. While this worked to protect purely anadromous fishes, it did not work for anadromous rainbow trout, which often were one genetic population that included resident trout populations. So, NMFS and US Fish and Wildlife Service (USFWS) created the Distinct Population Segment (DPS) management concept for steelhead, in order to list anadromous forms of coastal rainbow trout while not listing resident forms. In practice, this has allowed sympatric, interbreeding resident rainbow trout and steelhead in the same stream to be treated as different DPSs. This is despite the official DPS Policy, which states that a group of organisms form a distinct population segment if it is "markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, and behavioral factors." [61 *Federal Register* 4722 (Feb. 7, 1996)].

Because NMFS has jurisdiction over anadromous fishes, while USFWS has jurisdiction over resident fishes, this concept allows resident and anadromous forms to be separated on the basis of who is allowed to manage them. This additional flexibility in applying the ESA was intended to benefit the steelhead life history in coastal rainbow trout populations, because upstream resident populations were often in good shape while steelhead were in decline. In many instances the distinction does not matter much either because resident trout are infrequent or because isolated resident trout populations are also threatened with extinction (e.g. southern steelhead).

Life History: In general, rainbow trout, which include steelhead, exhibit the largest geographic range and most complex suite of traits of any salmonid species. Steelhead and resident rainbow trout in many rivers are part of a single, complex gene pool, which contributes to the ability of coastal rainbow trout to adapt to systems that are highly unpredictable, subject to large variations in streamflow, and undergo frequent disturbance. The continuum of life history strategies and migration timing of steelhead in California is covered in detail in the Klamath Mountains Province winter-run steelhead account. In general, steelhead are rainbow trout that rear in streams for 1-3 years before turning into smolts and migrating out to sea. They remain in the ocean for varying lengths of time, where they feed on large crustaceans and fish. Spawning adult steelhead typically spend at least one year in the ocean and some may repeat spawning 2-4 times.

NC steelhead enter estuaries and rivers between September and March (Busby et al. 1997). Further migrations upstream occur as late as June, but timing depends upon rainfall and consequent suitable stream discharge for passage into upper sections of watersheds. Shapovalov and Taft (1954) reported steelhead entering the Eel River estuary as early as August, migrating upstream on increasing stream flows, but not moving during peak flows. Spawning happens primarily in the winter between December and early April (Busby et al. 1997), though favorably wet conditions may lengthen the spawning period into May. These steelhead arrive at spawning areas in reproductive condition (ocean- maturing ecotype). Because steelhead spawning occurs over a protracted period, fry emergence may also take place over a long period, which influences young-of-the-year redistribution and potentially result in emigration into estuaries (Day 1996, M. Sparkman, CDFW, pers. comm. 2016).

Unlike Pacific salmon, steelhead can spawn more than once (iteroparity). Steelhead utilize iteroparity to maximize reproductive success, spread survival risk over cohorts, and to buffer against short-term decline and catastrophic events such as wildfires, earthquakes, or landslides (Ricker 2016). Hopelain (1998) reported that repeat spawning varies considerably among runs and populations, from 18 to 64% of spawners, depending upon watershed and habitat conditions. Females likely make up the majority of repeat spawners (J. Fuller, NMFS, pers. comm. 2017), and potentially the majority of spawners that are successful (Busby et al. 1996). In Freshwater Creek, between 10 and 26% of steelhead are repeat spawners, though the proportion of so-called kelts may be higher in other short, coastal watersheds (Ricker 2003). For example, in relatively small coastal watersheds in Mendocino County, such as the Garcia River, most post-spawn females exhibit very good body condition, suggesting a high likelihood of repeat spawning potential (J. Fuller, NMFS, pers. comm. 2017). Females lay between 200 and 12,000 eggs (Moyle 2002) depending on size, age, and body condition. Outmigration of spawned adults can occur as late as June, but typically occurs no later than May in most watersheds and March marking the peak of outmigration on the Mad River (Busby et al. 1997, M. Sparkman, CDFW, pers. comm. 2016). Shapovalov and Taft (1954) noted that hundreds of spawned-out adults often schooled above Benbow Dam on the South Fork Eel River. Additionally, in years with low spring outflows, steelhead may become stranded in their natal streams for the summer (e.g., Noyo, Navarro Rivers; S. Harris, CDFW, pers. comm. 2007).

Newly emerged steelhead school together and seek shallow waters along riffle margins or pool edges, while older juveniles maintain territories in faster and deeper locations in pool and run habitats. Where steelhead coexist with larger coho salmon juveniles, they prefer pool habitats for faster growth, although young-of-year steelhead can be competitively displaced to riffle habitats (Smith and Li 1983). Yearling steelhead occasionally emigrate from their natal rivers to estuaries for rearing (M. Sparkman, CDFW, pers. comm. 2016), although more typically, NC steelhead juveniles rear in streams for two years.

Juvenile steelhead favor areas with cool, clear, fast-flowing riffles, ample riparian cover and undercut banks, and diverse and abundant invertebrate life (Moyle 2002). Growth rates vary with environmental conditions. NC steelhead grow from 0.24 to 0.37 mm/day instream in the Navarro and Mattole rivers, respectively, though estuarine growth rates are likely much higher (Zedonis 1990; Cannata 1998, J. Fuller, NMFS, pers. comm. 2017). In Redwood Creek, growth rates were greater, ranging from 0.26 to 0.73 mm/day (M. Sparkman, CDFW, pers. comm. 2007). NC steelhead juveniles of all sizes can show some movement in their streams and typically individuals leave during higher spring flows with movement peaking during late April or May. Young-of-year steelhead will emigrate further downstream in a system or to estuaries as late as June or July (M. Sparkman, CDFW, pers. comm. 2016). In Freshwater Creek, out-migrating steelhead averaged 156 mm FL, while the back-calculated ocean entry check for migrating spawners was at 194 mm FL, suggesting that additional rearing takes place in the estuary (Ricker 2003). In the Navarro River, it appeared that a greater proportion of age 2+ juveniles reside in the estuary than in the river; however, it is possible that these observations are confounded by sub-yearling fish that grew quickly to a comparable size due to abundance of food and favorable growth conditions in the estuary (J. Fuller, NMFS, pers. comm. 2017).

Minimum growth in the estuary appears to occur when the river mouth is closing and a shift from estuarine to lagoon conditions occurs, typically between mid-August and mid-September (Cannata 1998). In the Mattole lagoon, juveniles display benthic feeding strategies. Within the lower lagoon, they primarily eat amphipods (*Corophium* spp.), while in the upper lagoon they eat primarily caddisfly larvae (Zedonis 1990).

Smoltification (the physiological process of adapting to survive in saline ocean conditions) occurs in early spring and smolts typically emigrate from the river to the estuary or ocean between March and June. However, conditions may prevent exit from the estuary until late fall. A common process in small estuaries supporting NC steelhead is formation of a summer lagoon when beach sands form a bar across the mouth of the river. Strong salinity stratification in lagoons without sufficient inflow or very strong winds can lead to poor water quality (see discussion in Habitat). Steelhead then seek refuge near the surface, in near-shore waters where more mixing occurs, or upstream beyond the seasonally stratified zone. In the Navarro River, some NC steelhead enter the ocean as they begin their third year of life after spending at least one year in the estuary (Cannata 1998). Prior to formation of the bar across the mouth of the Navarro River, larger juvenile steelhead were observed in the estuary close to the ocean where water temperatures were cooler and salinities were higher. Following creation of the bar, these fish moved back into the upper lagoon.

Where lagoon habitats exist, steelhead juveniles can display a "double-smolting" strategy, whereby they enter an estuary or lagoon for a short time before migrating back upstream to fresh water. The following winter, they move back to the lagoon and then to the ocean once the lagoon breaches (FishBio 2016). This life history expression results in tradeoffs among carrying capacity, food availability, and likelihood of survival at different sizes in freshwater, brackish, and saltwater habitats. Lagoon habitats allow expression of the full spectrum of life history diversity of the DPS (FishBio 2016).

California steelhead can spend up to four years in the ocean, though many steelhead returning to Freshwater Creek, a small coastal tributary of Humboldt Bay, spend just two years at sea (e.g., Ricker 2003). In coastal California basins, the most common life history patterns for first time spawners are 2/1 (years in fresh water/ocean), 2/2, and 1/2 (Busby et al. 1996). The majority of returning steelhead in the Mad River were three years old (Zuspan and Sparkman 2002; Sparkman 2003).

NC steelhead were captured in August during trawl surveys north and south of Cape Blanco (Brodeur, Fisher et al. 2004, Harding 2015), suggesting much of their time in the ocean is spent close to their natal streams. Steelhead grow rapidly at sea, feeding on fish, squid, and crustaceans in surface waters (Barnhart 1986, Harding 2015). Steelhead use their strong homing sense to return to natal areas where they were born to spawn (Moyle 2002).

In the Klamath River, Redwood Creek and the Mad, Eel, and Mattole Rivers, a small number of "half pounder" steelhead (Snyder 1925) are observed annually. For a thorough discussion of the half-pounder life history, see the Klamath Mountains Province winter steelhead account. Half pounders in the Northern California Coast are likely distinct from the half pounder steelhead in the Klamath Mountain Province, which are reported to enter and leave the river as immature, sub-adult fish to spend up to about 4 months at sea (Kesner and Barnhart 1972, Lee 2015, FishBio 2016). The NC steelhead half pounders are generally larger (25-35 cm FL or more) than Klamath fish and have similar ocean foraging habitat off of the mouth of the Klamath River, which is constrained by a narrow band of sea surface temperatures (Hayes et al. 2016b). However, recent advances in sonar technology (such as DIDSON acoustic sonar technology) allow for enumerating half-pounders, as in the sonar studies in Mad River and Redwood Creek (M. Sparkman, CDFW, pers. comm. 2016). The high phenotypic plasticity in juvenile and adult life histories demonstrated by NC steelhead suggest the 'half pounders' may represent small reproductive fish, large resident fish, or a mixture of life history strategies. It is possible half-pounders result from juveniles that fortuitously migrated to sea at time when there were favorable sea surface temperatures (less than 14°C) in a small region just off the mouth of the Klamath River (Hayes et al. 2016), creating conditions for rapid growth.

Habitat Requirements: Steelhead require distinct habitats for each stage of life. The abundance of steelhead in a particular location is influenced by the quantity and quality of suitable habitat, food availability, and interactions with other species. In general, suitable habitats are often distributed farther inland than those of Chinook and coho salmon, as well as in smaller headwater streams (Moyle 2002). Adult steelhead require high flows with water at least 18 cm deep for passage (Bjorn and Reiser 1991). However, this depth may not take into account the deep-bodies of coastal NC steelhead, which would require slightly more flow to allow passage (Bajjaliya et al. 2014). Reiser and Peacock (1985 in Spence et al. 1996) reported the maximum leaping ability of adult steelhead to be 3.4 m. Temperatures of 23-24°C can be lethal for adults (Moyle 2002), although migrating winter steelhead usually do not encounter these conditions (Table 1). For spawning, steelhead in the DPS are well adapted to extremely flashy systems at the southern extent of the range with some years resulting in small migration windows due to timing of sandbar breaching at the mouths of lagoons (J. Fuller, NMFS, pers. comm. 2016). Once in fresh water, adult steelhead require loose gravels at pool tails for optimal conditions for redd construction. Redds are usually built in water depths of 0.1 to 1.5 m where velocities are between 0.2 and 1.6 m/sec. Steelhead use a smaller substrate size than most other coastal California salmonids (0.6 to 12.7 cm diameter), however, redd capping studies in Prairie Creek have shown that steelhead may spawn and superimpose redds of the larger Chinook salmon (M. Sparkman, CDFW, pers. comm. 2016). After spawning in relatively short coastal systems (e.g., Gualala and Garcia rivers and Russian Gulch) female adult steelhead often rapidly make their way back out to sea; achieving an entire migration and spawning cycle less than ten days if flow conditions allow (J. Fuller, NMFS, pers. comm. 2016). However, male spawning steelhead are likely to reside in freshwater longer, maximizing their spawning opportunities. Due to the relatively short migration distances that these fish travel up small coastal watersheds, as flow conditions allow, their survival rates and incidence of repeat spawning are much higher than steelhead in the much larger Russian or Eel rivers. During the drought, adult steelhead were observed waiting for sufficient flows to access spawning tributaries, which left them in reduced states of fitness for spawning (J. Fuller, NMFS, pers. comm. 2016).

Table 1. Steelhead thermal tolerances in freshwater from Richter and Kolmes (2005), McEwan and Jackson (1996), and Moyle (2002). All lethal temperature data is presented as incipient upper lethal temperatures (IULT), which is a better indicator of natural conditions because experimental designs use a slower rate of change (ca. 1°C/day). Fish living in the wild experience temperatures that fluctuate on a daily basis and rarely stay in warmer water for long. This is a synthesis of data from throughout the range of steelhead, so may not precisely reflect the tolerances of NC steelhead. Values may vary according to acclimation history and strain of trout.

	Sub-Optimal	Optimal	Sub-Optimal	Lethal	Notes
Adult Migration	< 10ºC	10-20°C	20-23°C	> 23-24°C	> 21ºC: Migration usually stops. 22-24ºC: Lethal temperature under most conditions; fish moving at higher temperatures are stressed and searching for cool refuges.
Adult Holding	< 10ºC	10-15°C	16-25°C	> 26-27°C	Summer steelhead survive the highest holding temperatures. Frequent high temperatures reduce egg viability.
Adult Spawning	< 4ºC	4-11°C	12-19°C	> 19°C	Egg viability may be reduced at higher temperatures.
Egg Incubation	< 4°C	5-11°C	12-17°C	> 17°C	This is the most temperature sensitive phase of life cycle.
Juvenile Rearing	< 10°C	10-17°C	18-26°C	> 26°C	Past exposure (acclimation temperature) has a large effect on thermal tolerance. Fish with high acclimation temperatures may survive 27ºC for short periods of time. Optimal conditions occur under fluctuating temperatures, with cooler temperatures at night. Heat-shock proteins (sign of stress) are produced at 17°C.
Smoltification	< 7ºC	7-15°C	15-24°C	> 24°C	Smolts may survive and grow at suboptimal temperatures but have a harder time avoiding predators or avoiding infection or disease.

Steelhead embryos incubate for 18 to 80 days depending on water temperatures, which are optimal in the range of 5 to 13° C. Hatchery steelhead take 30 days to hatch at 11°C (Leitritz and Lewis 1980 in McEwan and Jackson, 1996), and emergence from the gravel occurs after two to six weeks (Moyle 2002; McEwan and Jackson 1996). High levels of sedimentation (> 5% sand and silt) can reduce redd survival and emergence due to decreased permeability of the substrate and dissolved oxygen concentrations available for the incubating eggs (McEwan and Jackson 1996). When fine sediments (< 2.0 mm) compose > 26% of the total volume of substrate, poor embryo survival is observed (Barnhart 1986). Out of the gravel, emerging fry can survive at a greater range of temperatures than embryos, but they have difficulty obtaining oxygen from the water at temperatures above 21.1°C (McEwan and Jackson 1996).

During the first couple years of freshwater residence, steelhead fry and parr require cool, clear, fast-flowing water (Moyle 2002). Exposure to higher temperatures increases the energetic costs of living and can lead to reduced growth and increased mortality. As temperatures become stressful, juvenile steelhead will move into faster riffles to feed on increased prey (Smith and Li 1983 and bioenergetic box in SONCC coho account) and seek out cool-water refuges associated with cold-water tributary confluences and gravel seeps. Optimal temperatures for growth are estimated to be around 10-17°C (Table 1). As part of the North Coast Regional Water Quality Control Board's Mattole River Total Maximum Daily Load (TMDL) requirements, temperature thresholds were established for steelhead, such that temperatures less than 17°C were "good", 17°-19°C were "marginal" and higher than 19°C were "unsuitable/poor" (Coates, Hobson et al. 2002). In the Mattole River, juvenile steelhead are found over-summering throughout the basin, although water temperatures often restrict their presence in the estuary. Cool water areas, including some restoration sites, provide refuge from temperatures that can rise above 19°C in the Mattole (Group 2005). However, juvenile steelhead can live in streams that regularly exceed 24°C for a few hours each day with high food availability and temperatures that drop to more favorable levels at night (Moyle 2002, M. Sparkman, CDFW, pers. comm. 2016).

Steelhead have a body form adapted for holding in faster water than most other salmonids with which they co-occur. Thus, Hawkins and Quinn (1996) found that the critical swimming velocity for juvenile steelhead was 7.7 body lengths/sec compared to juvenile cutthroat trout that moved between 5.6 and 6.7 body lengths/sec. Adult steelhead swimming ability is hindered at water velocities above 3 m/sec (Reiser and Bjornn 1979). Preferred holding velocities are much slower, and range from 0.19 m/sec for juveniles and 0.28 m/sec for adults (Moyle and Baltz 1985). Physical structures such as boulders, large woody debris, and undercut banks create hydraulic heterogeneity that increases habitat available for steelhead in the form of cover from predators, visual separation of juvenile territories, and refuge during high flows.

Within California, Bajjaliya et al. (2014) found important differences in steelhead morphology based on flow regimes and habitats occupied. NC steelhead had larger individuals, on average, than populations of steelhead from elsewhere in the state. In general, coastal steelhead that occupied smaller, slower coastal rivers were deeper bodied, longer, and more robust than steelhead from larger inland rivers with higher velocities. It is hypothesized that low flows associated with more inland rivers and tributaries make passage of larger bodied adults more difficult, and therefore select for smaller, more streamlined fish.

Juvenile steelhead rear in the estuaries of Redwood and Freshwater creeks, Humboldt Bay, and the Eel, Navarro, Garcia, Gualala rivers; and most likely any estuaries that offer adequate rearing habitat in the state. Lagoon and estuary habitats are critical for steelhead for rearing, feeding, and growth after smoltification begins in upstream rearing habitats, triggering downstream migrations (FishBio 2016, J. Fuller, NMFS, pers. comm. 2017). Estuary ecotones may serve as important transitional habitat for both juvenile and adult salmonids, allowing greater feeding opportunities, and resting and acclimation habitat to changing salinity, if needed, before fish undertake migrations (Wallace et al. 2015). In addition, juveniles that rear in ponds, sloughs, and other inundated estuary habitat grow more quickly than juveniles rearing in streams or tidally influenced freshwater habitats (CDFW and PSMFC 2014). As freshwater inflows decline during late spring, many of these estuaries become closed with sand bars, forming lagoons. Algal mats may then form, which reduce dissolved oxygen (DO) levels, eliminating much of this productive habitat from use by juvenile steelhead. Dissolved oxygen levels below 4.5 mg/L negatively affect juvenile steelhead trout (Barnhart 1986), although they can survive DO levels as low as 1.5-2.0 mg/L for short periods of time (Moyle 2002).

In saltwater, many California steelhead juveniles spend a few months feeding in the California Current off the Klamath-Trinidad region, then move northwest to cooler waters offshore in the North Pacific to feed for one or more years (Mantua et al. 2015, Hayes et al. 2016a, Hayes et al. 2016b). Recent trawl surveys by NMFS indicate that steelhead feed on pelagic organisms such as krill, fish, and amphipods in surface waters during their time at sea

(Hayes et al. 2016). After feeding for several years in a narrow range of sea surface temperatures (apparently 8-14°C), they return to their natal rivers for spawning (Harding 2015, Hayes et al. 2016). During their ocean period they are not found in the same areas as the much more abundant species of salmon, so are rarely caught in commercial fisheries (Harding 2015). Most catches of steelhead in research trawls are in the upper 30m of the water column, suggesting their distribution is more closely tied to sea surface temperatures than other salmonids in the ocean (J. Nelson, CDFW, pers. comm. 2016, Hayes et al. 2016b).

Distribution: Along the eastern Pacific, rainbow trout are distributed from Southern California north to Alaska and range west to Siberia (Sheppard 1972). In California, the NC winter steelhead DPS includes all naturally spawning populations in California coastal river basins below upstream barriers to migration from Redwood Creek (Humboldt Co.) to just south of the Gualala River (Mendocino Co.) (NMFS 2016). This distribution includes the Eel River, the third largest watershed in California, with its four forks (North, Middle, South, and Van Duzen) and their extensive tributaries. Scott Dam, on the mainstem Eel River, blocks migration to Gravelly Valley, some of the best historical spawning habitat in the drainage (Shapovalov 1939), which is approximately 19km upstream of the historical Van Arsdale Fish Station. Recent research suggests that Scott Dam blocks approximately 291-463 km of potential steelhead migration, spawning, and rearing habitat (Cooper et al. *In progress*). On the Mad River, Matthews Dam serves as the upstream limit to steelhead, with only about 13km of Pilot Creek accessible for spawning (Stillwater Sciences 2010). With few exceptions, NC steelhead are present wherever streams are accessible to anadromous fishes and there are sufficient flows and cool water to complete their life history (J. Fuller, NMFS, pers. comm. 2017). Water bodies with no recent direct access to the ocean, such as Big and Stone lagoons between Redwood Creek and Little River, contain steelhead, although the source of these fish is unknown (M. Sparkman, CDFW, pers. comm. 2016). According to NMFS (2016) NC winter steelhead are divided into five geographic diversity strata (Figure 1). Within these five diversity strata, there are 19 essential independent populations, and 22 potentially independent populations that comprise the DPS.

Figure 1: NC Steelhead Winter-Run Essential and Supporting Populations

Figure 1. NC steelhead diversity strata from NMFS 2016, Fig. 1, pg. 3.

Trends in Abundance: Little historical abundance information exists for naturally spawning populations of NC steelhead, but current abundance of this species is quite low relative to historical estimates (Figures 2, 3, Table 2).

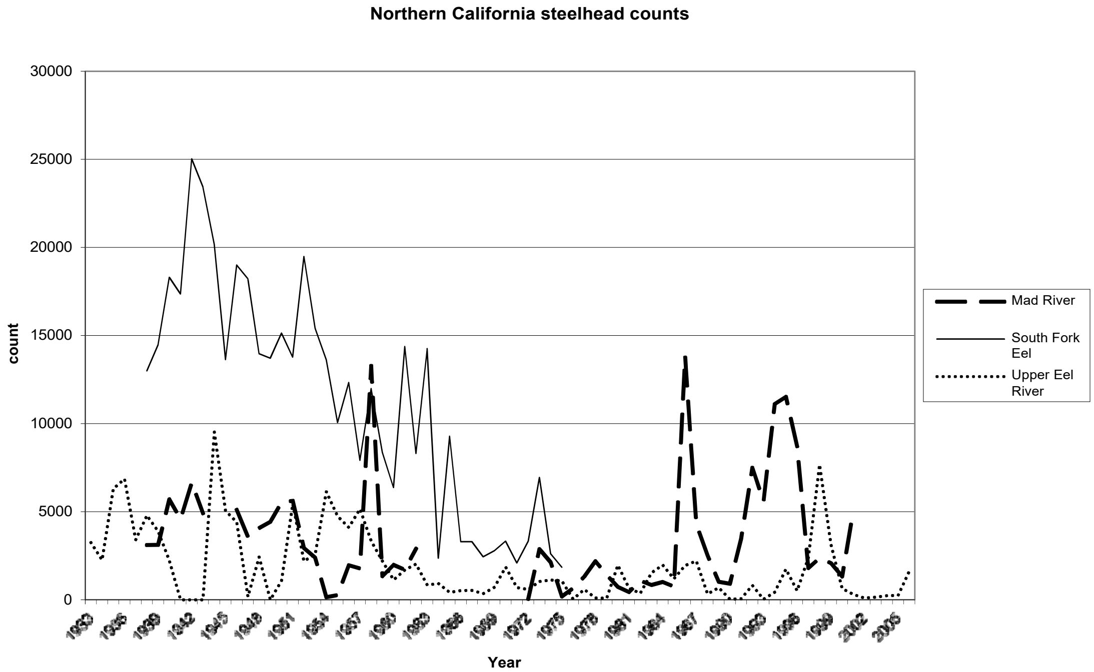


Figure 2. Northern California steelhead DPS counts from two locations in the Mad River (Sweasey Dam (pre-1964) and Mad River Hatchery (1972+), South Fork Eel River (Benbow Dam), and Upper Eel River (Van Arsdale Station). Data from Taylor 1978, California Department of Fish and Game, and Grass 2007.

Table 2. Number of adult salmonids, including steelhead, estimated to return the Mad River Hatchery 1972-2104 from CDFW 2016, Table 9, pg. 42.

Season	Coho	CHIN	Steelhead	Season	Coho	CHIN	Steelhead
1971/1972	337	323	42	1993/1994	39	11	5,591
1972/1973	466	1,036	52	1994/1995	74	67	11,118
1973/1974	327	495	2,872	1995/1996	12	56	11,520
1974/1975	160	231	2,138	1996/1997	259	64	8,713
1975/1976	2,103	278	190	1997/1998	40	7	1,807
1976/1977	1,193	661	658	1998/1999	13	40	2,364
1977/1978	648	250	1,317	1999/2000	20	50	3,085
1978/1979	577	246	2,190	2000/2001	17	11	1,399 (11)
1979/1980	352	145	1,411	2001/2002	13	52	5,893 (238)
1980/1981	503	86	730	2002/2003	9	11	4,519 (54)
1981/1982	135	251	442	2003/2004	No trapping		
1982/1983	622	900	1,087	2004/2005	0	1	1,880 (15)
1983/1984	87	437	838	2005/2006	0	1	1,671 (19)
1984/1985	24	82	1,015	2006/2007	0	0	1,528 (12)
1985/1986	45	275	753	2007/2008	1	0	3,005 (1)
1986/1987	324	299	13,833	2008/2009	0	0	305 (2)
1987/1988	953	846	4,303	2009/2010	0	0	2,441 (5)
1988/1989	845	242	2,529	2010/2011	0	0	4,846 (70)
1989/1990	256	46	1,027	2011/2012	0	0	3,948 (133)
1990/1991	92	1	915	2012/2013	0	0	3,118 (21)
1991/1992	37	10	3,463	2012/2013	0	0	3,192 (22)
1992/1993	67	27	7,497	2013/2014	0	0	1,841 (19)

Note: 1) 1999 BY to present production is AD-marked; ( ) represent number of non-AD-marked steelhead; HOR steelhead counts include 2 yr. old steelhead trout (jacks or jills). 2) The annual reports starting in 1971 do not record the number of broodstock released spawned or unspawned. From the 2004/2005 season to 2013 the Chinook and coho that enetered the fish ladder and trap were all released unharmed back to the river. Beginning in 2014, the number released and any mortalities will be recorded and reported in the annual report.

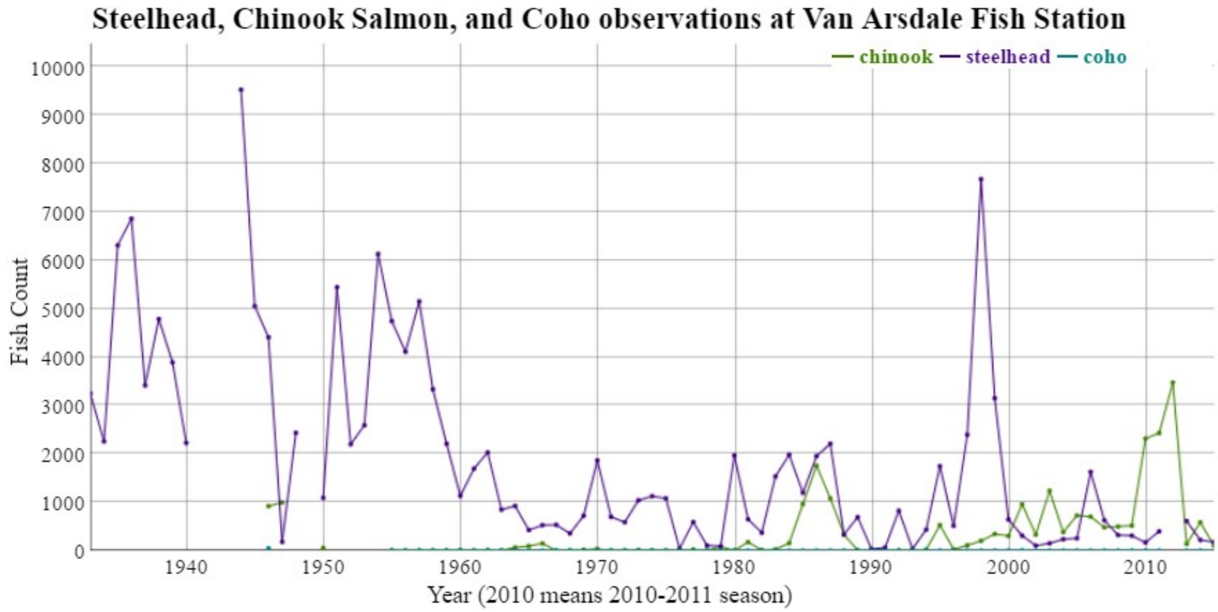


Figure 3. Counts of Steelhead, Chinook salmon and coho salmon at Van Arsdale Fish Station, 1930-2015. Data from Potter Valley Irrigation District, 2016. The 174 winter steelhead counted in 2015 represent one of the lowest totals since recordkeeping began in 1933.

Sound estimates of NC steelhead numbers are lacking because high, turbid flows during migration and spawning periods make safe, reliable sampling impractical. However, redd surveys, angler surveys, carcass surveys, and DIDSON sonar and other methods have been used to provide rough estimates of returning adult spawners throughout the DPS. In addition, the Coastal Salmonid Monitoring Protocol has been in place for several years for the North Coastal Diversity Stratum (Redwood Creek, Mad River, Humboldt Bay tributaries, and Eel River) and has helped streamline data collection and synthesis (Williams et al. 2016). Overall, populations of NC steelhead are currently estimated to be between 5-13% of viability targets (Williams et al. 2016).

Redwood Creek, in Northern Humboldt County, has produced several thousand wild adult steelhead each year. In contrast to trends observed in other basins, at the height of the drought in 2015, 1+ steelhead and 2+ steelhead abundance was recorded to be 47% higher and 105% higher, respectively compared to the averages of 38,000 and 8,900 from 2004-2014 (Sparkman, 2016). Data collected over decades from screw traps indicate that juvenile steelhead also had the greatest observed fork lengths on record in this period, indicating they had no problem finding food and suitable rearing habitat despite near record low flows throughout the basin. Redwood Creek had 150 redds over the last 4 years, while Prairie Creek, a major northern tributary, had about 40 observed spawners over the last 14 years. These redd counts are minimum estimates because they do not cover the entire duration of the spawning window. More recent sonar counts put mid-to-upper Redwood Creek estimates at 1,000 adults in 2014 (M. Sparkman, CDFW, pers. comm. 2016). Humboldt Bay tributaries excluding Freshwater Creek had about 88 redds, while Freshwater Creek itself had about 170 spawning fish over the last 15 years (Williams et al. 2016).

In the Mad River, CDFG (1965) estimated that about 6,000 steelhead spawned annually (Table 2). The Mad River Hatchery trapped an average of 1,160 fish annually from 1971 through

1980; 2,674 in 1981-1990; and 5,648 fish in 1991-2000. Since 2000, the number of steelhead returning to Mad River Hatchery has declined because operation of the hatchery was reduced due to funding shortages and genetic concerns over historical out-of-basin fish planting from the hatchery. In 2000-2001, Zuspan and Sparkman (2002) estimated approximately 17,000 steelhead spawned above Mad River Hatchery, with wild fish comprising only 8.3% (1,419) of the total. In 2013/14, an estimated 4,336 hatchery-origin steelhead and 3,449 natural-origin steelhead returned to Mad River, based on DIDSON sonar and rigorous, species apportionment methods (M. Sparkman, CDFW, pers. comm. 2016).

The Eel River is the most important steelhead producing river in this DPS and may have once supported 100,000-150,000 winter and summer steelhead, with the South and Middle forks together holding about 70% of the spawning fish (NMFS 2016). A time series of data analyzed in Good et al. (2005) estimated that the overall trend in adult returns was downward. Annual counts of steelhead in the Eel River were historically made at the Benbow Dam Fishway on the South Fork Eel River and at Van Arsdale Dam on the mainstem Eel River (Taylor 1978), which both show long-term declines in abundance (Figure 2). Recent estimates of winter steelhead in the South Fork Eel River hover around 19,000 adults per year (NMFS 2016). Over the last 16 years, about 630 adult steelhead were counted at the Fish Station each year, with natural origin fish accounting for half, which is well below viability targets. While the long-term trend is negative, a recent trend for natural-origin fish has been positive (Williams et al. 2016). Between 1991 and 1995, the annual mean number of juvenile steelhead per square meter in Van Duzen basin streams ranged from 0.27 to 0.98 fish (Hopelain et al. 1997). Over the past five years, spawner counts averaged 132 fish on the Van Duzen River.

The Mattole, Big, Navarro, and Gualala Rivers were estimated to each contain at least 12,000 spawning steelhead in 1963 (CDFG 1965), while Ten Mile, Noyo, and Garcia Rivers each contained at least 4,000 steelhead. The Garcia River in particular remains open to the ocean year-round, and as a result is home to a few Chinook and coho salmon as well as a fairly solid steelhead population (J. Fuller, NMFS, pers. comm. 2016). During 2003-2006, redd surveys on the Mendocino's Casper Creek, Little River, and Noyo River indicated escapement of steelhead was between 16 and 18 spawners annually (S. Harris, CDFW, pers. comm. 2007). Annual mean number of juvenile steelhead per square meter ranged from 0.18 to 1.88 in the Upper North Fork Mattole River (Hopelain, Flosi et al. 1997). Densities of steelhead were reasonably equivalent in Mendocino's Pudding and Casper Creeks, where they were present at 0.12 to 1.03 fish/m^2 (S. Harris, CDFW, per comm. 2007). Over the last two years, the Mattole River had only 298 returning adults counted (Williams et al. 2016). Big River had the largest estimates of any of these more central North Coast watersheds, with an average of 633 fish over the last 6 years and a positive abundance trend (Williams et al. 2016). In the Gualala River, the southernmost watershed in the DPS, retired fisheries biologist R. DeHaven (2011) estimated average annual adult steelhead abundance from 2001-2010 to be 1,735 in the Wheatfield Fork, with no discernable trends, albeit with differing amounts of sampling effort in some years due to challenging environmental conditions for sampling. The Albion, Navarro, and Gualala rivers showed positive abundance trends over the last 6-year period (Williams et al. 2016).

Abundance estimates of any kind for the small coastal watersheds in Sonoma and Mendocino counties are difficult to obtain, but CDFW has collected some information on presence and absence of steelhead over the last decade that support the idea that adult steelhead opportunistically utilize small watersheds for spawning over time (Table 3).

Table 3. Juvenile steelhead presence/absence data for select Mendocino and Sonoma coastal watersheds, 2005-2009. Data from Dan Logan, NMFS, 2016.

	2005		2006		2007		2008		2009	
Stream	**Date**	*# of Steelhead*	**Date**	*# of Steelhead*	**Date**	*# of Steelhead*	**Date**	*# of Steelhead*	**Date**	*# of Steelhead*
Fuller Creek	22-Oct-05	33	25-Oct-06	32	9-Nov-07	36	24-Oct-08	33	27-Nov-09	25
Miller Creek	22-Oct-05	26	25-Oct-06	28	9-Nov-07	26	24-Oct-08	26	27-Nov-09	21
Kohlmer Gulch	18-Nov-05	27	Did not sample	N/A	16-Nov-07	30	7-Nov-08	22	18-Dec-09	0
Fort Ross Creek	18-Nov-05	27	Did not sample	N/A	16-Nov-07	41	7-Nov-08	25	18-Dec-09	17
Mill Creek	Did not sample	N/A	Did not sample	N/A	30-Nov-07	0	Did not sample	N/A	Did not sample	N/A
Russian Gulch	Did not sample	N/A	Did not sample	N/A	30-Nov-07	58	20-Nov-08	30	21-Nov-09	32
Willow Creek	22-Nov-05	12	Did not sample	N/A	24-Nov-07	1	Did not sample	N/A	Did not sample	N/A
Jenner Gulch	Did not sample	N/A	Did not sample	N/A	Did not sample	N/A	20-Nov-08	26	21-Nov-09	27

Overall, CDFG (1965) suggested that close to 200,000 NC steelhead once spawned in the region's rivers combined. Optimistically, annual spawning returns today are likely less than 15,000-20,000 fish. However, data sets that allow long term trends to be determined quantitatively are lacking. It is clear, however, that current numbers are much lower than they were historically.

Factors Affecting Status: Steelhead populations are affected by both natural and human factors, but when increasingly severe anthropogenic pressures are added to naturally stressful conditions (floods, droughts, fires, poor ocean conditions, etc.) the result is severe decline. More general factors (e.g., freshwater and estuarine habitat degradation, water diversions, and gravel extraction) are discussed in accounts for Central California Coast steelhead and Central California Coast Chinook and Coho salmon.

Dams. Both the Eel and Mad rivers have dams that prevent access to considerable steelhead habitat in their basins. Approximately 36% of potential steelhead habitat in the Mad River lies above Matthews Dam, while in the upper mainstem Eel River several hundred kilometers of high quality, cold tributaries are blocked by Scott Dam, perhaps best exemplified by the historical Gravelly Valley (NMFS 2016, Cooper et al. *In progress*). While these represent significant habitat constrictions, culverts and bridges are barriers to steelhead passage in numerous smaller watersheds across the NC winter steelhead DPS. These barriers constitute a major limiting factor on steelhead abundance: large reductions in gravel quantity and quality necessary for successful spawning and egg hatching in the Eel River (NMFW 2016).

Another significant problem associated with Scott Dam is the timing of flow reductions into the mainstem Eel River, due to storage and delivery of the water to the Russian River via the Potter Valley Project. While pumping only takes about 3% of the available water (NMFS 2016) for transfer among the watersheds, the net effect of the storage and diversion is increased summer flows in the Russian River, with a muting in the recession of springtime flood-emulating flows in the Eel River (J. Fuller, NMFS, pers. comm. 2017). This flow reduction has the potential to both positively and negatively impact mainstem water quality during summer and fall, reducing stream complexity, and constricting the period of outmigration by juvenile steelhead during the spring and summer, although summer habitat may be improved in the reach between Scott Dam and the Cape Horn Dam. For example, if coldwater pool stored behind the dams is exhausted, then rearing conditions downstream will degrade and provide conditions more favorable to warmwater species (J. Fuller, NMFS, pers. comm. 2017). Even in this reach, it is not certain if the higher flows and colder temperatures help steelhead populations. Barrier inventories have been completed across the NC steelhead range, but most are still in place because considerable effort is required to eliminate even high-priority barriers (NMFS 2016).

Flow releases in the reach between Scott Dam and Cape Horn Dam have improved summer flows and temperatures. As a result, juvenile steelhead grow faster than those rearing in tributaries; some may reach over 19 cm in a single year of growth, a size which is suitable for smolting and migrating out to sea (SEC 1998). Unfortunately, the smolts leaving the interdam reach tend to migrate downstream several weeks later than those from the tributaries, exposing them to less favorable conditions (higher temperatures, lower flows) than fish that migrate earlier (SEC 1998).

Logging. A significant proportion of the NC winter steelhead landscape is industrial timberlands, both private and public, which have already undergone intense logging in the 19th century. The cumulative, synergistic effects of these operations is difficult to grasp, though direct impacts to steelhead from logging include increased sedimentation and stream temperatures, reduced canopy cover, destruction of instream habitat, and altered flow timing and volume. The channel of the Eel River and its tributaries have become shallower, braided, and less defined over time (Lisle 1982). These changes in the aquatic ecosystem have reduced the ability of adults to reproduce, juveniles to forage, and migrants to safely pass to the ocean, as well as having indirect effects such as reducing productivity of aquatic invertebrates that are the principal food for the fish.

Areas subjected to logging in many steelhead watersheds also suffer from increased effects of fire, a natural phenomenon in most coastal landscapes, especially outside the fog belt. The history of timber management combined with natural variability in conditions create a complex mosaic of potential fire regimes (Noss et al. 2006), but in many areas both the frequency and intensity of fires has been increased by a long history of inadequate forest management focused on tree production. An additional problem has been "salvage logging" where large dead trees are removed after a fire, enhancing the erosion following a fire by increased road building and reducing availability of trees to fall into streams and create steelhead habitat. Removal of large wood in any capacity starves rivers of valuable instream cover sources. In many areas, existing harvest rotations of 40-60 years do not allow time for large trees to grow and fall into the stream, creating important stream habitat.

Agriculture. Agricultural and ranching land use practices can negatively impact adjacent streams containing steelhead and other anadromous fish. The trampling and removal of riparian vegetation by grazing livestock destabilizes and denudes stream banks, increasing sediment and stream temperatures (Spence et al. 1996). These activities can lead to a reduction in canopy over stream channels and siltation of pools necessary for juvenile rearing (Moyle 2002). Other impacts of agriculture include stream channelization, large woody debris removal, and armoring of banks to prevent flooding of fields (Spence et al. 1996). These types of activities remain "best management practices" for agriculture, vineyards, and ranching in some parts of the NC steelhead range. All of these activities, in combination with diversions for irrigation, degrade aquatic habitat quality and quantity, reducing its suitability for steelhead or other native fishes while enhancing its suitability for non-native fishes (Harvey, White et al. 2002).

These land uses have also altered floodplain hydrology, increased bank instability, increased sediment delivery and transport of pollutants. Within the river channel, these activities disrupt substrate composition, divert flows, reduce water quality, and inhibit natural processes of temperature regulation. In addition, lagoon and estuary habitats often store excess sediments, have reduced habitat complexity, and are impaired by temperature increases. All of these factors can affect suitability of impacted reaches for steelhead and numerous populations inhabit impaired watersheds where TMDL Basin Plans are being developed.

In the past few decades, illegal water diversions and subsequent habitat degradation of remote headwater streams for marijuana cultivation has become perhaps the most important limiting factor for salmonid survival in first- and second-order streams in the DPS. The unregulated pesticide use, damming, and habitat destruction that typically accompanies illegal grow operations pose a serious threat to the long term persistence of steelhead in the DPS (NMFS 2016), particularly when juvenile steelhead rear and smolt at age-2. Specifically, Mendocino and Humboldt Counties, with their sparse, rural populations and heavily forested landscape, serve as epicenters of these illegal activities. A CDFW study (Bauer et al. 2015) found that in the headwaters of Redwood Creek, home to a key independent population of NC steelhead, illegal and unregulated diversions for marijuana cultivation are likely to consume over 20% of the available water during the lowest baseflow periods of the year. These diversions have significant consequences on habitat quantity and quality for salmonids such as elevated temperature and sediment, increased competition, predation and disease risks, increased stranding rates and delayed migrations, lower growth rates, and reduced survival, and are likely occurring on some scale in remote streams throughout the DPS.

Harvest. While sport fishing regulations require a zero take for naturally produced NC winter steelhead, fishing for steelhead and "trout" continues in large portions of the two largest systems, the Mad and Eel rivers. Angling is allowed on the Mad River for ten months. The fishery is directed towards hatchery steelhead, which are marked, and supports an angler success rate that is normally higher than other North Coast rivers (Sparkman 2003). Natural steelhead populations in the Mad River are at very low levels, reflected in the low harvest of natural produced fish (Sparkman 2003). The mainstem Eel River and its forks support catch-and-release fisheries, which are monitored through the Steelhead Report Card Program. It appears that between 1999-2005, wild steelhead were caught on the Eel and Van Duzen rivers as early as August and as late as May, though a majority of fishing effort was expended during January and February. Steelhead fishing on the South Fork Eel River was limited to between November and March, and the catch rate for wild and hatchery fish did not show any clear relationship in these three basins. Across the DPS as a whole, fishing effort decreased significantly during the period from 1995 to 2005, yet harvest rates did not change much (Williams et al. 2016).

Poaching steelhead, especially on the South Fork Eel and Redwood Creek, are major concerns. In addition, human-induced lagoon breaching facilitates poaching on small coastal rivers such as the Gualala, Garcia, Noyo, Navarro, Big, and Albion rivers by making steelhead more vulnerable in shallow water. In recent years, poaching may have significantly affected small spawning populations in these small, flashy systems. Poaching is especially egregious and widespread on the Garcia River, where state officials lack access to tribal lands for enforcement, creating a challenge to enforcing protective regulations (Scully 2013, J. Fuller, NMFS, pers. comm. 2016). Gill-netting, gigging, clubbing, and other illegal activities catalyzed action from NMFS and CDFW agents to work with local tribes and communities, such as the Pomo Garcia River Tribe, to help ameliorate impacts, but threats still exist (NMFS 2015). While the bag limit for hatchery-reared steelhead is 2 fish per person per day, even catch and release fishing can have an impact at times when conditions are naturally stressful to wild steelhead (NFMS 2016).

NMFS and CDFW worked together in 2016 to improve low-flow fishing closures on many coastal rivers throughout Humboldt, Mendocino, and Sonoma counties to limit harassment and fishing pressure on fish when stream flows are inadequate for salmonids to passively move upstream (NMFS 2016). The closure was due in part to protect fish that were exhibiting stressed behavioral responses due to low flow conditions and were forced to hold in sub-optimal reaches for extended periods of time (J. Fuller, NMFS, pers. comm. 2016). Poaching, especially under such low flow conditions, has been documented to be a major concern in the Eel, Garcia, and Russian river watersheds. Angler outreach campaigns have been enacted to raise awareness and increase compliance when water levels are low, especially during summer and fall months during drought (e.g., 2012-2016).

Estuarine alteration. Estuary/lagoon quality and extent of available habitat are limiting factors for all steelhead (NMFS 2016). What suitable estuarine habitat remains is subject to high turbidity, poor water quality and sedimentation from runoff, and hypersaline conditions in mid-summer that cause juvenile steelhead to leave (CDFW 2014). The estuaries of the Eel and Mad rivers and Redwood Creek have been leveed, subjected to armoring with hard structures, drained, altered by tidegates, and converted for agricultural and rural development, robbing juvenile salmonids of valuable habitat. Essential water purification and sediment storage functions of the Eel Delta, at the bottom of a watershed that delivers more sediment per area than any river in America, have been significantly hampered by human development (Taylor 2015).

Hatcheries. The principal steelhead hatchery in the region is the Mad River Hatchery, using stocks that originated from the South Fork Eel River. Trout from this hatchery have been widely planted throughout the NC steelhead region and may account for some of the genetic ambiguity that exists among populations (Bjorkstedt et al. 2007). Zuspan and Sparkman (2002) estimated 88.5% of the hatchery-produced adult steelhead in the 2000-01 run did not enter the hatchery, suggesting these fish are likely having a significant impact on naturally produced steelhead in the Mad River and elsewhere. The released fish do leave the system rather rapidly, with no juvenile steelhead being captured four weeks after their initial release (Sparkman 2002b). The impacts of juvenile steelhead on the summer-run, winter-run, and half-pounder life histories are not well-known, but the most recent data suggest residualism in the Mad River itself is low among steelhead juveniles, probably because the Mad River Hatchery is located lower in the river than other hatcheries, and fewer juveniles are released compared to years past (250,000 pre-2009; 150,000 post-2009; M. Sparkman, CDFW, pers. comm. 2016). Competition among adults of hatchery and natural origin remain unknown, but broodstock for the hatchery fish originate in the basin now to reduce genetic impacts to natural populations such as outbreeding depression. Large numbers of hatchery origin fish spawn downstream of the Mad River Hatchery each year, which potentially increases competition for redds with natural origin fish and redd superimposition (CDFW 2016). The Hatchery and Genetic Management Plan for Mad River aims to reduce depression, domestication, and reduced fitness by incorporating natural origin spawners with hatchery origin spawners. The goal of the HGMP is to produce fish that are more like the natural counterparts, such that if a hatchery fish strays and spawns in the wild, negative impacts will be lessened (M. Sparkman, CDFW, pers. comm. 2016).

Sparkman (2002b) showed that the late release timing of Mad River Hatchery steelhead smolts in late March/April led to very little overlap among migrating Chinook, coho, and steelhead of natural origin (CDFW 2016 Mad River Hatchery Plan). During this emigration window, flows in the Mad River are high, fast, and turbid, and predation in-river on smolts is likely low as a result. However, carrying capacity of rivers may be exceeded during the outmigration of hatchery smolts, increasing competition for limited food (Spence et al. 1996). Hatchery steelhead have been documented to displace a large percentage of wild steelhead in some streams (McMichael et al. 1999) and they may directly prey upon smaller young-of-year wild steelhead. Other risks from hatcheries include disease transmission, alterations of migration behavior in wild fish and genetic changes that affect subsequent fitness in wild populations such as reduced fitness and productivity of natural stocks (Waples 1991). Such effects have not been recorded in the Mad River but should be explored.

Alien species. Non-native species are present in many NC winter steelhead watersheds, but the invasion of the Eel River system by Sacramento pikeminnow has been the most troubling (Brown and Moyle 1997). Pikeminnow not only prey directly on juvenile steelhead, but also displace them from pool habitat into less desirable riffle habitat, resulting in reduced growth and survival.

Table 4. Major anthropogenic factors limiting, or potentially limiting, viability of Northern California winter steelhead populations. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years; a factor rated "high" could push the species to extinction in 10 generations or 50 years; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction unlikely as a result; and a factor rated "no" has no known negative impact to the taxon under consideration. Certainty of these judgments is high based on peer reviewed and gray literature, direct observation, expert judgment, and anecdotal information. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Major dams present on the Mad and Eel Rivers, which block access to important spawning and rearing habitat.
Agriculture	Medium	Conversion of estuarine wetlands to agricultural lands, diversions, influx of fertilizers and other pollutants into estuaries; impacts of illegal diversions for marijuana cultivation likely high.
Grazing	Medium	Some impacts in lowland areas, especially where marshes have been converted to pasture.
Rural/ residential development	Medium	Effects localized, but increasingly an issue in Humboldt Bay tributaries.
Urbanization	Low	Increasingly an issue in Humboldt Bay tributaries.
Instream mining	Low	No known impact but occurs in some streams.
Mining	n/a	
Transportation	Medium	Roads are an ongoing source of sediment input, habitat fragmentation, and channel alteration.
Logging	Medium	Major activity in many watersheds; dramatic historical impacts in many areas.
Fire	Low	Increased stream temperatures, sediment input to inland streams.
Estuary alteration	High	Estuaries are vitally important rearing, feeding, and migrating habitat and have been significantly altered, reducing their ability to support steelhead.
Recreation	Low	
Harvest	Medium	All natural-origin steelhead must be released, and restrictive fishing regulations have been implemented to protect wild fish, but there is no doubt some fishing mortality of wild fish. Poaching and harvest remains an issue in small coastal streams.
Hatcheries	Medium	Large numbers of juveniles produced by Mad River Hatchery.
Alien species	Low	Sacramento pikeminnow may play an increasing role in predation on juvenile salmonids in the Eel River watershed.

Effects of Climate Change: Climate change is a major threat to the continued persistence of NC winter steelhead. Moyle et al. (2013) rated this DPS as critically vulnerable to extinction from climate change by 2100 in most watersheds, due to degradation of freshwater habitat by:

1. More frequent early-season high-flow events, scouring redds
2. Higher stream temperatures reducing habitat quality
3. Lower stream flows in summer reducing habitat quantity
4. Earlier spring snowmelt reducing juvenile outmigration success
5. Increased frequency and intensity of catastrophic wildfires, threatening salmonid survival with attendant erosion, mass wasting, etc.
6. Altered woody debris availability and characteristics reducing holding areas for juvenile salmonids
7. Increased eutrophication of estuaries that serve as important nurseries and foraging habitat for juvenile and sub-adult salmonids

In addition to these general impacts, high greenhouse gas emissions scenarios associated with the most warming by 2100 are linked to increasing frequency and duration of critical drought, which reduces overall streamflow and increases variability in timing of precipitation events in California (NMFS 2016). Changes in precipitation patterns could lead to larger flood events, contributing sediments from highly erodible terrain that smothers valuable gravel and fills in pool habitat (Williams et al. 2016). Reductions in suitable freshwater habitat are also expected to result in a northward and higher elevational shift in the range of cold water fishes (Haak et al. 2010). As a result, NC steelhead may experience local extirpations and range contractions, as higher gradient headwater streams are inaccessible behind falls, boulder fields, or dams in the DPS. These fish may also shift their distribution northward, where local stocks are less inclined to persist in warmer temperatures (M. Sparkman, CDFW, pers. comm. 2016).

Persistent drought is likely to exacerbate acute problems associated with depletion of summer baseflows, reduction of coldwater refugia, or even stream dewatering during the early fall months by reducing spawning, rearing, and migration habitat and opportunities to reach tributaries for spawning or rearing. Outmigrating smolts may be most susceptible to climate change impacts because spring flows will decline sooner, causing a shrinking of the migration window for juvenile salmonids and reduction of food in the ocean. In estuaries, which are used during smoltification, drought can decrease freshwater inflow, increase duration and extent of saltwater influence and temperature, create hypersaline conditions, and reduce dissolved oxygen. Increased smolt survival at sea is linked to size at migration, so these effects can reduce feeding, growth, and ultimately survival of juvenile salmonids once they reach the sea (CDFW 2014). In Salmon Creek (Humboldt Bay), juvenile steelhead left the estuary in June as freshwater inflows and water quality conditions declined over time due to low freshwater inflows (CDFW 2014).

In the ocean, teasing out impacts of climate change and drought on salmonids remains challenging. Drought and poor ocean conditions tied to climate change and El Nino conditions likely reduced salmonid survival at sea by reducing coldwater upwelling and food availability (Moyle et al. 2013, Williams et al. 2016). Such shifts have the potential to alter ocean circulation and primary productivity in upwelling currents, reducing steelhead growth and survival in the marine environment, creating thermal migration barriers to juveniles and adults in both marine and freshwater, and shifting the range of suitable habitat (sea surface temperatures) northward (Hayes et al. 2016b). In 2014-2015, a persistent patch of very high sea surface temperatures, known as the "warm blob," may have blocked thermal migratory corridors during winter and spring migrations for southern steelhead stocks (Hayes et al. 2016b). While temporary, the occurrence of warm patches of sea surface temperatures for extended periods could restrict marine migration corridors for feeding steelhead and other salmonids and potentially restrict access to ocean ecosystems for fish with anadromous life histories in the future (Hayes et al. 2016b). This is an area in need of more targeted research.

As populations continue to decline and become more fragmented, stochastic events such as increased catastrophic fire may change genetic structure, breeding, and population dynamics in ways that are unrecoverable. Using a threat vulnerability analysis, NMFS forecast that NC winter steelhead populations in Redwood Creek, South Fork and North Fork Eel, and Mattole rivers are most susceptible to climate change impacts in the near future (NMFS 2016).

Status Score = 3.3 out of 5.0. Moderate Concern. NC winter steelhead have a moderate risk of extinction in the next 50-100 years, although better information and improved management could improve this rating (Table 2). The entire DPS, which includes summer steelhead, was listed as threatened under the Federal Endangered Species Act on June 7, 2000 (NMFS 2000), a status that was reaffirmed on January 5, 2006 (NMFS 2006). It is considered to be a Sensitive Species by the US Forest Service. This status will deteriorate rapidly if restoration efforts are not put into effect. NC winter steelhead do not currently have special conservation status with the state of California beyond being a fishery species. Due to their continuing decline, NC winter steelhead should be officially recognized as threatened under the California Endangered Species Act by the Fish and Game Commission or at very least declared a Species of Special Concern.

Table 5. Metrics to determine the status of Northern California winter steelhead, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	3	Multiple watersheds in CA.
Estimated adult abundance	3	A few thousand wild spawning steelhead present annually in the Mad and Eel rivers combined; other populations (Redwood Creek, Mattole, and Garcia) may contain as many, though information is lacking. Total is likely less than 15,000-20,000.
Intervention dependence	3	Require continuous monitoring, protection from poaching, and habitat improvement for recovery.
Tolerance	4	Steelhead are iteroparous and have broad tolerance in fresh water.
Genetic risk	4	Genetically diverse with gene flow among populations, although hatchery influence is a concern.
Climate change	2	Vulnerable in most watersheds but possible refuges present. Moyle et al. (2013) vulnerability score is 17/35 (highly vulnerable).
Anthropogenic threats	2	2 High and 7 Medium threats. Dams and estuary alteration preclude recovery of NC winter steelhead.
Average	3.3	23/7.
Certainty	2-3	Actual numbers of fish poorly known.

Management Recommendations: Northern California winter steelhead are trending downward over time. The DPS requires significant action to recover from legacy impacts of road building, logging, forest fires, poor water quality, and disjointed land use throughout their range. Increasing rural development and illegal diversions and withdrawals for illegal marijuana cultivation throughout the DPS range, coupled with five years of ongoing drought, have significantly stressed populations and fueled their downward trajectory. This trend may be a harbinger of the effects of climate change on the fish.

Actions have been taken recently to help stem the decline during the ongoing (through 2016) drought. Over the past few years, law enforcement has focused on cracking down on illegal diversions for marijuana, violations of fishing closures, and other activity that could negatively impact steelhead populations. Partnerships between NGOs, local landowners, and municipalities have increased utilization of real-time sensors and modeling to reduce water diversions from juvenile rearing habitat during critical summer periods (Lehr 2016). In the future, California needs to continue to develop innovative approaches to conservation by utilizing available legal provisions, such as AB 2121 (California Water Code 1259.2 and 1259.4 for CDFW and NFMS, respectively) to maintain instream flows in coastal streams in order to protect fisheries resources downstream of water diversions (Williams et al. 2016).

In addition to these actions, Governor Brown allocated over $35M as emergency funding for CDFW to monitor water quality, conduct baseline population and stressor monitoring, and carry out fish rescues across the state during the drought. Emergency fishing closures under CDFW COR Title 14 Section 8.00 were enacted in the last three years throughout watersheds in the DPS range due to concerns over low flows and dissolved oxygen levels, high water temperatures, reduced fish passage, and increasing rates of disease and infection among adult populations across Northern California (Nelson 2016). Through the work of NMFS, CDFW, the Native Fish Society, and others, the low-flow closures for the Northern California steelhead DPS were amended by the California Fish and Game Commission to better reflect real-time changes in streamflow in the flashy systems. To improve accuracy of flow conditions-based fishing closures, the South Fork Gualala and Navarro River flow gages were used to develop flow triggers to protect vulnerable salmonids in pools during periods of low flows (D. DeRoy, Native Fish Society, pers. comm. 2016). These actions will aid in reducing poaching and eliminate fishing pressure when salmonids are most stressed and vulnerable during extended low-flow periods. This reduction in angling pressure during these unique but critical periods will likely increase the number of successfully spawning adults (J. Fuller, NMFS, pers. comm. 2016).

The recently released NMFS Multispecies Recovery Plan that covers Northern California winter and summer steelhead highlights specific actions and criteria to help recover the DPS, including attaining low extinction risk in 27 independent and 10 supporting populations across 5 geographic diversity strata (NMFS 2016). Specific threats across diversity strata include dearth of large woody debris and cover for rearing fish, abundance of roads and railroads adjacent to sensitive watersheds and associated sedimentation/erosion, illegal diversion and degradation, presence of barriers to migration, and lack of sufficient high quality spawning and rearing habitat due to uncoordinated land use practices (NMFS 2016). The California Department of Fish and Wildlife is currently revising a steelhead restoration and management plan, which will help compile threats and identify specific actions to restore and manage steelhead in California (Nelson 2016).

However, the current uncoordinated management of steelhead exemplifies the difficulty of placing steelhead stocks into groups based on broad geographic distribution and run-timing (e.g., winter vs. summer steelhead). Although designation of ESUs and DPSs are based upon distinctiveness of life-history traits and distinguishing genetic characteristics, such distinctions are not conserving steelhead life history diversity at the smaller watershed scale. Protection of life history diversity at relevant ecological scales is essential for maintaining large populations of steelhead in the future.

Despite management challenges, important progress has been made to date. CDFW and NMFS have been developing a statewide coastal salmonid monitoring program for a number of years, although it has not been implemented across all regions and coordinated. Developing comprehensive abundance and trend data for coastal salmonids is essential for assessing the viability and recovery of NC steelhead at the relevant ecological scale. California matches federal funds from the Pacific Coast Salmon Recovery Fund to provide annual grants for restoration activities through the CDFW Fisheries Grant Restoration Program, and coordination with the State Coastal Conservancy grant programs has leveraged funds to meet the identified significant habitat restoration needs of NC steelhead.

Timber operations are currently undergoing changes as well. A majority of timberlands along California's coast have or are developing Habitat Conservation Plans (HCP) for listed species, including NC steelhead. For example, Mendocino Redwood Company's HCP will cover 6 high priority watersheds for NC steelhead, as well as Chinook and coho and will be implemented soon (Williams et al. 2016). While these efforts are important, there is a general lack of quantitative monitoring to evaluate the effects of harvest rates, road densities, sediment, and other factors on NC steelhead and other salmonids. Ongoing HCP planning efforts should be vetted by the new viability and recovery framework for NC steelhead developed by Spence et al. (2007), and there is need to better integrate HCPs with other watershed-based management actions and restoration activities across basins.

Perhaps the single greatest opportunity for increasing robust runs of wild salmon and steelhead in California remains large scale restoration and recovery of the Eel River through implementation of action items prioritized by the Eel River Forum and other partners. This ecosystem scale approach to managing salmonid and nonnative fish in the Eel River is essential to maintain the steelhead population in the tributaries and forks of this basin in the long term. The Eel River Forum, a partnership of resource users, landowners, managers, residents, and others, recently released its Eel River Action Plan, which will "coordinate and integrate conservation and recovery efforts in the Eel River watershed to conserve its ecological resilience, restore its native fish populations, and protect other watershed beneficial uses" (Eel River Forum 2016). Key "action" items identified in by the Forum include:

1. Evaluate flow releases from Potter Valley project
2. Prioritize block flow releases to assist out-migrating salmon and steelhead in the spring
3. Evaluate extent of pikeminnow invasion and impacts on salmonids
4. Explore and document salmonid habitat upstream of Scott Dam
5. Determine water dynamics and quality for Lake Pillsbury
6. Evaluate potential use of PG&E lands for salmon and steelhead habitat restoration
7. Understand past and future potential carrying capacity of the Eel watershed for salmonids and how proposed projects would impact it
8. Consider potential changes to operations during FERC relicensing of the Potter Valley Project, slated to begin in April 2017.

It is worth nothing that these actions are in fact proposals to evaluate various problems but not to address them directly. Getting past the planning stage to actually performing in-the-water actions to directly benefit fish should be the highest priority for the agencies and NGOs working on the river.

Hatcheries can also play a significant role in conservation of Northern California Coast steelhead. Further genetic monitoring, in order to implement the hatchery genetic management plan, should be undertaken to find ways to minimize negative effects of hatcheries on naturally-produced NC steelhead. Implementation of the Mad River Hatchery Genetic Management Plan (2016) is also essential for recovering wild steelhead in the DPS by reducing negative impacts of hatchery fish on remaining wild fish. In particular, the use of broodstock of Mad River basin fish and reduced numbers of fish should reduce interactions between natural origin fish and hatchery origin fish, while balancing the goal of providing sportfishing opportunities for steelhead in California.

Innovative, large-scale restoration activities that seek to regulate land use, manage sediment transport and input into streams, restore floodplain and estuary function as rearing habitat, and re-introduce large woody debris instream have occurred throughout the DPS range in the last several years. For example, a decades-long, multi-stakeholder project on the Salt River, in the Eel River estuary, continues to create valuable habitat for rearing fish on floodplain habitat purchased from a willing cattle rancher. Restoration of the stream-estuary ecotone of the Elk River, seasonal flooding of marginal agricultural lands, and construction of off-channel habitats on Salmon and Jacoby creeks, have created habitat that is used seasonally by over 17 species of fish (including federally threatened species such as coho salmon and tidewater goby) and helps support a relatively intact native fish assemblage (Taylor 2015, Scheiff et al. 2016). In September 2013, over 100 whole large conifers and their intact root wads were placed strategically instream to provide scour pools, velocity refuges, and foraging and rearing cover for juvenile salmonids. By allowing large woody debris to interact with flows, the river itself can generate habitat complexity in the form of side-channels, scour pools, and meanders to provide rearing habitat for juveniles and resting areas for migrating adults (Mattole Salmon Group 2015). These types of cutting-edge projects are scaling up from proof-of-concept to become replicable across watersheds. While such projects are important to test ideas and strategies, more coordinated and extensive restoration is required to bolster wild NC steelhead populations.

SOUTH-CENTRAL CALIFORNIA COAST STEELHEAD
Oncorhynchus mykiss irideus

Critical Concern. Status Score = 1.9 out of 5.0. South-Central California Coast steelhead are in long-term decline across their range, despite recent recovery actions taken in core watersheds. Without widespread efforts to restore stream flows and improve access to historical habitat, steelhead will be extinct in southern California within fifty years.

Description: South-Central California Coast (SCCC) steelhead are similar to other steelhead in their meristics and morphology (see North Coast steelhead for full description). This Distinct Population Segment (DPS) is distinguished from other steelhead DPSs by their distribution and ecological/zoological differences between regions (Boughton, unpubl. data). The most recent genetic information suggests that these fish could be grouped with the southern steelhead DPS, but to remain consistent with the management agencies, we treat the south-central California coast steelhead as a distinct entity. Recognizing this DPS separately provides a reason to focus on steelhead conservation in the watersheds of south-central California.

Taxonomic Relationships: Broad taxonomic relationships and a discussion of the nature of ESUs and DPSs can be found in the Northern California Coastal winter steelhead account. In California, steelhead are observed to generally follow a genetic pattern of geographic isolation with distance, which is evident within the SCCC steelhead DPS. Across the West Coast, steelhead diversity generally declines with latitude; south-central California Coastal steelhead and southern steelhead have lower allelic richness (and smaller populations) than those in the North (Garza et al. 2014). As currently designated, the Pajaro River (Monterey and Santa Clara counties) marks the boundary watershed between the Central California Coastal steelhead DPS and the SCCC DPS. Garza et al. (2004) found that clear genetic differences between south-central Coast steelhead and southern steelhead were not apparent. While other steelhead DPSs can be distinguished genetically, the SCCC DPS and the southern steelhead DPS are genetically intermixed (Girman and Garza 2006). These two DPSs are more similar to each other than to steelhead DPSs further north, with little justification for separation (Clemento et al. 2009). The SCCC DPS therefore seems to continue to exist mainly for management convenience, existing between the historical boundaries of the original ESU (Evolutionarily Significant Unit) used to describe the form. More recently, Garza and colleagues (2014) suggest that the DPS boundaries should be updated to reflect barriers on the coast to migration; they suggest that steelhead from Morro Bay north to San Francisco Bay should be grouped together for conservation and management based on similar genetics. According to the study, the current DPS boundaries do not accurately represent the biological structure of steelhead in the region. However, the five-year status review did not recommend any changes, citing lack of necessary information.

Aguilar and Garza (2006) used a molecular marker to determine that a genomic region associated with thermal tolerance and spawning time may have been under positive selection pressure in the population in Chorro Creek (SCCC DPS). Boughton et al. (2006) reported that rainbow trout are found above artificial barriers in 17 of 22 basins in the range of the South-Central/Southern and California Coast steelhead DPSs. In the Salinas and Arroyo Grande watersheds, a genetic comparison of trout above barriers with juvenile steelhead below barriers demonstrated these populations were closely related (Girman and Garza 2006). These findings have been replicated for both SCCC and Southern steelhead (Clemento et al. 2009).

Based on this genetic information and distributional information, Boughton et al (2006) identified 41 historically independent populations of SCCC steelhead in the DPS, including three populations in the large Salinas River watershed. These latter three populations each use spawning areas separated by the mainstem Salinas River; one grouping includes steelhead found in the Nacimiento, San Antonio, and upper Salinas rivers. The 41 populations are divided into four biogeographical regions including (from north to south): Interior coast range, Carmel Basin, Big Sur Coast, and San Luis Obispo Terrace (Boughton et al. 2007).

Life History: Very few biological studies have been done on SCCC steelhead, although they appear to express a diversity of life history patterns similar to other steelhead (see Southern steelhead account). SCCC steelhead complete their life history cycle in freshwater or spend 1 to 3 years in fresh water before migrating into the ocean for 2 to 4 years and returning to natal rivers to spawn. SCCC steelhead and CCC steelhead encounter similar physical habitat features that bound the trajectories of their juvenile life history, including principally small, steep coastal watersheds with flashy flows that reduce juvenile growth, and age at outmigration, as well as seasonally open estuaries, which control smoltification, marine survival, and migration patterns.

Besides exhibiting the three categories of juvenile steelhead life history strategies discussed in the CCC steelhead account (anadromous, freshwater resident, lagoon-anadromous, and variations of these), SCCC steelhead may use fine-scale movements among habitats such as making intra-seasonal movements between lagoons and fresh water and within fresh water movements between reservoirs and tributaries (Boughton et al. 2006, Hayes 2016). Immature steelhead spend several weeks to months in estuaries before entering the ocean. In larger basins (e.g. Pajaro, Salinas Rivers), juvenile life history patterns are limited by desiccation of tributary streams in dry years, which eliminates low elevation reaches as over-summering habitat. Fish may be forced to move upstream into headwater areas with perennial flows, remain in reaches supported by reservoir releases, or to emigrate downstream to the estuary. While the mainstem Carmel, Big, and Little Sur rivers provide decent rearing habitat year-round, the interior rivers (Pajaro, Salinas) are too warm for steelhead from late spring through summer, and are primarily used as migration corridors (J. Casagrande, NMFS, pers. comm. 2016). However, warm temperatures can create thermal barriers to migration. Juvenile steelhead in Southern California grow mostly during the winter in fresh water when temperatures are optimal (Krug et al. 2012).

Adult steelhead return from feeding in the ocean to enter watersheds to spawn in SCCC streams between January and May, and as lagoon bars breach or there is sufficient flow to allow passage (Boughton et al. 2006). SCCC steelhead embryos likely have accelerated hatching rates due to warmer stream water temperatures. In years with low rainfall, lagoon barriers may not breach during the rainy season and migratory access between the ocean and fresh is impossible. Presumably under such circumstances, adults spend another year in the ocean before returning to try again and older juveniles suffer high mortality.

SCCC steelhead display a high degree of life history plasticity that is likely determined based on the interaction of genetics, environment, and other factors. According to recent genetic work, Chromosomes Omy5 and Omy12 in rainbow trout have been found to be associated with aspects of sexual maturity and run-timing (NMFS 2016, Pearse et al. *in review*), suggesting a genetic basis for this life history flexibility. Traits with a genetic basis can be inherited such as expression of anadromy, smoltification, and various run timings (Kendall et al. 2015). Pearse et al. (2014) found that rainbow trout in anadromous waters had one kind of Omy5, while those in waters formerly available to anadromous fish upstream of impassable dams, had an inverted region of the Omy5 gene locus on the same chromosome. Interestingly, the notable exception to this pattern was found in the adfluvial population in Upper Arroyo Grande Creek above Lopez Dam, which has predominantly anadromous Omy5 genes. The potential for steelhead to make life history switches between adult life histories has been demonstrated for anadromous and resident fish (Zimmerman and Reeves 2000). Adding to this complexity, inland resident juvenile trout may exhibit smolt characteristics, and populations without anadromous adults can give rise to smolts that emigrate to sea (Boughton et al. 2007). Rundio and colleagues (2012) found that despite genetics, female juvenile rainbow trout were more likely to emigrate to sea than males; this could be because their fecundity is more closely tied to body size, which increases significantly with time spent at sea, than their male counterparts. In addition, Pearse and others (*in prep*.) found that in the Big Sur River, emigration to the ocean in age-0 juveniles was associated with chromosomes, sex, and juvenile body size, rather than a single factor.

Habitat Requirements: SCCC steelhead have habitat requirements similar to those of steelhead populations further north, but presumably share thermal tolerances with southern steelhead (NMFS 2016). In general, they need cool, flowing waters, a diversity of spawning, rearing, and feeding habitats, access to the ocean, and available prey. These requirements can be difficult for SCCC steelhead to find, especially in dry years. Optimal mean monthly temperatures in potential rearing areas with limited food supplies are considered to be 6-10ºC, with temperatures over 13ºC being unfavorable for growth and survival (NMFS 2007, see southern steelhead account for more details on temperature requirements). A recent study of the mainstem Big Sur River (Holmes et al. 2014), however, found average water temperatures of 16º C during summer that dropping to 14º C in the fall, with juvenile steelhead of multiple age classes using different habitat types opportunistically. Near the mouth of the Big Sur River stream temperatures in reaches containing steelhead were recorded as high as 17.2º C in 2014 and 15.8º C in 2015. The fact that this river is coastal and influenced by cooling from marine fog suggests that other rivers in the watershed are regularly above the 13ºC temperature threshold. In Uvas Creek, Casagrande (2010) found that juvenile steelhead grew rapidly in reaches where peak summer daytime temperatures were in excess of 20º C, because areas receiving more sunlight were associated with higher invertebrate production (food); these areas also cooled down to less than 16º C at night, providing temperature relief. In addition, juvenile steelhead rearing in waters with temperatures >16º C were almost exclusively found in high velocity habitats, such as runs and riffles, where invertebrate drift was higher to help them offset energy expended in greater than optimal temperatures (J. Casagrande, NMFS, pers. comm. 2016). Consistent with this study, Thompson et al. (2012) found no steelhead at sites in tributaries to the Salinas River where the maximum temperature exceeded 26ºC or where the mean temperature exceeded 21.5º C.

In addition, juveniles and adults require access to a diversity of habitats. In general, CDFW passage guidelines suggest that adult steelhead require depths > 24cm, while juveniles > 15cm in length require depths > 15cm, and juveniles < 15cm require depths > 9.1 cm (CDFW 2010, pg. 42). These depths must be maintained across > 25% of the channel width, and must be > 10% of the continuous channel width. Finally, pool depth must be at least 1.25 times the necessary jump height for steelhead to leap over a waterfall or obstruction to allow at least partial passage (CDFW 2010). Holmes and colleagues (2014) found differential habitat use in juvenile steelhead based on their size in a study spanning 2010-2012 on the Big Sur River (Table 1).

Table 1. Characteristics of habitat used by different size classes of steelhead in the Big Sur River, based on data in Holmes et al. 2014.

Juvenile Size	Preferred Depth	Preferred Velocity	Preferred Cover Type	Notes
< 6 cm	< .25 m	< 15 cm/s	95% used hard substrates	70% in pools or runs; size class stayed closest to bank
6 – 9 cm	.5 m	40 cm/s	65% used cobbles or boulders	65% in riffles or runs; 95% in areas with no overhead cover; middle distances from bank
10 –15 cm	≥.5 m	45 cm/s	Most used wood and hard substrates	Found furthest from bank with almost no overhead cover

Juvenile steelhead use deeper, faster water as they grow to access more food to meet higher metabolic demand; all size classes strongly to avoid water < 10 cm deep (Holmes et al. 2014).

Often, mainstem river and lower reaches of tributary creeks are seasonally dry and are primarily used as migratory corridors. In cases when large wood provides over-summering habitat, SCCC juvenile steelhead will use mainstem creeks and rivers with perennial flows (Thompson et al. 2012), especially in watersheds where headwater streams are dry during this period (Boughton et al. 2006). On San Luis Obispo Creek, Spina et al. (2005) observed juvenile steelhead using essentially every pool. Boughton et al (2006) presented a similar result and found that that shortly following rains that created flow in streams, juvenile steelhead were observed utilizing newly-wetted stream segments. Thus, sufficient habitat with perennial flows and cover are critical requirements for juvenile rearing and full expression of life history variation. For cover, SCCC steelhead depend on instream wood often made up of hardwood trees and root wads (Thompson et al. 2008, 2012). Thus their populations are particularly susceptible to wildfires that that remove wood (sources of habitat complexity and pool scour), which are important for juvenile overwintering survival (J. Casagrande, NMFS, pers. comm. 2016) and cause sedimentation, especially in conjunction with drought and climate change.

In addition, lagoons at the mouths of coastal streams are important for juvenile feeding, rearing, and saltwater acclimation prior to ocean entry (Hayes et al. 2008). Lagoons offer juvenile steelhead of various sizes opportunities to grow more quickly than they can in small, intermittent or ephemeral tributaries. Juvenile steelhead rearing in lagoons can grow fast enough to smolt and migrate to the ocean at age 1+. Juvenile survival at sea has been linked to greater size at ocean entry, so the more rapid growth associated with a lagoon-rearing life history is likely important for maintaining runs. In addition, fish that spent time rearing in estuaries are disproportionately represented in adults returning to coastal watersheds (Hayes et al. 2008).

Distribution: SCCC steelhead are distributed between the Pajaro River (Santa Clara/San Benito counties) south to, but excluding, the Santa Maria River (San Luis Obispo/Santa Barbara counties, Figure 1). Although habitat quality is low and population sizes in most coastal basins seem too small for persistence, prior to the 2012-2016 drought, steelhead were still found in almost all SCCC DPS coastal watersheds in which they were historically present (Boughton et al. 2007). Steelhead have also been found in basins in the DPS range with no recent historical records of steelhead, including Los Osos, Vincente, and Villa creeks, illustrating the opportunistic nature of the species (Boughton et al. 2005) and the importance of protecting such watersheds for recovery (Garza et al. 2014). Becker and Reining (2008) provide a comprehensive guide to SCCC steelhead distribution and status prior to 2008.

Watersheds in this DPS are separated into four biogeographic population groups (BPGs) that are categorized by migration connectivity and reliability, summer cool-water refuges, intermittence of streamflow, and winter precipitation (Boughton et al. 2007). In the Big Sur Coast and San Luis Obispo Terrace BPGs, 37 streams contain steelhead and bear more ecological resemblance to steelhead streams in northern California (J. Smith, SJSU, pers. comm. 2008) than to streams in the interior regions of the DPS. These watersheds are kept cool and moist by marine-based weather patterns and fog layers. The Big Sur River, unlike the Pajaro, Salinas, and Carmel rivers, remains undammed and provides high quality habitat that likely supports migrants to other watersheds (Holmes et al. 2014). With its unimpeded flow, the Big Sur River is considered a steelhead stronghold by the California Stronghold Team (2012).

Other BPGs include rivers that cut across the coastal ranges and extend inland through long valleys dominated by agriculture. These include the Pajaro River, Gabilan Creek, Arroyo Seco, Salinas River, and Carmel River in the Interior Coast Range and Carmel River BPGs, and have warmer climates that alter the timing and delivery of water for steelhead habitat.

Figure 1. SCCC DPS boundaries and core watersheds. From: NMFS 2016 Figure 4, pg. 15.

At sea, SCCC steelhead are likely found as far south as northwestern Mexico and appear to be more solitary than other salmonids (Busby et al. 1996; Good et al. 2005). It has been hypothesized that more northern populations of steelhead migrate to a cool water patch far offshore of the Klamath-Trinidad coastline before migrating to the North Pacific feeding grounds, but more study on this subject is necessary. Steelhead that are captured in trawl surveys are generally encountered much farther offshore, and in fewer numbers, than Chinook or coho salmon (Harding 2015). But there is no specific information on marine habitats of SCCC steelhead.

Trends in Abundance: Historically, annual runs of SCCC steelhead seem to have been around 27,000 adults (NMFS 2007) in wetter years. CDFG (1965) suggests that the DPS-wide run size was as high as 17,750 adults in 1965, after much of the South-Central California coast had been developed following World War II (NMFS 2013). CDFG (1965) optimistically estimated that the main watersheds in the DPS, the Pajaro, Salinas, Carmel, Big Sur and Little Sur rivers supported a run of about 4,750 adults annually during the 1960s. Good et al. (2005) estimated less than 500 adults returned annually to each of these rivers in 1996 (<2,500 total). By these rough estimates, the SCCC steelhead DPS experienced declines in run sizes of about 90% by 1996 (Boughton et al 2007). NMFS has set a target of 4,150 total anadromous adults in the DPS per year to attain recovery, which is only about 20% of presumed historical numbers. However, very little population monitoring exists for SCCC steelhead. All signs point to lower abundances across the DPS during the 2011-2016 years of drought (CDFW 2014, 2016). Given the information presented below, we estimate there are now an average of about 500 (200-800) spawners returning each year, total, to all streams in the DPS region, although our confidence in this estimate is low because the data is so poor.

Estimating adult steelhead returns in the DPS from redd surveys, spawner surveys, cameras, or other means has remained a challenge. Sampling the mostly inaccessible mountain tributaries on private lands during the rainy winter months to obtain spawner estimates has not been adequately addressed. While imperfect, redd counts have been made for several years in the Carmel River, but are not comprehensive due to high flows and turbidity. On the Carmel River, DISON sonar has been used to estimate steelhead abundance in turbid flows, but has been shown to be unreliable as individual fish may move back and forth in front of the camera several times in one migration (NMFS 2016). Other attempts to estimate adult abundance in the DPS include visual imaging (Cuthbert et al. 2014) and a counter on a fish ladder (MCWRA 2013) on the Salinas River, but these have issues with reliability due to their specific implementation.

Opportunistic spot-check sampling has remained the best source of information on steelhead presence/absence and juvenile abundance during California's ongoing drought. Estimating relative juvenile abundance and documenting distribution occurs in some locations as well, such as electrofishing on Uvas and Llagas creeks (Pajaro River), and seine surveys in the estuary of the Pajaro River. Although flows were improved for adult access and smolt outmigration, no adults were observed, and only one potential redd was found in lower Carnadero Creek (lower Uvas Creek) in 2016. Young-of-year (YOY) abundance was the lowest since sampling began in 2005, marking the first time that no YOY were found following a winter when anadromous access was possible. Five juvenile *O. mykiss* (age 2+) were captured, however. The extremely low juvenile abundance in 2014 and 2015 also likely resulted in very few smolts produced in spring 2016 (Casagrande 2016).

Annual electrofishing surveys have also been conducted in Corralitos Creek (Pajaro River) since 2008. Further south, intermittent sampling of juveniles occurs on the Big Sur River (Holmes et al. 2014) and Big Creek (Monterey Co.), and Chorro, San Luis Obispo, Pismo, and Arroyo Grande creeks (San Luis Obispo Co., CDFW 2014, 2016). In 2014, the Salinas, Carmel, and Big Sur rivers in Monterey County and Santa Rosa, Chorro, San Luis Obispo, and Pismo creeks in San Luis Obispo were all cut off from the ocean due to low rainfall associated with the drought, and intermittent flow blocked adult and juvenile access to and from lagoons (CDFW 2014). Habitat connectivity and availability was greatly reduced through significant dry-back of stream reaches, resulting in mortality and loss of recruitment, as seen in electrofishing data from Chorro and San Luis Obispo creeks (CDFW 2014, Figure 2).

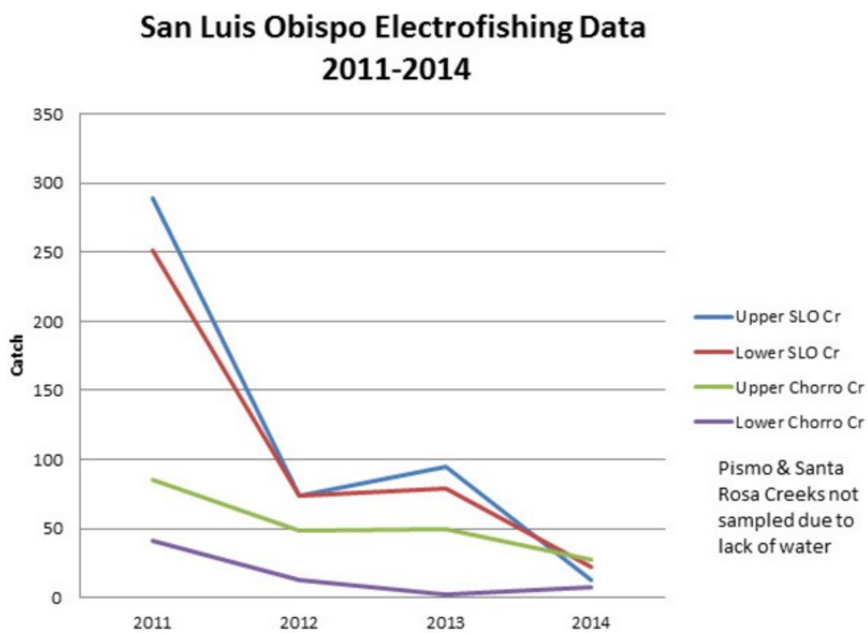


Figure 2. Juvenile steelhead abundance from electrofishing data in Chorro and San Luis Obispo Creeks, 2011-2014. From CDFW 2014, Fig. 3.

Fish rescue efforts have provided some information on adult occupancy in select watersheds. For example, in the Pajaro River watershed, a local citizens group, the Coastal Habitat Education and Environmental Restoration (CHEER) conducted fish rescues annually from 2006 through 2013. During that time, CHEER rescued an average of 12 adult steelhead from isolated pools and drying habitat from 2006-2013. Since historic drought took hold in California in 2012, juvenile steelhead produced in the Uvas Creek subbasin have dropped sharply (J. Casagrande, NMFS, pers. comm. 2016). Even when winter storms arrived, sufficient flow for passage was unavailable, as water tables were so low that water soaked into parched ground without running off (CDFW 2016b). As a result, there has been no documented spawning in Uvas Creek since 2013, because adults could not reach spawning areas before drying events occurred (Casagrande 2015). Before the drought, reasonable densities of juvenile steelhead or rainbow trout (20-30 fish/30m) were documented in the Uvas Creek watershed in some years (Casagrande 2013, 2014), but these densities have plummeted. In 2013, 39 adults and 260 juveniles were counted in Uvas Creek and its tributaries (Casagrande 2014); in 2014, only one adult and seven age 1+ or 2+ juveniles were captured (Casagrande 2015). On January 31, 2014, after years of reduced releases from the upstream Uvas Reservoir and significant drying of the stream, NMFS and San Jose State University biologists captured 180 juvenile steelhead from reaches subject to drying and relocated them to downstream areas with sufficient habitat (CDFW 2014b, Casagrande 2014). In 2015, no adults and 51 age 0+ juveniles were captured during electrofishing efforts in summer and fall (Casagrande 2016). In 2016, no YOY juveniles were detected in Uvas Creek or its tributaries for the first time since surveys began in 2005 (Casagrande 2017). In the nearby Salsipuedes Creek, steelhead smolts have been documented emigrating through College Lake (Podlech 2011).

On the Salinas River, monitoring of adult returns occurred in 2011, 2012, and 2013, with a mean of 22 adults returning per year, with the largest recent count of migrating adults topping out at 46 upstream of the lagoon, but the weir was taken down during high flows and some adults could have passed undetected. From 2014-2015, the Salinas River did not reach the ocean, so no upstream migrants were detected. At the upper reaches of the Salinas, juvenile *O. mykiss* were present in the Nacimiento River (below Nacimiento Dam) as of summer 2015, but were last documented in the San Antonio River in 2012/2013. Arroyo Seco retains juvenile rainbow trout but has not supported any anadromous fish in the last several years due to lack of sufficient flows for passage (J. Casagrande, NMFS, pers. comm. 2016). Redd surveys were conducted in the Big Sur River in 2012, 2014, and 2015, while annual redd surveys began in San Luis Obispo Creek in 2015. Juvenile abundance surveys have been conducted in other watersheds in San Luis Obispo County, but have not used the Coastal Monitoring Plan techniques for population estimates; however, they do provide distribution and trend information over time (NMFS 2016).

The one time series of abundance greater than twenty years in duration in the DPS is from the Carmel River (Monterey Co.). Adult steelhead counts on the Carmel River at the former San Clemente Dam ranged from 0 to 1,350 between 1962 and 2002, with an average run size of 821 adults (Good et al 2005, MPWMD 2007). Steelhead in the Carmel River underwent a drastic decline that lasted into the late 1980s (Good et al. 2005), and saw a rebound in the early 2000s due to habitat restoration and hatchery releases (NMFS 2016). The Carmel River population has continued on a downward trajectory until removal of San Clemente Dam in 2015 (Figure 3).

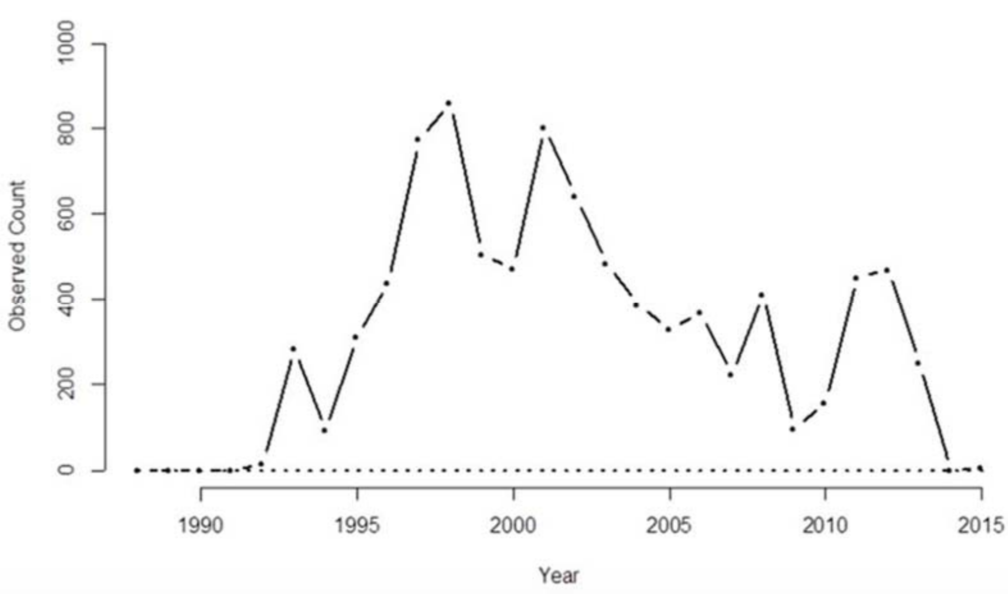


Figure 3. Adult steelhead at San Clemente Dam, 1990-2015. From NMFS 2016, Fig. 8, pg. 29.

This decline of nearly 17% per year coincides with considerable effort to release thousands of smolts per year from Sleepy Hollow Steelhead Rearing Facility (SHSRF, Figure 4). During the 1988-91 drought, the Carmel River Steelhead Association used a seawater facility to raise and release hundreds of thousands of steelhead smolts per year, which could have contributed to a spike in adult returns after 1991 (NMFS 2016). Most returning adults since 2000 are from the captive breeding program. In the river, wild young-of-year (YOY) steelhead size and numbers are also declining (Boughton 2016). Arriaza (in review) suggests that a decline in smolt growth rates may be reducing survival to adulthood.

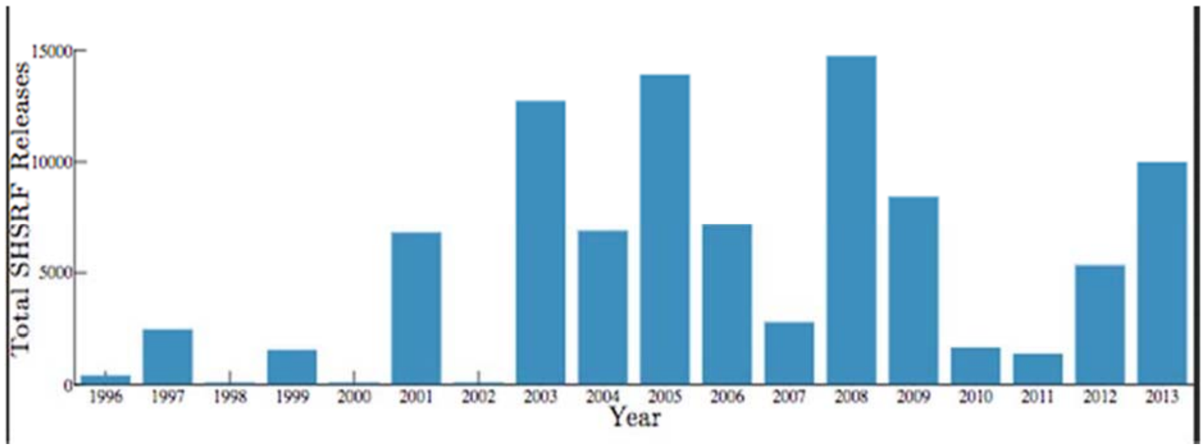


Figure 4. Sleepy Hollow Steelhead Rearing Facility smolt releases, 1996-2013. From Boughton, 2016, Fig. 3, pg. 9.

While data from surveys of juvenile steelhead are difficult to evaluate in the context of run size and viability, this is the principal data available from a number of watersheds and is at least an indication of habitat integrity and presence/absence rainbow trout. Although steelhead numbers within the SCCC steelhead DPS have declined dramatically, about 90% of historical habitat continues to be occupied (NMFS 2016). In Big Creek, a spring-fed watershed along the Big Sur coast, a fair number of age 1+ and 2+ steelhead (NMFS 2016) inhabit the lower reaches during summer. It is possible that higher rainfall, lower air and water temperatures, perennial flows, and relatively few anthropogenic stressors in watersheds of the Big Sur Coast Region allow populations in this region to persist, despite limited habitat area. The resilience of SCCC steelhead depends on favorable over-summering conditions that benefit age 1+ and 2+ steelhead.

As indicated previously, it is reasonable to assume that the average number of SCCC steelhead spawners throughout their range in a wet year is probably on the order of 1,000 fish. According to the most recent National Marine Fisheries Service (NMFS) five-year status summary review, there are currently less than 500 spawning adults across the DPS each year (NMFS 2016), although previous assessments likely underestimated this number in some watersheds, such as Uvas and Carralitos creeks, which had adult returns of perhaps 60 or more adults in 2008 alone (J. Casagrande, NMFS, pers. comm. 2016). Recent and reliable data is notably absent from the primary core-1 populations in Little Sur River and San Jose, San Simeon, Santa Rosa, Pismo, and Arroyo Grande creeks (NMFS 2016). Williams et al. (2016) suggest that there are very small (<10 adults) but persistent runs in most streams at the southern edge of the SCCC DPS range each year, except in years when there has been insufficient winter flows to breach bars at the mouths of lagoons. This persistence could be due to migrants from source populations elsewhere or contributions of smolts from non-anadromous fish in these basins. The drought years of 2012-16 resulted in limited spawning and rearing throughout the DPS. This drought, coupled with above-average sea surface temperatures and poor productivity in the ocean (Mantua 2015), has presumably sharply reduced abundance of SCCC steelhead.

Factors Affecting Status: NMFS (2013) identified several principal natural threats to SCCC steelhead in their Recovery Plan for the DPS, including: (1) land management practices that alter natural stream flow patterns, (2) estuarine degradation, (3) alteration of floodplains and channels from agricultural development, (4) dams and diversions, (5) shifts in climate and ocean productivity, (6) urban and rural development, and (7) catastrophic fire.

The threats posed by alteration of streams and lagoons are principally associated with land use practices in the Pajaro, Carmel, Salinas (including Arroyo Seco, San Antonio, and Nacimiento Rivers). Surface water diversion and groundwater pumping associated with agriculture and urbanization have resulted in the dewatering of streams, modification of river and creeks channels, and addition of toxic materials and nutrients that degrade water quantity and quality. These activities have reduced the frequency, duration, timing, and magnitude of flows available for rearing and migrating steelhead. Due to the geography of the landscape, high winter and spring flows are critical for breaching lagoon mouths to allow adult steelhead spawning migration, and juvenile steelhead emigration to the ocean. The encroachment of agricultural, industrial, and residential developments into riparian and floodplain channels of SCCC steelhead rivers and creeks has caused loss of riparian cover to maintain suitable stream temperatures, instream cover, food resources, and over-summering habitats for juveniles. In addition, dams have blocked a significant portion of spawning and rearing habitat, and have reduced flows, altered downstream habitats, and blocked or impeded migration. Many Big Sur watersheds are on public lands or in areas with less human development, and are more able to maintain populations of steelhead.

Dams. Dams and diversions throughout the SCCC DPS range alter the amount and timing of flows throughout the Pajaro, Salinas, and Carmel river basins, block migration corridors for spawning and rearing steelhead, and alter natural flow regimes that are essential for maintaining habitat accessibility and connectivity (NMFS 2013). Major water supply and flood control facilities tend to threaten the larger rivers in the DPS, though smaller diversions reduce available steelhead habitat and alter flow regimes as well (NMFS 2016). In general, impacts of dams on steelhead are negative and are most pronounced on the main stems of rivers, blocking access to generally high-quality habitat in upstream reaches and tributaries, increasing temperatures of downstream water, altering fish communities, and reducing gravel and sediment transport from the upper watershed to the coast. These projects have increased direct mortality of steelhead, altered stream banks and channel morphology, and increased erosion, sedimentation, loss of channel complexity, pool habitat, and large wood inputs to steelhead streams (NMFS 2013).

Although impoundments and dams such as Uvas Dam (Pajaro River) have blocked access to historical spawning and rearing habitat in the headwater reaches and disrupted natural hydrologic processes, management of flow releases for aquifer recharge and flood prevention provide opportunities for successful rearing and both inland and seaward migrations of steelhead in some populations. In addition to actions seeking fish passage past these dams (or dam removal where feasible), improved adaptive management of these reservoir operations could yield benefits to steelhead production (J. Casagrande, NMFS, pers. comm. 2016).

In 2015, San Clemente Dam was removed from the lower Carmel River, and significant habitat restoration began to restore and enhance access and habitat for spawning adults and rearing juveniles. In addition to re-routing the river channel around sediment left after dam removal and step-pool creation throughout the project site, a feasibility study for removal of Los Padres Dam upstream of the project site has begun. This project could serve as an example for future dam removal projects throughout the DPS range.

Agriculture. Extensive agricultural development for row crops and orchards in the Pajaro, Salinas, Pismo, San Luis Obispo, and Arroyo Grande basins has significantly degraded mainstem river habitats, floodplains, and estuaries in the SCCC DPS (NMFS 2013). Agricultural diversions (a) reduce surface flows, especially in critical summer and fall months, (b) reduce groundwater recharge, thus lowering the water table, and (c) often degrade water quality in the streams. Agricultural return water flows off the fields and into streams at higher temperatures and with increased levels of nutrients and pesticides, making much of the habitat unsuitable for steelhead.

Estuarine alteration. Estuaries and lagoons are critical to rearing steelhead, especially in the small, coastal watersheds throughout the SCCC DPS range (Boughton et al. 2007). Much of this estuarine habitat has been lost, especially in northern portions of the range (NMFS 2013). Urban encroachment and associated water diversions alter estuary dynamics by reducing water quantity and quality, shrinking the habitat juvenile steelhead have available for rearing, especially in summer and fall months. Not only are these habitats important for increasing size at smolting and ocean entry for juvenile steelhead, they represent choke points through which all migrating juveniles and adults must pass. Reductions in their size and function impact all life history stages of steelhead.

Fire. Wildfires are a natural occurrence and are important to the chaparral plant community that dominates much of the SCCC DPS range. However, after decades of suppression practices by federal and state agencies, NMFS (2013) considered catastrophic fire events, such as the recent Pfeiffer and Soberanes fires along the Big Sur coast, as "very high" threats to the recovery of SCCC steelhead throughout their range. Fire removes large wood that can provide shading and instream cover for all life stages of steelhead, and cause extensive sedimentation and smothering of substrates after precipitation events (Thompson et al. 2008, 2012). Because many SCCC steelhead-bearing coastal streams are small in size and area, and bounded by rugged mountainous canyons, extensive fires can threaten entire core populations.

Hatcheries. Despite decades of stocking out-of-basin rainbow trout throughout the SCCC DPS region, Garza et al. (2014) found that this practice has not altered the genetics of steelhead in coastal basins, with a few exceptions. For several years, no hatchery rainbow trout have been stocked in anadromous waters and CDFW only plants unviable triploid fish in waters that are inaccessible to the ocean (NMFS 2016). More recent investigation (Abadia-Cardoso et al. 2016) has documented limited genetic introgression between wild and hatchery fish, even in the Carmel River watershed, even with continuing operation of a captive breeding program at Sleepy Hollow Steelhead Rearing Facility on the lower mainstem. While only limited introgression has been documented due to the use of natural-origin, rescued adult steelhead from the Carmel River, evaluation of selection pressures on remaining wild fish have not been fully evaluated.

Harvest. It is illegal to harvest (take) SCCC steelhead. There is very low catch of steelhead at sea and such fish are not retained. Legal catch-and-release fishing is permitted for steelhead and is monitored by CDFW through steelhead report cards. Records indicate extremely low levels of fishing effort from recreational anglers in the DPS during 1993-2014 (CDFW 2014). Despite this low effort, Good et al. (2005) concluded that recreational angling is a limiting factor on SCCC abundance; as a result, CDFW has restricted legal fishing access in the region to certain days of the week and downstream of major road crossings in recent years to protect migrating adults. Due to extremely low levels of abundance in most watersheds and reduced flows during ongoing drought, legal catch-and-release fishing likely results in some mortality or sublethal impacts on wild fish. In the absence of widespread monitoring, fishers provide valuable data on adult presence through catch reporting and anonymous tips regarding illegal activities and poaching. Poaching remains a major issue in the Pajaro and Salinas watersheds, with considerable documented effort to trap fish illegally (Casagrande 2014).

Alien species. Alien species play a role in limiting expansion of some populations in the SCCC DPS. Efforts have been undertaken on Chorro Creek to remove invasive Sacramento (*Ptychocheilus grandis*), a known predator of juvenile salmonids. In San Luis Obispo Creek, efforts are being made to control invasive, water-thirsty giant reed (*Arundo donax*) that can take over riparian habitat. On the mainstem Salinas River, Monterey County Resource Conservation District has removed significant stands of Arundo to reduce water use and flood potential (J. Casagrande, NMFS, pers. comm. 2016). In Santa Rosa Creek, projects aimed at removing invasive eucalyptus trees have helped reduce water demand in the watershed. Riparian restoration efforts that include reducing invasive plants are ongoing in Pismo and Walters creeks near Morro Bay. Cucherousset and Olden (2011) showed that predation on juvenile steelhead by American bullfrogs (*Lithobates catesbeianus*) and basses (*Micropterus* spp. and *Morone saxatilis*), can be significant and both bass and frogs may compete for resources with juvenile steelhead. Striped bass have been documented in the Pajaro, Salinas, and Carmel rivers; CDFW has been trying to remove them from the Carmel Lagoon for a decade (J. Casagrande, NMFS, pers. comm. 2016). In the Pajaro River drainage, striped bass and several other species of nonnative predatory fishes have been captured far inland in Miller Canal and Carnadero Creek; this is noteworthy because of the potential threat they pose to out-migrating steelhead smolts in spring (Casagrande 2011). Striped bass are known to occur in the Pajaro River Lagoon, and have recently been documented in low numbers in San Felipe Lake in the upper reaches of the watershed (Casagrande 2010). In the both the upper and mainstem Pajaro River watershed, shading due to nonnative evergreen acacia trees (*Acacia auriculiformis*) is potentially limiting productivity and feeding opportunities for juvenile steelhead, while Himalayan blackberry (*Rubus armeniacus*) is holding potential gravel and cobble at stream edges, which need to be mobilized in the river to provide rearing and spawning habitat (Casagrande 2017).

In addition to the factors described above, synergistic effects of disease, predation, and other factors likely have minimal consequences in healthy watersheds with large steelhead populations, but may negatively impact the small remaining populations of SCCC steelhead, especially when considered in concert with the low flows and high temperatures. These conditions are strongly associated with drought in degraded lower watersheds such as the Salinas River. Miller et al. (2014) found that diseases likely reduce already low salmonid populations when remaining fish are exposed to warmer waters than are preferable for the species. Osterback et al. (2015) determined that western gulls (*Larus occidentalis*) predation on juvenile steelhead could be as much as 2.5 times higher than historically due to the increased abundance of gulls due to artificial feeding opportunities. Taken together, these impacts are likely to reduce small populations of SCCC steelhead even further.

Table 2. Major anthropogenic factors limiting, or potentially limiting, viability of populations of South-Central California Coast steelhead. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Major and minor dams in the Pajaro, Salinas, and Carmel watersheds block or alter spawning and rearing habitat.
Agriculture	High	Extensive water diversions for agriculture, especially in the Pajaro and Salinas basins, limit populations and reduce water quantity and quality; water often present at wrong time for steelhead.
Grazing	Low	Livestock grazing is not widespread in the SCCC range.
Rural /residential development	Medium	Diversions for rural residential use likely impact rearing and over-summering habitat for juveniles, especially in watersheds bordered by mostly private land.
Urbanization	Medium	The region is relatively rural, though urbanization in the Pajaro, Salinas, and Carmel River basins is increasing demands for limited water and degrading its quantity and quality. The City of Salinas is the largest city in the DPS >170,000 people and relies on groundwater pumping from the over drafted Salinas aquifer.
Instream mining	Low	Gravel mining not widespread and regulated.
Mining	Low	Heavy metal contaminants from historical mining may settle in lagoons and reduce suitability for juvenile rearing.
Transportation	Low	Highway 1 bisects every watershed, and most have several bridge crossings with only short stretches of habitat altered.
Logging	Low	Mostly legacy impacts of timber removal that reduce large wood input into streams and reduce cover.
Fire	High	While uncommon, recent large fires have closed access to several Big Sur watersheds, reduced recruitment of logs to streams, and increased sedimentation.
Estuarine alteration	High	With the exception of the Big Sur BPG populations, most estuary and lagoon habitats in the DPS have been altered by encroachment of developments, agriculture, and water resource development.
Recreation	Low	Recreation may negatively impact behavior of steelhead.
Harvest	Low	Harvest of wild steelhead is prohibited, and catch-and-release fishin effort is extremely low.
Hatcheries	Low	Decades of stocking hatchery trout has had little genetic impact.
Alien species	n/a	Predation by alien species, such as striped bass and catfish, may be problem in the Pajaro and Salinas estuaries, and alien plants alter stream habitats. Impact on steelhead smolts is unknown.

Effects of Climate Change: Climate change will likely exacerbate the decline of SCCC steelhead primarily by reducing the temporal and spatial availability and accessibility of usable habitat throughout their range. As a consequence, Moyle et al. (2013) rated SCCC steelhead as "highly vulnerable" to climate change.

As temperatures rise and precipitation patterns shift to reduce rainfall in Central and Southern California, SCCC steelhead will be exposed to greater periods of higher water temperatures and more flow variability, eventually outpacing their ability to avoid such exposure (Wade et al. 2013). A changing climate will reduce freshwater inflow to rivers, streams, and estuaries, alter nutrient cycling, and reduce sediment transport (NMFS 2016). With low flows, remaining water may become too warm and oxygen-poor to support juvenile steelhead growth and survival. In general, regions with lower latitude and elevation will be subject to the greatest increase in duration and intensity of higher air and water temperatures (Wade et al. 2013). The SCCC steelhead DPS is very low in elevation and near the southern edge of the species' range. In the ocean, climate change is likely to increase sea surface temperatures and ocean acidity, reduce estuarine habitat through sea level rise, reduce upwelling currents that provide prey sources, and lower marine productivity and salmonid survival off of California's coast (NMFS 2016). For a full description of how climate change impacts steelhead at the southern edge of their range, see the Southern steelhead account.

Climate change is also likely to cause more frequent and more intense drought, and reduce usable habitat. According to Boughton et al. (2007), multi-decade droughts have occurred in the South-Central California Coast area throughout history, yet steelhead have persisted. However, the ongoing drought in California has been characterized as a "hot drought," with several of the hottest years on record coming in the last five years alone. A hotter, drier climate regime for California is expected, and is likely to increase the frequency and magnitude of catastrophic wildfires in California. In addition to above average temperatures and below average rainfall across most of California, precipitation patterns shifted as well.

The changes that took place during the 2012-2016 drought are a harbinger of things to come. For example, half of all the precipitation in 2014 fell during a few storms during a very short time period (December), causing a quick flushing of rivers to the ocean. For steelhead, ideal flows should occur throughout winter and spring to provide passage for spawning adults and smolt outmigration. In 2015, total rainfall in the region was even lower than in 2014, on 25% of average, causing drying of streams and necessitating emergency fishing closures and fish rescues (CDFW 2014) in the DPS. During this time, flows in the Big Sur River were less than the 0.6 m^3 flows needed for successful spawning and rearing in the river (Holmes and Cowan 2014), presumably negatively affecting the steelhead population. As a result of reduced flows, the lagoon to become disconnected from the ocean. These impacts culminated in two CDFW fish rescues in September 2015 that resulted in moving 105 juvenile and one adult steelhead downstream to the lagoon. That winter, only nine steelhead redds were counted in the entire river, significantly below the long-term average (CDFW 2016). Similarly, the Carmel River experienced a sharp decline in steelhead numbers during this time (NMFS 2016). In their five year status review, NMFS concluded that the ongoing hot drought and poor ocean conditions associated with reduced upwelling likely reduced salmonid survival across DPSs and ESUs for listed steelhead and salmon (NMFS 2016).

Another potential impact of climate change are rising sea levels, which may lead to inundation and displacement of estuaries/lagoons. For proper function, estuaries must have intact sandbars and sufficient inflows from the stream during the dry seasons (J. Smith, SJSU, pers.

comm. 2008). Research on CCC steelhead indicates these habitats are critically beneficial to productive steelhead runs, especially in the adjacent DPS to the north (Bond 2006; Hayes 2008. Due to the small size and coastal location of estuaries in the SCCC steelhead DPS, these areas have been subject to intense pressures from human developments, water use, and pollution.

Drought has increased temperatures and natural variability in precipitation (Williams et al. 2015), and reduced natural spawning, rearing, and migration habitat for already low populations across the SCCC steelhead DPS. Williams et al. (2016) found that about climate change impacts on salmonids are increasing over time, suggesting that building resilience in remaining populations will be essential for persistence of steelhead in Southern California. Without resilience of population size, habitat diversity and quantity, and genetic variation, climate change will reduce long-term viability of DPS (NMFS 2016). Providing access, through removal of barriers and restoration of degraded mainstem migration corridors, to a variety of habitats that allows expression of all steelhead life history strategies remains the best approach for building resiliency to climate change in steelhead populations (Wade et al. 2013).

Status Score = 1.9 out of 5.0. Critical Concern. All steelhead populations in Southern California are likely to be extinct within 50 years without serious intervention (Table 3). SCCC steelhead were listed as threatened by NMFS in 1997 and their status was recently reaffirmed (NMFS 2016). They are considered a Sensitive Species by USFS. NMFS (2016) cited threats to a few small populations by loss of accessible habitat, low abundance, degraded estuaries, and altered hydrology; they determined that recovery potential for SCCC steelhead is low to moderate. This finding has not changed since the five-year status review in 2011 (NMFS 2016).

Table 3. Metrics for determining the status of South-Central California Coast steelhead, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	3	Multiple watersheds occupied in small numbers.
Estimated adult abundance	1	Most populations probably contain less than a few dozen spawners, with a total of less than 500 in the entire DPS in recent years.
Intervention dependence	2	Barrier removal, habitat restoration, and updated water management practices are critical to recovery as are restored access to historical spawning, rearing, and refuge habitat and reconnection of resident and anadromous populations.
Tolerance	3	Moderate physiological tolerance, iteroparity uncommon.
Genetic risk	2	While introgression with hatchery rainbow trout is minimal, limited gene flow among populations make them vulnerable.
Climate change	1	Rated highly vulnerable. Effects on small populations documented.
Anthropogenic threats	1	4 High and 2 Medium factors.
Average	1.9	13/7.
Certainty (1-4)	3	Little monitoring of most populations. High confidence that the DPS is in serious decline, low confidence in actual population size.

SCCC steelhead are threatened by loss of freshwater and estuary habitat, increasing human land and water development, poor ocean conditions, and altered hydrology, as well as climate change, wildfires, and drought. These impacts may be insurmountable without coordinated short-and long-term societal and managerial changes. Socially, municipal and county governments will need to focus on restoring aquatic habitats in estuaries and along mainstems and tributaries that flow through residential areas. Best management practices for water use and management must be implemented in cooperation with municipalities, private landowners, agricultural interests, and industrial water users to conserve and restore floodplain and riparian habitats throughout the DPS. NMFS (2013) identified extensive public education, development of cooperative relationships, and interagency collaboration as critical to recovery of SCCC steelhead. These steps are necessary to ensure that funding and strategic planning result in effective, sustained funding and implementation of SCCC steelhead recovery efforts.

Management Recommendations: NMFS (2016) gives the SCCC steelhead DPS only a moderate-to low potential for recovery as a result of the myriad threats that the low numbers of steelhead face throughout their range. Recovery of these populations hinges upon restoring access to high quality habitats throughout their range so they can use their flexible life history strategies to build sustainable populations. Such a strategy should be coordinated among resource management partners and citizens, and be based on promoting sustainable land and water uses, restoring natural river and estuary processes, and building resilience to climate change impacts (NMFS 2013). The most important aspect of such a strategy relates to securing adequate streamflows for all steelhead life stages. Programs that address and expand water conservation, efficiency, and re-use throughout South-Central California will be essential to establishing necessary flows throughout the DPS range. Toward that end, a recent effort to use residential storage tanks in periods of low flow rather than groundwater pumping is gaining traction among private landowners and water users in the Upper Pajaro and Salinas watersheds (Clifford 2016). In order for such programs to function efficiently, there is a need for alternative sources of water in the dry season, incentives for landowners to change their practices, water rights exchanges or purchases, and ease of permitting, funding, and enforcement.

Next, fish passage must be addressed on a broad and coordinated scale to allow juvenile outmigration and adult steelhead to access high quality habitat in the upper main stems and tributaries of core watersheds (NMFS 2013). Because viability of a population increases with population size, restoring habitat access in key watersheds is most likely to help populations meet viability criteria (Boughton et al. 2006, NMFS 2016). The limited number of fish returning to streams within the Interior Coast Range and Carmel Basin regions indicates that mainstem restoration may be necessary for maintaining viability in the DPS. However, habitat for smaller populations is also needed for aid in dispersal and connectivity across the landscape; long-term viability of the DPS depends upon migrants from neighboring basins to maintain metapopulation dynamics and genetic diversity (Garza et al. 2014).

Toward this end, significant habitat restoration has been undertaken in the Carmel River watershed. In 2013, river channel re-routing began, and San Clemente Dam was removed in 2015. Significant habitat restoration, including construction of step-pools to aid adult steelhead migration, has been conducted downstream of the old dam site, restoring access to over 40km of historical spawning and rearing habitat and sediment transport from the upper watershed. Thus, populations of resident rainbow trout currently above the dam will be reconnected to steelhead, creating stability and resilience in this population. In addition, the smaller Old Carmel River Dam will be removed in coming years, opening further habitat for spawning and rearing steelhead (Alberola and Kirschenman 2015). Continued monitoring of sediment and steelhead populations will be critical to adaptively managing the newly accessible habitat. In addition, feasibility studies to remove Los Padres Dam, located 40km further upstream, have begun. Long-term monitoring resources will be required to assess habitat changes after removal of several dams in the watershed and to determine effects on the population (Alberola and Kirschenman 2015).

Downstream, in the Carmel Lagoon, restoration has been ongoing in the past decade to battle poor water quality associated with overdraft of surface and ground water in the basin, which has led to improvements in fish habitat. The California American Water District has been discharging advance-treated wastewater near the lagoon to replenish the groundwater table during the dry season, likely adding 250,000m^3 of water per year to the lagoon for fish habitat. However, a study of CAWD's wastewater found concentrations of chemicals associated with pharmaceuticals and personal care products. Research is ongoing to determine if this well-meaning practice is having a negative impact on steelhead juveniles or lagoon habitat (Alberola and Kirschenman 2015).

In a display common in Southern California, manual breaching of the Carmel River lagoon bar is currently required every year to protect infrastructure and housing encroaching on its margins. A proposed Ecosystem Protective Barrier study is currently being funded by Proposition 84 funds to determine the best way to manage the lagoon to avoid further reduction in habitat quantity and quality for juvenile steelhead and federally-listed California red-legged frog (*Rana draytonii*), western snowy plover (*Charadrius nivosus*), and Smith's blue butterfly (*Euphilotes enoptes smithi*) in the lagoon (Alberola and Kirschenman 2015).

The actions taken on the Carmel River are indicative of the numerous beneficial actions that can be taken now to reduce the threats to SCCC steelhead, including:

- Completing fish barrier removal or passage projects in smaller coastal streams (i.e. Arroyo Grande Creek) and larger interior rivers (Carmel, San Antonio, Nacimiento Rivers) to provide access to historical habitat and increase connectivity among anadromous and resident populations. Management partners can leverage CDFW Code 5937 to maintain fish populations below impoundments in good condition.
- Establishing low flow regimes in the Pajaro, Salinas, and Carmel river systems to support functioning riparian corridors and floodplain habitats to increase the spatial distribution and productivity of SCCC steelhead. Uvas, Pacheco, and Corralitos creeks on the Pajaro River have reservoir or diversion bypass flows to protect steelhead (J. Casagrande, NMFS, pers. comm. 2016). On the Salinas River, adequate flows for steelhead protection have been developed but are currently undergoing re-examination. Last, SWRCB has ordered California-American Water to reduce its overdraft on the Carmel River. A new project, Pure Water Monterey Groundwater Replenishment Project, is nearly approved and will recycle and re-use up to 3,500 acre-feet annually from agricultural or industrial return flows in the Salinas Basin in exchange for a reduction of an equal amount of water from the Carmel Basin. SWRCB currently lacks the oversight and regulatory authority to effectively manage groundwater development, especially on private properties on small tributary streams throughout the DPS. Trout Unlimited's water storage tank programs,

which reduce groundwater pumping and surface diversions from critical tributaries during low flow months of summer and fall are a program that warrants expansion to benefit rearing steelhead (J. Casagrande, NMFS, pers. comm. 2016). This is essential and complementary to the need for SWRCB to establish and enforce minimum instream flow requirements for Monterey and San Luis Obispo County streams for SCCC steelhead, and closely monitor flows throughout the entire DPS (CDFW 2014).

- Training regulatory agencies and biologists to protect stream corridors, facilitate assessment of waste discharges (sediment, pesticides, and other non-point source pollutants), and reduce the filling in, artificial breaching, and draining of estuaries.
- Improving estuarine/lagoon habitat function through regulation of land use practices that degrade water quantity and quality that eventually settles out in lagoons.
- Initiating life cycle monitoring in the Pajaro, Salinas, and Carmel rivers to prioritize and implement recovery actions to support SCCC steelhead. Data collection should be amended and streamlined across Southern California so that the data may be used to reliably estimate adult abundance, smolt production, and survival at sea.
- Capitalizing on ongoing drought to identify and prioritize cool water refugia throughout core watersheds in the DPS for cataloging and protection, and re-thinking core-1/2/3 watersheds throughout the range.
- Updating, funding, and implementing the California Coastal Monitoring Plan (CMP) protocols throughout all core-1 watersheds in the SCCC DPS.
- Completing a Fishery Management and Evaluation Plan (FMEP) that CDFW can implement with NMFS and other management partners.

The continuing increase in human populations in the region, coupled with climate change changing rainfall patterns and increasing water temperatures, means that long term (> 50 years) persistence of SCCC steelhead in most watersheds is not likely without large-scale intervention. Possible exceptions may exist in the larger streams that parallel the Big Sur Coast (e.g., Big Creek, Big Sur River), which still benefit from the summertime cooling effect of ocean proximity.

SOUTHERN CALIFORNIA STEELHEAD
Oncorhynchus mykiss irideus

Critical Concern. Status Score = 1.9 out of 5.0. Southern steelhead populations are in danger of extinction within the next 25-50 years, due to anthropogenic and environmental impacts that threaten recovery. Since its listing as an Endangered Species in 1997, southern steelhead abundance remains precariously low.

Description: Southern steelhead are similar to other steelhead and are distinguished primarily by genetic and physiological differences that reflect their evolutionary history. They also exhibit morphometric differences that distinguish them from other coastal steelhead in California such as longer, more streamlined bodies that facilitate passage more easily in Southern California's characteristic low flow, flashy streams (Bajjaliya et al. 2014).

Taxonomic Relationships: Rainbow trout (*Oncorhynchus mykiss*) historically populated all coastal streams of Southern California with permanent flows, as either resident or anadromous trout, or both. Due to natural events such as fire and debris flows, and more recently due to anthropogenic forces such as urbanization and dam construction, many rainbow trout populations are isolated in remote headwaters of their native basins and exhibit a resident life history. In streams with access to the ocean, anadromous forms are present, which have a complex relationship with the resident forms (see Life History section). Southern California steelhead, or southern steelhead, is our informal name for the anadromous form of the formally designated Southern California Coast Steelhead Distinct Population Segment (DPS). Southern steelhead occurring below man-made or natural barriers were distinguished from resident trout in the Endangered Species Act (ESA) listing, and are under different jurisdictions for purposes of fisheries management although the two forms typically constitute one interbreeding population. Genetic analyses indicate that Southern California steelhead and upstream resident trout form a cluster of related fish from the Santa Maria River south to Baja California. Their distinctiveness presumably relates to their adaptations to the unique environment of Southern California. Similar adaptations are present in south-central California coast steelhead which do not have clear genetic differences from southern steelhead, but there are important ecological and zoological differences between the two regions (Boughton *unpubl. obs.*).

A large-scale population genetics study of rainbow trout in Southern California revealed that some of the resident trout in the southernmost portion of the DPS have hybridized with hatchery fish (Jacobson et al. 2014; Abadia-Cardoso et al. 2016). These wild self-sustaining hybrid trout populations persist above barriers in headwaters of the Los Angeles, San Gabriel, Santa Ana, San Juan, San Diego, and Sweetwater rivers. However, pure native rainbow trout populations of steelhead ancestry still exist at least three locations: 1) the San Luis Rey River in San Diego County, 2) Coldwater Canyon Creek of the Santa Ana River in Riverside County, and 3) San Gabriel River system in Los Angeles County, with the exception of the Iron Fork and Devil's Canyon Creek populations which are mixed lineage. In contrast to the southern DPS, populations in the larger watersheds in the northern part of the DPS (Santa Clara, Ventura, Santa Ynez and Santa Maria rivers) have remained largely un-hybridized.

The extent to which hybrid rainbow trout will reproduce in the wild with steelhead from neighboring watersheds once barriers are removed is unknown, but may increase population viability (Jacobson et al. 2014). Two of these southern-most populations have strikingly low genetic diversity compared to other Southern California populations, suggesting long-term genetic isolation and susceptibility to extirpation (Abadia-Cardoso 2016). These authors proposed that hybrid populations derived from "....hatchery rainbow trout are members of the same species, so some introgression does not necessarily render such small populations less viable than purely native populations . . . In fact, the introduction of some novel genetic diversity from hatchery trout into these small, isolated populations will likely increase heterozygosity, providing more variation to adapt to changing environmental conditions and reduce inbreeding." Therefore, isolated native and hybrid populations should be considered cautiously for use in programs designed to perpetuate the persistence of rainbow trout populations in Southern California (Abadia-Cardoso et al. 2016).

Life History: The ecology of southern California steelhead has not been as thoroughly studied as more northern populations (see Upper Klamath Trinity River winter steelhead account). Distinctive life history patterns of southern steelhead mainly relate to the variable environment in which they evolved and to their opportunistic life history strategies (Sloat and Reeves 2014, Kendall et al. 2015). Southern steelhead are dependent on winter rains to provide upstream passage through seasonally opened estuaries and mainstem river flows providing hydrologic connectivity to upstream tributaries. The reliance on rainstorms for permitting passage through lower portions of watersheds suggests a restricted spawning period for steelhead, with considerable flexibility in timing. Spawning typically occurs between January and May, with a peak in February through mid-April (NMFS 2012), although variation may occur across diverse geographies (M. Capelli, NMFS, pers. comm. 2017).

Three predominant life history patterns have been described for south-central coastal steelhead which are also likely important for southern steelhead: fluvial anadromous, freshwater resident, and lagoon-anadromous (Boughton et al. 2007; Kendall et al. 2015). Juvenile steelhead usually remain in freshwater for one to three years before emigrating to the ocean (Shapovalov and Taft 1954, Moore 1980a, Quinn 2005). Southern steelhead, however, probably spend less time in freshwater during migrations because of inhospitable conditions (low flows, warm temperatures, poor water quality) in the lower reaches of their streams.

Juvenile and adult life history plasticity is characteristic of southern steelhead populations (Kendall et al. 2015). In fluvial anadromous life history, southern steelhead outmigration is dictated by the breaching of lagoon sandbars (physical barriers of sand at the mouth of lagoons), typically between January and June, with a peak from late March through mid-May (NMFS 2012). Ocean swells and high tides can lead to temporary bar breaching during the summer and fall, draining lagoons and allowing juvenile trout to emigrate to the ocean. Observations of adult steelhead appearing in the lower Ventura River following a temporary mouth bar breach from large swells and high tides suggests that movement into fresh water is extremely opportunistic. While barriers may limit access to upstream areas, out-migrating juveniles often originate upstream of such barriers. In below-barrier reaches they can mature and interbreed with anadromous individuals. Perennial habitats are limited in lowland reaches, however, amplifying the significance of lagoons as rearing habitat (Kelley 2008, Hayes and Kocik 2014).

Channel connectivity is critical for steelhead to access spawning areas. It is likely that during dry years the largest steelhead populations historically occurred in streams where upstream spawning and rearing habitats were close to the ocean, such as the short coastal streams along the Santa Barbara coast, the Ventura River, Santa Monica Mountains watersheds, and San Mateo Creek, San Diego County (USFWS 1998). Different size classes of juvenile steelhead use different parts of the available habitat. For example, in one stream, Spina (2003) found young-of-year steelhead preferred water less than 40 cm deep, while age one and two fish preferred deeper water. All three sizes were found mainly at velocities of < 10 cm/sec but this largely reflected habitat availability. In a survey of resident trout in Pauma Creek in San Luis Rey River in San Diego County, trout were proportionally captured most frequently in the complex cascade-pool habitats and to a lesser extent in pools, flatwater and riffles (Barabe, 2013; Barabe and O'Brien, 2013).

The freshwater growth rate of rainbow trout may determine whether juvenile fish out-migrate and become steelhead, or whether they remain as year-round residents. A higher growth rate seems to be necessary for fish to reach sufficient size to undergo smoltification and survive in the marine environment, but may not be the only factor that determines outmigration to the ocean (Ward et al. 1989, Bond 2008, Satterthwaite et al. 2010, Sloat and Osterback 2013); however, the mechanisms for controlling anadromy and residency are in the early stages of investigation (Phillis et al 2016; Pearse et al 2014). For example, juvenile growth rates in Topanga Creek in the Santa Monica Mountains are high enough to produce smolts in one or two years (Dagit et al. 2015) and yield juveniles of sufficient size (> 170 mm) for high marine survival (~ 10%) (Ward et al. 1989). A combination of environmental factors (e.g. growth rate, lipid storage) and genetic factors (*e.g.,* chromosomal structure and DNA sequence heterogeneity) likely co-operate to shift the resident vs. migratory life history behavior in steelhead (Kendall et al. 2015; Pearse et al. 2014; Pearse 2016; Satterthwaite 2012; Beakes 2010). They also alter gene expression profiles and tissue structure while acclimating to saltwater conditions. Partial anadromy is an active area of research to gain insight into underlying environmental and genetic influences. This multigenic trait has important implications for endangered steelhead recovery and fisheries management strategies.

Headwater streams may have limited food resources, resulting in slow growth for juvenile summer steelhead (Boughton et al. 2009) Higher water temperature and lower dissolved oxygen levels may be tolerated for short periods of time if nearby cool refuge areas are available (Boughton, et al. 2015, Sloat and Osterback 2013, Sloat and Reeves 2013). Estuaries are important rearing and gateway environments for steelhead that seasonally migrate between fresh water and the ocean. During winter and spring rains, some steelhead juveniles utilize high river flows to emigrate to the ocean. If conditions permit, they may stay in the lagoon/estuary for further rearing and smoltification for several weeks. River flows sufficient to breach bars on the mouths of lagoons allow steelhead to move into the ocean while adult steelhead use this transient access and flow surge to migrate upstream to spawning areas.

Although steelhead typically use estuaries to acclimate to salt and fresh water and to grow, some steelhead may remain in the estuary or lagoon year-round, then return upstream to spawn. This lagoon-anadromous life history has been observed in South-Central Coast steelhead and provides an alternative for juvenile steelhead to increase their size without exposure to ocean predators. In one reported case in Scott Creek in Central California, estuary-reared steelhead comprised between approximately 85-95 % of returning adults despite being a much lower percentage of the juvenile population (Bond et al. 2008). This hybrid life-history scenario observed in more northern steelhead populations may accelerate steelhead recovery in Southern California.

Because of periodic droughts in Southern California, streams may be inaccessible from the ocean during some years, such that adult steelhead may spend additional years in the ocean before having an opportunity to spawn. The increased growing time in the ocean, plus richer food sources in southern coastal waters, may account for the large size (9+ kg) attained by steelhead in some southern California streams (e.g., the Santa Ynez River). These fish may be 5-6 years old, compared to the typical 4-year old spawners in more northern areas in California. When droughts last multiple years and anadromous steelhead are unable to spawn, the freshwater-resident populations are presumably essential for the long-term viability of populations within some watersheds. Likewise, when catastrophic events (i.e., fires, debris flows) extirpate resident trout from a watershed, the anadromous fish are important for recolonization of the streams. Therefore, the ability to migrate to new habitat patches is key to the species survival, and removal of fish passage barriers and dams in Southern California is a high priority for conservation. See Climate Change section below for further discussion.

Habitat Requirements: Southern steelhead require cool, clear, well-oxygenated water with sufficient food, but they have adapted to living under highly variable environmental conditions. Thus, their physiological tolerances may be broader than other steelhead. In general, southern steelhead seem to tolerate warmer water temperatures than their northern counterparts. Their body temperature and metabolic rate fluctuate with the temperature of the surrounding environment. As temperature increases, their metabolic and feeding rate increases until the temperature approaches an upper threshold of about 25˚C where they stop feeding and/or move to a refuge area, but this response depends on proximity of refuge areas, cover and food availability (Boughton, et al. 2015, Sloat and Osterback 2013, Sloat and Reeves 2013).

Important aquatic environmental factors for steelhead include temperature, dissolved oxygen, salinity and water depth. Temperature and dissolved oxygen levels are two critical parameters which can vary diurnally and seasonally to a significant degree. Estimation of ranges for these parameters (Figure 1) comes from studies in the Santa Monica Mountains (Bell et al. 2011, Bell et al. 2012, Dagit et al. 2015), the Santa Clara River in Ventura County (Sloat and Osterback 2013), Moyle et al. 2008, Myrick and Cech 2000; and others cited below. The ranges have uncertainty because they are based on synthesis of data from diverse studies where upper limits of temperature and dissolved oxygen vary with age, food availability, thermal acclimation status, available refuge areas, and waterbody type.

So Cal Water Quality Ranges – Steelhead/Rainbow Trout Habitat

	Good	Fair	Poor
Temp (°C)	<20	20-25	>25
DO (mg/L)	10-7	7-5	5-2

Figure 1. Southern California steelhead/ rainbow trout water quality tolerance range estimates. Data compilation by S. Jacobson, CalTrout, pers. comm. 2016.

Preferred temperatures of juvenile steelhead are reported as 10-17ºC, but southern steelhead seem to persist in environments that regularly reach temperatures outside this range. For example, Carpanzano (1996) found steelhead juveniles in the Ventura River persisted where temperatures peaked daily at 28ºC. Santa Ynez steelhead trout have been observed at temperatures of 25ºC (SYRTAC 2000). In Sespe Creek, Matthews and Berg (1997) found that trout selected cool seeps in flowing water or areas of pools that had lower temperatures despite associated low oxygen levels. Spina (2007), in contrast, found that thermal refuges were often not available to juvenile southern steelhead but they consistently were able to survive daily temperatures of 17.4-24.8°C. These fish maintained higher body temperatures than reported elsewhere and actively foraged during the day, presumably as a means to support their higher metabolic rates. Dissolved oxygen levels above 5 mg/L are generally regarded as sufficient for survival, and the incipient lethal level of dissolved oxygen for adult and juvenile rainbow trout is approximately 3 mg/L (Matthews and Berg 1997). In other California coastal streams, it seems that a period of rapid growth during spring is sufficient to compensate for slower growth at high temperatures during summer and low temperatures during fall and winter (Hayes et al. 2008, Sogard et al. 2009). In upper reaches of Pauma Creek in the San Luis Rey watershed, temperatures ranged from 6°C in winter to 22°C in summer (Jacobson et al. 2010), with a diurnal swing of 10°C during spring and summer months. This temperature fluctuation in Pauma Creek is noteworthy considering the robust trout population there.

Upper lethal temperatures for salmonids tend to be higher when they have been gradually acclimated to warmer temperatures (Threader and Houston 1983) as would occur in streams in Southern California where water temperatures are warm most of the year. The ability to move into different aquatic microenvironments is likely important for juveniles to survive poor habitat conditions. Topanga Creek is an informative example of a trout stream in the southern DPS where warm-season peak daytime water temperatures regularly exceed 21ºC, and trout seek habitats associated with cooler ground water, although these habitats made up only 16% of the total available habitat (Tobias 2006). Bell et al. (2012) reported daily maximum water temperatures in Topanga Creek were generally less than 22ºC during most of the fall, winter, and spring, but usually exceeded 23ºC for nearly a week each summer in late July and early August. Despite these high summer water temperatures, trout in Topanga Creek grew quickly and adults over 270 mm FL were not uncommon (Bell et al. 2011). Another study (Krug et al. 2012) observed that growth rates of Southern steelhead are similar to those in central coast streams during fall and spring, but are significantly higher in winter, suggesting that increased food availability, reduced thermal stress, or other factors make these flashy productive streams conducive to juvenile growth.

In streams without cold-water refuges, steelhead persist by adopting different strategies. Tobias (2006) found groundwater discharge areas typically had greater surface area, greater depth, and more shelter than other nearby areas. Spina (2007) observed that steelhead preferred such areas even without cool groundwater discharges. Trout densities were inversely correlated with aquatic macrophyte densities, likely due to low dissolved oxygen concentrations in these areas at night and the presence of non-salmonid fish species (Douglas 1995). Headwater areas that are continually fed by spring water may be best locations for long-term survival of juvenile steelhead, especially during extended drought.

Distribution: The southern California steelhead DPS includes all naturally spawned anadromous *O. mykiss* populations below natural and human-made barriers in streams from the Santa Maria River, San Luis Obispo County, California (inclusive) to the Tijuana River on the U.S. - Mexico border. Steelhead are most abundant in the four largest watersheds (Santa Maria, Santa Ynez, Ventura, and Santa Clara rivers) in the northern portion of the DPS. Recent observations and genetic analyses indicate that steelhead are sporadically present in the southernmost watersheds. Adult steelhead have been documented in San Juan Creek, Santa Margarita River, San Luis Rey, and San Mateo Creek in Orange and San Diego counties (Hovey 2004; CDM 2007; Cardno TU, 2013 and references therein; NOAA 2012; Kajtaniak and Downie 2010 and references therein).

The entire range of the DPS spans over 30,000 km^2, has over 41,500 km of stream, a significant amount of which has intermittent flow, and contains more than 22 million people (NMFS 2012).

In contrast to often sporadic presence of steelhead, resident rainbow trout occupy the upper watersheds of most river systems in the southern steelhead DPS range and may be largely responsible for maintaining steelhead runs. Resident rainbow trout may be offspring of either anadromous steelhead or resident trout, although many basins have barriers restricting anadromous adults from reaching headwaters that contain resident trout. Anadromous offspring of resident parents occur with sufficient frequency that landlocked fish derived from steelhead should be considered integral components of steelhead recovery, particularly in Southern California (Courter et al. 2013, Kendall et al. 2015, Abadia-Cardoso 2016).

The southernmost rainbow trout in North America reside in headwaters of the Rio Santo Domingo in Baja California, Mexico and in several watersheds of north-central Mexico (Behnke 2002, Miller 2005; Nielson et al. 1996; Nielsen et al, 1998; Abadia-Cardoso et al. 2014, 2015, 2016). These trout, recognized as *O. m. nelsoni*, are quite distinct genetically but originated from steelhead. The southernmost native rainbow trout population known in the United States resides in the San Luis Rey River watershed (San Diego Co.) (Jacobson et al. 2014; Abadia-Cardoso et al. 2016).

The Southern California Steelhead Recovery Plan distinguishes populations in the Southern California DPS based on their relative intrinsic potential to support an independently viable populations:

> "population viability is more likely achievable by focusing recovery efforts on larger watersheds in each Biogeographic Population Group (BPG) capable of sustaining larger populations, and DPS viability is more likely achievable by focusing on the most widely-dispersed set of such core populations capable of maintaining dispersal connectivity" (NMFS 2012, pg. 7).

The five BPGs recognized by NMFS (2012) are, from north to south: *Conception Coast* (surrounding Santa Barbara), *Monte Arido Highlands* (watersheds including Santa Maria River, Santa Ynez River, Ventura River and Santa Clara River), *Santa Monica Mountains* (coastal creeks surrounding Santa Monica such as Arroyo Sequit, Malibu Creek and Topanga Creek), *Mojave Rim* (coastal rivers surrounding Long Beach containing Los Angeles River, San Gabriel River, and Santa Ana River systems) and *Santa Catalina Gulf Coast* (watersheds in Orange County and San Diego County containing San Juan Creek, San Mateo Creek, Santa Margarita River, San Luis Rey River, San Diego River, Sweetwater River and Tijuana River). Within the five BPGs, 46 southern steelhead populations have been identified (Boughton et al 2007), although over half of the populations have been extirpated (Boughton et al. 2005).

In the Santa Catalina Gulf Coast BPG, resident trout occur in a majority of streams above barriers including Trabuco Creek, San Luis Rey River, San Diego River, Pine Valley Creek and Sweetwater River (Boughton et al. 2007; R. Barabe, CDFW, pers. comm. 2016). There are numerous high country habitats that support wild rainbow trout, although fish passage barriers and modified flows impede or block upstream migration from the ocean to headwater spawning and rearing habitats.

The adjacent Mojave Rim BPG watersheds have dispersed populations of resident trout upstream of barriers in the Los Angeles, San Gabriel, and Santa Ana rivers. Populations in the Mojave Rim in the Upper West Fork, East Fork and North Fork San Gabriel River, as well as

Bear Creek, Iron Fork, Fish Fork and Cattle Canyon tributaries and Coldwater Canyon Creek are native trout of coastal steelhead descent. Significant hatchery introgression is present in Devil's Canyon, Santa Anita Creek and lower Iron Fork tributaries of San Gabriel River; and San Antonio Creek, Fuller Mill and North Fork San Jacinto of the Santa Ana River watershed (Abadia-Cardoso et al 2016).

The northern BPG, Conception Coast, has had steelhead observations in numerous watersheds (Boughton 2005) that are connected seasonally to the ocean. These included populations on Santa Anita Creek, Gaviota Creek, Arroyo Hondo, Goleta Slough Complex, Mission Creek, Montecito Creek, San Ysidro Creek, Romero Creek, Arroyo Paredon, and Carpinteria Creek. Resident trout are present in a number of Conception Coast basins above barriers including Jalama Creek, Tajiguas Creek, Dos Pueblos Canyon, Tecolote Creek, and Rincon Creek (Stoecker and Project 2002).

Trends in Abundance: Southern steelhead have been either significantly depleted or extirpated from rivers and streams in which they historically occurred. There are still remnants of self-sustaining populations in the Santa Ynez, Ventura, Santa Maria, and Santa Clara rivers and Topanga Creek. Episodic runs occur in some watersheds of all BPGs, including Gaviota, Arroyo Honda, Goleta Slough Complex, Mission, Malibu, San Gabriel, and San Mateo creeks and Santa Margarita and San Luis Rey Rivers. Historical runs in this DPS that numbered in the thousands are now reduced to single digits over the past 20 years for the Santa Clara, Ventura, and Santa Ynez rivers and Malibu and Topanga creeks (NMFS 2016). However, accurate counts of annual runs are difficult to quantify in most watersheds (M. Capelli, NMFS, pers. comm. 2017). Indeed, run size estimates made without using standard protocols in Southern California watersheds are likely to be highly subjective and based on sparse and potentially misleading data (Good et al. 2005). For the most part, estimates should be not be viewed as absolute population estimates, but, rather multiple year trends.

Steelhead numbers are naturally highly variable in all Southern California streams. Some of the key factors underlying population dynamics include life-history variability in resident and anadromous individuals, population spatial structure that connects streams with other watersheds, a robust rainbow trout population in the watershed, ocean conditions, estuary integrity, unblocked migratory pathways, and high-quality summer and winter rearing habitats.

In the Santa Maria River, estimates of historical numbers are lacking but southern steelhead have been observed in the mainstem and also in Sisquoc River, one of the Santa Maria's major tributaries (Stoecker 2005). Stoecker (2005) found densities of steelhead to be highest in the South Fork Sisquoc River and lowest in the Lower Sisquoc River. Within the Sisquoc, Stoecker (2005) observed the overall age class distribution from 841 juvenile steelhead to be have 52% 0+ fish, 24% 1+ fish, 17% 2+fish, and 7% 3+fish. In May 1991, 14-25 adult steelhead were observed in the Ventura River (R. Leidy, USEPA, memorandum to B. Harper, USFWS, May 8, 1991), but no steelhead were reported in 1992, and only one pair was reported in 1993 (F. Reynolds, memo. to B. Bolster, October 13, 1993).

In the Santa Ynez River, which probably supported the largest historical runs of southern steelhead, runs may have been as large as 20,000 to 30,000 spawners in the 19th and early 20th centuries (Busby et al. 1996). In 1944, the minimum number of steelhead in the Santa Ynez River was estimated to be 13,000-14,500 fish following a favorable wet period (Good et al. 2005). While 1944 estimates of abundance are the best available for the system, a significant portion of rearing and spawning habitat was already blocked by dams on the Santa Ynez by then.

In 1940, CDFG personnel salvaged more than 525,000 young steelhead trout from pools in the Santa Ynez River as it dried in summer (Shapovalov 1940) which is indicative of the productivity of southern steelhead watersheds during wet periods.

Historical run size estimates on the Ventura River were 4,000-5,000 steelhead; while these estimates were made by CDFW personnel with extensive experience in the watershed, and when the fishery was closely regulated, with daily bag limits (Good et al. 2005). Steelhead runs in the Matilija Creek basin (Ventura River) were around 2,000 fish (Good et al. 2005), though this included hatchery raised rainbow trout that were intended to support a put-and-take summer fishery, not to enhance the steelhead fishery (M. Capelli, NMFS, pers. comm. 2017). In the Santa Clara River, historical runs have been estimated at 7,000-9,000 adult fish (Moore, 1980b, Good et al. 2005).

The Santa Clara River is one of the largest watersheds in Southern California (*ca.* 4,100 km^2) so it was presumably once capable of supporting large numbers of steelhead. Anecdotal accounts chronicle a precipitous decline in run sizes during the 1940s and 1950s, presumably due to the combination of drought, angling, urbanization and associated water use, and dam construction. The Santa Clara River drainage now also supports only a small fraction of its historical steelhead populations, the result of 129 natural and human-made fish migration barriers (Good et al. 2005). The Vern Freeman (VF) Diversion Dam alone blocks access to 99% of the watershed (Good et al. 2005). The VF Diversion Dam is downstream of the major southern steelhead spawning tributaries Piru and Sespe Creeks. Sespe Creek provides a large amount of high quality habitat. Although smaller than the above drainages, Santa Paula Creek provides some of the highest quality habitat in the watershed (NMFS 2008, Stoecker and Kelley 2005). Fish passage barrier removal efforts are underway in the Santa Clara River and Santa Paula Creek tributaries and are in the design discussion stages.

Further south, Topanga Creek in the Santa Monica Mountains offers an example of high population variability, and the value of persistence in monitoring to document and understand population dynamics through time. Some streams where steelhead had been extirpated in the Santa Monica Mountains were re-colonized in the late 1990s, and have persisted to date in low numbers (Dagit et al. 2015). In 2006, there was an observed die-off of both native and alien species in Malibu Creek, followed by re-colonization in 2007 and record numbers of adult steelhead (five fish > 50cm) and over 2,200 young-of-year (YOY) fish < 10 cm observed in 2008 (Dagit et al. 2009). Population monitoring was established by Santa Monica Mountains RCD (SMMRCD) and includes population and size surveys, redd counts, a Life Cycle Monitoring Station, and DIDSON sonar detection since 2001 (Topanga Creek) and 2005 (Malibu Creek). In 2015, SMMRCD performed mark-recapture studies and PIT tagged 104 fish to track movement. The highly variable abundance of such streams is reflected in estimates of YOY observed recently, with only 11 observed in 2005, and a peak of 590 observed in 2008, 92 in 2013, 25 in 2014, and 112 in 2015 during extended drought (Dagit et al. 2016). Monthly population surveys performed in SM Bay by snorkel survey over the past 10 years showed distinct population fluctuations in Topanga and Malibu Creeks (Figure 2).

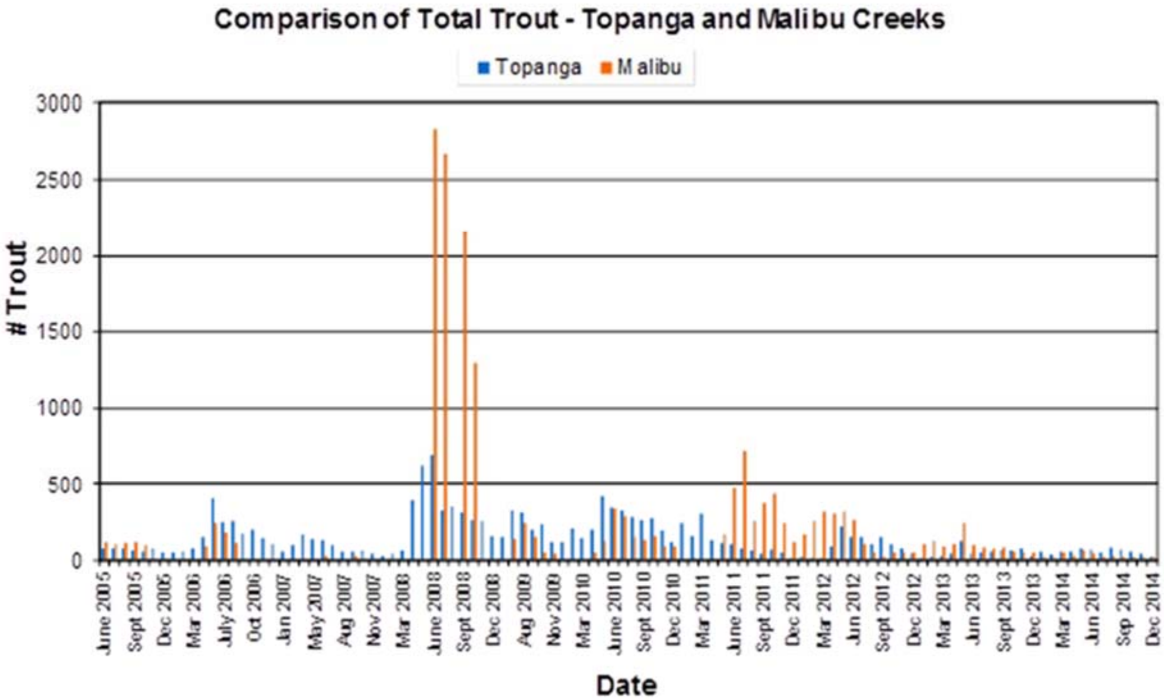


Figure 2. Rainbow trout population estimates from snorkel surveys in Topanga and Malibu creeks (SMMRCD 2015, Figure 3, pg. 8).

Overall, southern steelhead numbers have declined dramatically from estimated annual runs totaling a minimum of 30,000 adults in most years to significantly less than 500 returning adult fish combined annually in the past 50-75 years. There have been few comprehensive surveys conducted in recent years to provide a reliable estimate of total population size for southern steelhead. However, a recent compilation of monitoring data revealed strikingly low number of anadromous adults on several river systems. The data, however, are incomplete because sampling was conducted in small portions of respective watersheds and/or because of the difficulties associated with sampling during high discharge events, or because of other technical or environmental constraints. As such, the numbers represent raw counts or observations during certain periods and are not adjusted for observational errors or biases. Between 2001 and 2011, sampling on two primary tributaries to the Santa Ynez River and part of the mainstem below Bradbury dam showed an average annual adult return of 3.4 adults (SD = 5.2, Figure 3) and a mean smolt capture of 146 (SD = 116) (NMFS 2016). On the Ventura River, a mean return of 2.5 adults was observed at the fish passage facility at the Robles Diversion Dam between 2006 and 2009 and 0-2 adults were observed annually at the Freeman diversion dam fish ladder on the Santa Clara River between 1995 and 2009 (NMFS 2016). On Topanga Creek, observations ranged from 0 and 4 adults per year from 2001-2010, suggesting that the largest annual run would have been approximately 40 individuals under an assumption of a 10% observation probability (NMFS 2016). Finally, on Malibu Creek, a total of 11 anadromous adults were observed by snorkel survey between 2012 and 2015 (Dagit 2016 from NMFS 2016).

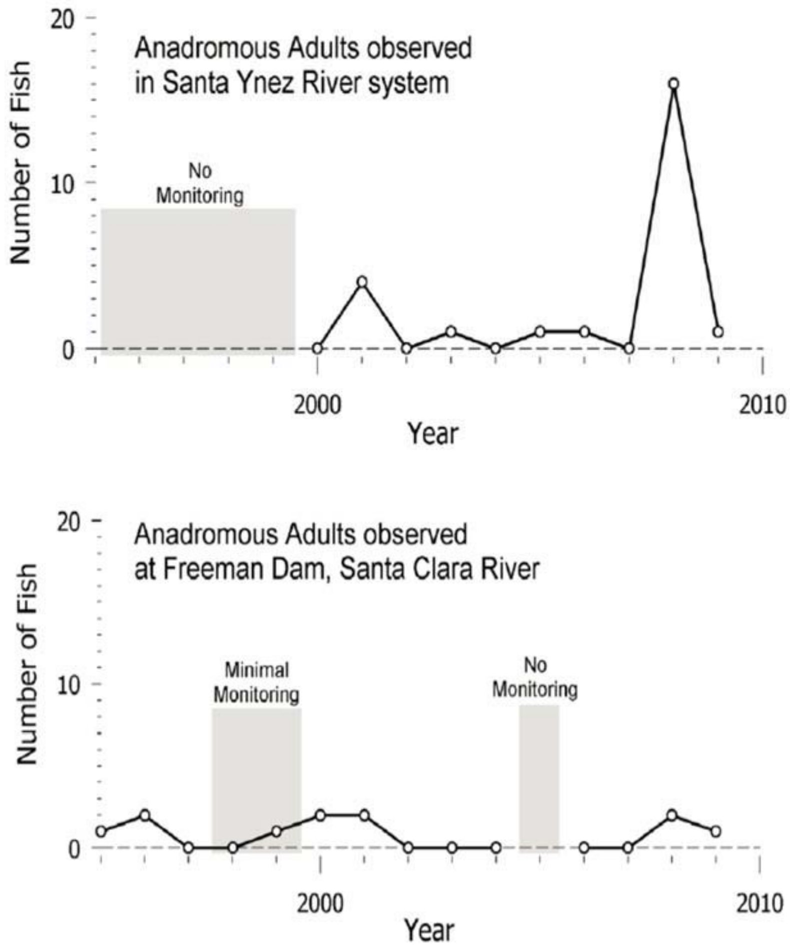


Figure 3. Anadromous adult *O. mykiss* in the Santa Ynez and Santa Clara Rivers, 1995-2010. Observations are incomplete counts unadjusted for observation probability. Williams 2011, Fig. 15, p. 74).

With steelhead abundance in Southern California extremely low, monitoring is critical to assess progress towards recovery goals. Monitoring is difficult due to patchy spawner distribution and large stretches of uninhabited water. With the recent focus statewide on protocol consistency, sample frame approaches and technology development, the Coastal Salmonid Monitoring Program (CMP) led by California Department of Fish and Wildlife should increasingly yield meaningful steelhead abundance data. Fish Bulletin 180 lays out the Viable Salmonid Population (VSP) conceptual framework designed to gain information on diversity of steelhead life history traits including abundance, productivity, spatial structure, diversity, and movement (Adams et al. 2011, HDR Engineering 2013). In contrast to Northern California with more easily measureable annual steelhead runs, the Southern California VSP analysis focuses on fixed counting stations and in-stream counting. As of 2016, there are currently weirs/dams and/or sonar cameras to quantify adult escapement on the Santa Ynez River, Arroyo Hondo, Ventura and Santa Clara rivers, as well as Topanga and Malibu creeks (D. McCanne, CDFW, pers. comm. 2016; NMFS 2016). Some of these include Fixed Life Cycle Monitoring Stations at Santa Ynez River, Topanga Creek and Malibu Creek.

Factors Affecting Status: The Southern California Steelhead Recovery Plan identified primary threats to southern steelhead viability, as summarized below (NMFS 2012).

Dams. The majority of spawning and rearing habitat for steelhead within the major river systems has been rendered inaccessible as a result of dams, debris basins, road crossings, and other in-stream structures which block or impede migration of adult steelhead to headwater spawning and rearing tributaries, as well as restricting the emigration of juveniles to the ocean (Stoecker and Project 2002, NMFS 2012). For example, Twitchell Dam eliminates access to half of the Santa Maria River's historically accessible habitat and dries up the river below it; water diversions further reduce connectivity among critical lower watershed tributaries (i.e., Sisquoc River) and the estuary. Bradbury Dam, which creates Cachuma Reservoir, is the largest barrier on the Santa Ynez River and operations restrict flows necessary to support suitable steelhead habitat (NMFS 2016). On the Ventura River, access to most upstream habitat is blocked by Matilija Dam on Matilija Creek and Casitas Dam on Coyote Creek. Rindge Dam on Malibu Creek blocks access to most upstream habitat. Diversion dams and poorly functioning fish ladders on the Santa Clara River have blocked steelhead access to spawning habitats and reduced available rearing habitat for steelhead offspring. Overall, fish passage barriers are also common in Southern California watersheds from bridge crossings with concrete and rock stabilization structures that create impassible drop structures for in-migrating steelhead.

Agriculture. Agriculture is a diminishing problem as farmland continues to be overtaken by urban development. Nevertheless, much of the early development of water in the region was to support irrigated agriculture for high value crops such as orchards of citrus and avocado and row crops of strawberries and vegetables. Agriculture continues to be important in some areas such as the lower Santa Maria and Santa Ynez Rivers, the small coastal watersheds along the Santa Barbara coast, the upper Ventura River and Ojai Basin tributaries, and portions of the San Mateo, Creek, San Luis Rey and San Dieguito River in the southernmost portion of the DPS. Agricultural water demands in the Santa Clara watershed have also had a significant impact on mainstem flows and fish passage (NMFS 2016). Agricultural development impacted steelhead habitat in the past by reducing habitat complexity, competing for water to irrigate crops, and producing runoff with nutrients that decrease water quality. The nutrients can also accumulate in estuaries that limit their utility as a growth and acclimation area for steelhead. Current efforts are focused on working with agricultural operations to stagger water consumption (both daily and seasonally) to promote steelhead viability, and implement Best Management Practices to reduce nutrient run-off and meet quantitative water quality objectives set by regional water quality control boards. Further initiatives to upgrade irrigation efficiency infrastructure will provide a foundation for community based water quality awareness and water conservation.

Urbanization. Over 20 million people live in Southern California, in a semi-arid environment where water is a limiting resource. Not surprisingly, most watersheds containing southern steelhead are urbanized. In particular, the four largest watersheds are heavily impacted by water diversions (both surface and subsurface), which reduce stream flows, and development of the floodplain and associated riparian corridor for agricultural, residential, industrial, and sand and gravel extraction uses. There has been extensive loss of steelhead populations, especially south of Malibu Creek, due to dewatering and channelization of rivers and creeks. Urban and rural waste discharges are also widespread, which degrades water quality and creates habitat conditions that favor alien aquatic organisms. However, the single biggest impact remains the demand on surface water and groundwater resources, and related water supply infrastructures such as dams, diversions, and groundwater wells. In some parts of watersheds with shallow aquifers, groundwater and surface water systems are tightly linked, such that depleted groundwater resources decrease surface flow, which can have immediate negative impacts on steelhead populations.

The Southern California population boom has led to increasing demand on surface water and groundwater resources which has depleted natural water storage. Typically 80-90% of water in Southern California is imported from Northern California, Owens River or the Colorado River (SDIRWMP 2013). To better meet the increasing demands for water, a robust effort to diversify and conserve regional water supplies is underway. However, altered river hydrology and stream flow is a widespread concern by decreasing natural flow or creating perennial flow in ephemeral streams. Due to the naturally ephemeral and flashy nature of Southern California streams, it is difficult to re-establish natural hydrologic regimes that benefit steelhead. Diversion of water and increases in non-permeable surfaces (e.g., roads, parking lots) have made the hydrograph show broader extremes in many streams, with flashier winter flows and lower summer flows, greatly reducing habitat quality and quantity. The problem is further exacerbated by upstream dams which inhibit sediment transport. Lack of flushing flows encourages sediment accumulation and can negatively affect spawning gravels and food sources.

Another major impact of urbanization is stream alteration through channelization, road crossings construction, stream bank stabilization, and flood-control levee maintenance. Floodplain development and construction of flood control structures and activities have also altered natural fluvial processes and reduced instream and riparian habitat. Increases in residential structures (and associated roads) on steep-sided slopes have accelerated erosion and sedimentation of some river and stream channels.

Mining. The principal ongoing mining in the region is drilling for and pumping of oil; the region is still a major oil producer. Waste from drilling and oil spills from pipelines are common problems but there is no evidence of direct impacts on steelhead. Hardrock mines, for gold, gems, and other minerals are common in the mountains along the coast but mostly abandoned. Aggregate mining in stream beds still occurs in Southern California, especially in the Santa Maria, Santa Ynez, Santa Clara Rives, Tujunga Wash, and streams that flow out of the San Bernardino and San Gabriel mountains; some of the mining operations have been limited or modified to provide better instream habitat protection, but many operate under old, outdated permits (Mount 1995). Removal of sediment disrupts river processes which presumably affects steelhead migration, spawning, and rearing habitat.

Fire. Periodic wildfires are an integral ecological feature of Southern California and helped shape the landscape and life history strategies of southern steelhead. Wildfires can increase wet-season runoff, reduce summertime surface flows, and increase stream temperatures (Boughton et al. 2007). When wildfires are followed by heavy rains in areas which are geomorphically unstable, high flows may cause an increase in sediment delivery to streams via debris torrents (Spina and Tormey 2000, Keeley 2006), that cover habitats and fish alike. Following a wildfire, if winter rains do not mobilize sediment but increase runoff, then favorable characteristics such as increased scour and nutrients may benefit steelhead. Because of increased frequency of drought, the severity and presumably frequency of wildfires is increasing in Southern California, making it more difficult for steelhead to persist in some watersheds. Emergency plans are either in place or under development by fisheries resource managers to react quickly to protect sensitive populations that are threatened by drought and fire.

Estuarine alteration. Southern steelhead are likely similar to South-Central Coast steelhead in their use of estuaries, which are essential for juvenile rearing, adult migration, and occasionally adult over summering (Kelley 2008, Bond et al. 2008, NMFS 2012). Many Southern California estuaries/ lagoons have disappeared due to human activities, while others are functionally degraded (Ferren et al. 1995; Lafferty 2005). Many are shallower, warmer, and more saline than they were originally, due to urbanization, channelization, and altered stream and sediment flows. Southern California estuaries also suffer from pollution, invasive riparian and aquatic vegetation, and filling to create uplands. Smaller lagoons along the rugged Gaviota Coast and Santa Monica Mountains are less disturbed than the estuaries associated with larger rivers, due to less upstream development. The degradation of remaining estuarine habitat as a result of both point- and non-point sources of pollution and artificial breaching of sandbars has reduced the suitability of these habitats for steelhead rearing and as transition zones between marine and freshwater environments.

Recreation. Southern California's streams are magnets for recreational use, especially at higher elevations where natural flows are maintained. The effects of aquatic recreation on steelhead are not well documented, but if it is fairly localized they may not be great. Recreational users could be an additional source of stress to juvenile steelhead that are living in water close their upper temperature tolerances, potentially resulting in sublethal impacts such as changes in behavior or even mortality.

Harvest. The once popular fishery for southern California steelhead closed as fish numbers were depleted and prohibited once the fish was declared an endangered species. However, adult steelhead in streams are highly vulnerable to poaching and removal of even a few individuals in some years can be a threat to the population.

Hatcheries. Hatchery rainbow trout have been planted in virtually all Southern California streams. These plantings were comprised predominantly of hatchery raised rainbow trout, and were intended to support a put-and-take summer fishery, not to enhance expression of the anadromous steelhead life history (M. Capelli, NMFS, pers. comm. 2017). Genetic studies show that fish of hatchery origin dominate rainbow trout populations in the more urbanized southernmost part of their range where the expression of anadromy may have been less frequently expressed because of the restricted hydrological connection between the ocean and upstream spawning and rearing habitats, although several substantial wild native populations still exist (Abadia-Cardoso et al. 2016). Stocking of hatchery rainbow trout has presumably resulted in the displacement or hybridization of native trout and steelhead from most of these waters. Where the expression of anadromy is most frequently observed in the northernmost populations of steelhead and trout (e.g. Santa Clara, Ventura, and Santa Inez rivers), populations show minimal introgression with hatchery fish, maintaining both native resident and steelhead life histories.

Alien species. The presence of invasive vegetation such as giant reed (*Arundo donax*), tamarisk, and Pampas grass have disrupted river morphology, creating monotypic riparian habitats that consume more water. These aggressive growers are highly disruptive and expensive to eradicate. However, the past ten years has seen a concerted effort to control them. The presence of non-native aquatic species such as bass, catfish, shiners, carp, bullfrogs and crayfish is pervasive in Southern California streams. These competitors thrive in warmer, slow moving water and can withstand lower water quality, dissolved oxygen levels and high sediment. Once non-native aquatic species are dominant, depletion and eradication requires long-term control programs. Although habitat may exist for southern steelhead in some watersheds from which they are currently missing, the presence of non-native fishes makes reestablishment of steelhead in these basins difficult. Designated wild trout streams such as Sespe Creek have in recent years been colonized by a host of non-native species, which have significantly degraded this wild trout fishery and impacted the important refugia habitat for rearing juvenile steelhead within the Santa Clara River watershed (M. Capelli, NMFS, pers. comm. 2017).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Southern steelhead. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	High	Many rivers in the DPS have major dams blocking passage.
Agriculture	Medium	Irrigation diversions in many streams reduce flows, especially during drought.
Grazing	Low	Some grazing on public lands, often for fire or weed control.
Rural /residential development	Medium	Many homes along streams throughout basins, outside of major urban areas. Effects included in urbanization.
Urbanization	High	Many important streams flow though highly urbanized areas across Southern California, which may overdraw water from surface and groundwater sources.
Instream mining	Medium	Gravel mining in some watersheds reduces habitat.
Mining	Low	Gold mining and dredging may increase siltation and negatively impact eggs and juveniles, but impacts largely unknown.
Transportation	Medium	Roads and railroads line streams, creating sediment and erosion throughout most basins, and highway crossings create migration barriers. Effects included in urbanization.
Logging	Low	Impacts are likely due mainly to legacy effects.
Fire	High	Fires in Southern California are very common and can cause siltation of streams, loss of riparian habitat, and increased water temperatures.
Estuarine alteration	High	Nearly all estuaries and lagoons highly altered with reduced rearing habitat.
Recreation	Medium	Recreational use common in streams; effects on steelhead unknown.
Harvest	Medium	Poaching may impact small populations.
Hatcheries	Low	Many populations introgressed with hatchery-origin rainbow trout in the southern portion of the DPS, but stocking has been curtailed in anadromous waters.
Alien species	High	Most watersheds significantly invaded by alien aquatic species.

Effects of Climate Change: Moyle et al. (2013) rated southern steelhead "critically vulnerable" to climate change, likely to go extinct by 2100 without strong conservation measures. Climate change is predicted to increase variability in rainfall, stream flow, and temperatures, reducing suitable stretches of stream, even when flowing, for steelhead/rainbow trout. In general, regions with lower latitude and elevation will be subject to the greatest increase in duration and intensity of higher air and water temperatures (Wade et al. 2013). The southern steelhead DPS, which lies at the southern edge of the species' range, is on the front line of climate impacts. As a result, southern steelhead will be exposed to periods of higher water temperature and flow variability, possibly outpacing their ability to avoid such exposure (Wade et al. 2013).

Longer and more severe droughts are also predicted in the future due to climate change, enhancing competition for water between a large and growing human population and a small and shrinking trout population in Southern California. Drought has increased temperatures and natural variability in precipitation (Williams et al. 2015; NMFS 2016), and reduced natural spawning, rearing, and migration habitat for already small populations. The ongoing "hot drought" in California has been characterized by several of the hottest years on record in the last five years (2012-2016) alone. No part of the state was impacted more by drought than Southern California, with significant reductions in precipitation over the last five years compared to long term averages and much of the region remaining in critical drought. For example, Lake Cachuma, which provides drinking and irrigation water to the City of Santa Barbara, holds less than 9% of its capacity as of this writing (January 2017, Santa Barbara County Flood Control District 2017). A hotter, drier climate regime in Southern California is expected in the future, and is likely to increase the frequency and magnitude of catastrophic wildfires in California. In their five year status review, NMFS (2016) concluded that the ongoing hot drought and poor ocean conditions associated with reduced upwelling likely reduced salmonid survival across DPSs and ESUs for listed steelhead and salmon. Another potential impact of climate change is sea level rise, which may lead to inundation and displacement of estuaries/lagoons. Reduction in these important rearing habitats will likely reduce smolt survival and further strain populations of southern steelhead.

Given the precarious position of the southern steelhead life history, there will be an increase in dependence on resident rainbow trout above manmade barriers to produce out-migrants that become steelhead. Unfortunately, climate change will likely create conditions that will select against the steelhead life history in Southern California. Maintenance of steelhead runs may therefore require artificial means from enhancing flows during critical periods, to fish rescues, to employing conservation hatcheries. Williams et al. (2016) found that climate change impacts on salmonids are increasing over time, suggesting that building resilience in remaining populations will be essential for persistence of steelhead in Southern California. Without resilience of population size, habitat diversity and quantity, and genetic variation, climate change will reduce long-term viability of DPSs (NMFS 2016). Providing access and connectivity among populations, through removal of barriers and restoration of degraded mainstem migration corridors, to a variety of habitats that allows expression of all steelhead life history strategies remains the best approach for building resiliency to climate change in steelhead populations (Wade et al. 2013).

Status Score = 1.9 out of 5.0. Critical Concern. Southern steelhead populations are in danger of extinction within the next 25-50 years due to stresses associated with the growing human population of Southern California and climate change (Table 2). They were listed as an endangered species under the Endangered Species Act (ESA) in 1997 (62 FR43937), and abundance remains precariously low. Southern steelhead initially included steelhead populations from the Santa Maria River (Ventura Co.) system south to Malibu Creek (Los Angeles Co.). The range of the listed steelhead was extended southward to the U.S. – Mexico Border in 2002 (67 FR21586). A final listing determination was issued in 2006 for the Southern California Coastal Steelhead Distinct Population Segment (DPS), which encompasses all naturally-spawned anadromous rainbow trout in the listing area whose freshwater habitat occurs below artificial or natural impassible upstream barriers, as well as rainbow trout above impassable barriers that can emigrate into water below barriers and exhibit an anadromous life-history. The National Marine Fisheries Service (5-Year Status Review) summarizes recovery progress to date and recommends that the Southern California Coastal Steelhead DPS remain listed as an endangered species with no changes to the geographic boundaries of the recovery area (NMFS 2016).

NMFS (2012) concluded there is moderate potential for recovery of southern steelhead. If resident rainbow trout populations are considered part of the southern steelhead complex, then the extinction threat of the overall population is somewhat less. Reconnecting the anadromous and resident forms of the native *O. mykiss* populations, however, is essential for maintaining both the anadromous and resident trout populations in the future.

Table 2. Metrics for determining the status of southern steelhead. The range is as follows: 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	3	Found in most of native range, if scattered; mainly in 4 northernmost rivers, but also in a number of short coastal watersheds in the central and southern portion of the DPS
Estimated adult abundance	1	Highly variable but with general downward trend and very low abundance in all populations.
Intervention dependence	2	Intensive efforts such as barrier modification, habitat restoration, and restoration of instream flows are essential.
Environmental tolerance	3	Moderate physiological tolerance to existing conditions, although limits are being reached.
Genetic risk	2	Limited gene flow among populations; some hatchery hybridization. Populations very small and geographically isolated.
Climate change	1	Climate change impacts likely throughout their range, exacerbating other factors.
Anthropogenic threats	1	5 High, 6 Medium, 4 Low factors.
Average	1.9	13/7.
Certainty (1-4)	3	Moderate.

Management Recommendations: The NMFS Recovery Plan for southern steelhead defines biological viability criteria for individual populations and for the DPS as a whole, providing quantitative benchmarks for recovery. A viable population is defined as a population having a negligible (< 5%) risk of extinction over a 100-year time frame. Therefore, a viable DPS must be comprised of a number of viable populations widely distributed throughout the DPS but sufficiently well-connected through ocean and freshwater dispersal to maintain long-term persistence and evolutionary potential. Population-level viability criteria apply to core populations in all of the geographic regions. These criteria include population characteristics such as mean annual run-size, persistence during varying ocean conditions, spawner density, and the anadromous fraction of the individual populations. For the entire DPS, viability criteria identify a minimum number of populations which must be restored to viability and the minimum spatial distribution between populations in each BPG: Monte Arido (4), Conception Coast (3), Santa Monica Mountains (2), Mojave River (3), and Santa Catalina Gulf Coast (8). Recovery efforts are further supported by designated critical habitat which encompasses 1,133 km (708 miles) of stream habitat within 32 watersheds.

To achieve the overarching goals of preventing extinction of the Southern California steelhead and ultimately de-listing this species, the Southern California Steelhead Recovery Plan outlines the following six objectives:

1. Protect existing populations and habitats.
2. Maintain and restore distribution to previously occupied areas that are essential for recovery.
3. Increase abundance of steelhead to viable population levels, including the expression of all life history forms and strategies.
4. Conserve existing genetic diversity and provide opportunities for interchange of genetic material between and within meta-populations.
5. Maintain and restore suitable habitat conditions and characteristics for all life history stages so that viable populations can be sustained.
6. Conduct research and monitoring necessary to refine and demonstrate attainment of recovery criteria.

Additional actions could be added to further help recovery: i) declare native resident rainbow trout populations part of the DPS and then manage them with steelhead as a threatened/endangered species, and ii) continue to utilize triploid (sterile) hatchery rainbow trout in Southern California streams where such fish can reach anadromous waters.

Successfully meeting these objectives will require protection and expansion of habitat for steelhead within each of the five biogeographic regions and reestablishment of large runs in streams that historically were highly productive for steelhead (i.e., Santa Maria, Santa Ynez, Ventura, Santa Clara River, and other rivers to the south). Restoration efforts focused at the watershed scale will ensure adequate flows and passage to historical spawning and rearing areas. It is important to provide connectivity among populations in different streams to allow exchange of genetic material. A key part of this connectivity is ensuring anadromous and resident populations in each watershed can overlap and interbreed during most years. Removal of the artificial distinction between resident rainbow trout and steelhead and manage them together in the DPS, with a special emphasis on expanding the anadromous life history would be consistent with the recovery strategy identified in NMFS Recovery Plan.

Restoration/reconciliation. Changes in water management are critical to restoring habitats and hydro-geomorphic processes important to southern steelhead. Water removal from streams now containing critically low numbers of steelhead should be restricted in order to leave minimum flows for fish in streams. The environmental impact of future development should be carefully evaluated and appropriate alternate measures reviewed by state and federal regulatory agencies (e.g., CDFW, NMFS) prior to accepting mitigation. Restoration techniques that increase habitat fairly rapidly for southern steelhead may include groundwater recharge projects, removal of barriers in watersheds with high quality habitat, and enhancement of instream and riparian habitats. Recycled water from treatment plants may provide an important means by which to recharge streams and groundwater. The effective allocation of recycled water could be instrumental for maintaining migration corridors later into the spring and rearing habitat for juveniles during the fall and early winter in lower reaches of Southern California streams.

Involvement of local citizens is crucial for steelhead recovery as well. Coalitions are already operational for Santa Clara River and San Diego/Orange County priority watersheds. These coalitions engage over 30 federal, state, and local agencies, tribal nations, resource conservation districts, and non-governmental organizations including California Trout to implement the Southern California Steelhead Recovery Plan. Many extant southern steelhead populations are on public lands, and effective management of these waters and cooperation by state and federal managers is needed to benefit these populations.

Dam removal and reoperation. Dams and fish passage facilities provide numerous opportunities for restoring southern steelhead into portions of watersheds with optimal spawning and rearing habitat. In many cases, resident trout persist upstream of these barriers. Considerable planning has gone into removal of Matilija Dam on Matilija Creek (Ventura River), and Rindge Dam on Malibu Creek, as well as construction of fish passage facilities (Robles Diversion Dam) on the Ventura River. Implementation of these projects should be more expeditious in order to benefit southern steelhead as soon as possible. Evaluation of fish passage barrier removal and remediation and associated water operation facilities in the Cuyama, Santa Ynez, Santa Margarita, San Juan Creek and San Luis Rey Rivers should be completed and implemented to reconnect freshwater and marine habitats. Dams on southern steelhead streams, such as Bradbury Dam on the Santa Ynez River and Twitchell Dam on the Cuyama River, can be operated more effectively to reestablish flows during periods critical for steelhead survival, especially during migrations and periods when fish are rearing in estuaries and lower river reaches.

Monitoring. In the short term, monitoring data will ensure that trends and restoration progress are quantifiable and understandable. Development of a baseline monitoring plan for steelhead and their habitat in Southern California watersheds is an essential task. Integration with existing Coastal Salmonid Monitoring frameworks in place in Northern California will contribute important information on steelhead biology and management, and will aid in decision making and coordinating recovery actions.

Conservation Hatcheries. The benefits and guidelines of instituting regional conservation hatcheries are presented in the NMFS (2012) Recovery Plan. Conservation hatcheries are fundamentally different in structure and purpose from large scale hatcheries designed to generate high numbers of trout for stocking waters for sport-fishing. The conservation hatchery is a limited-scope program that aims to conserve and propagate steelhead taken from the wild for conservation purposes, and return the progeny to their native habitats to mature and reproduce naturally. This type of hatchery is typically an outgrowth of an emergency translocation of trout that are facing extirpation from a catastrophic event such as drought, debris flows and/or wildfire.

The goals of operating a potential southern steelhead conservation hatchery are to preserve remaining genotypic and phenotypic characteristics, reduce short-term risk of extinction, reintroduce populations into restored watersheds, conduct research on Southern California stock relevant to species conservation, and incorporate scientific and management considerations into southern steelhead recovery. Within this context, it is imperative that protocols and rationale are in place prior to establishing a conservation hatchery.

TROUT

BULL TROUT
Salvelinus confluentus

Extinct in California. Status Score = 0.0 out of 5.0. Bull trout are extinct in their range in California; the last known individual was captured on the McCloud River in 1975. Bull trout were likely in decline for most of the 20th century prior to their eventual extirpation.

Description: The bull trout is a large, piscivorous salmonid that typically thrives in habitats with abundant juvenile salmonids such as salmon or steelhead fry or parr (Brenkman et al. 2007). It has fine scales (110 along the lateral line) and has white or cream leading edges on the pelvic, pectoral and anal fins. Live fish are olive green in color with tiny yellowish spots on the back and small red spots on the sides. The body usually has a few yellow spots at the base of the tail. In adults, the head is broad, flat between the eyes, and long, making up more than 25 percent of the body length in adults. The eyes are close to the top of the head to suit its ambush stalking tactics. The mouth is large, with conspicuous sharp teeth; the maxillary bone of the upper jaw extends beyond the eye. The lower jaw has a fleshy nob at its tip that fits into a notch on the top of the upper jaw (between the premaxillary bones). The adipose fin is large, making up about 50–85 percent of the depth of the caudal peduncle. For McCloud River fish, the branchiostegal rays numbered 13-15 per side; the mandibular pores, 7-9 per side; and the gill rakers, 15-18 per arch, with visible teeth on the anterior margin of each (Cavender 1997).

Taxonomic Relationships: Bull trout are part of the Arctic char (*Salvelinus alpinus*) complex, which includes species of the genus *Salvelinus* such as the Eastern Brook "trout." Bull trout were once considered to be the same species as Dolly Varden char (*S. malma*), a largely anadromous coastal species, and all three species have been lumped into once great taxon (Behnke 2002). However, studies by Cavender (1978), Hass and McPhail (1991), and others have eliminated doubts about their distinctiveness and species status (Behnke 2002). Museum specimens of California bull trout are distinct morphologically from other populations (Moyle 2002), so if they were still present they would probably be treated as an Evolutionary Significant Unit (ESU) confined to California.

Life History: Bull trout in California were largely unstudied until they became extinct (Wales 1939, Sturgess and Moyle 1978, Rode 1990) and the information summarized here is from other regions, as presented by Moyle 2002. Bull trout can be a) adfluvial, where adults live in lakes and spawn and rear in streams, b) fluvial, where all stages live in streams, but adults migrate up tributaries for spawning), c) stream resident fish with no separation of life history stages, or d) anadromous, where immature fish and adults undertake repeated estuary and/or ocean migrations to attain large sizes and seek more abundant sources of prey (Brenkman, Corbett, and Volk 2007). Most resident populations remain in small streams, and it is possible that many, if not all, are remnants of once-fluvial populations (e.g., populations in Klamath Basin tributaries in Oregon). The population was apparently once fluvial in the McCloud River, with adults concentrating in pools in the lower reaches of the river, migrating upstream to spawn in higher-gradient reaches below Lower Falls (Rode 1990).

Juvenile bull trout (<11 cm TL) feed heavily on aquatic insects, and fish gradually become more important in their diet as they grow larger. Bull trout more than 25 cm TL are primarily piscivorous, with juvenile trout, salmon, sculpins, and their own young making up the bulk of the diet. Frogs, snakes, mice, and ducklings have also been found in their stomachs. Bull trout are ambush predators, typically lying in wait underneath a log or ledge and opportunistically grabbing passing fish. High bull trout densities are often associated with concentrations of small fish from migratory populations. Presumably, Chinook salmon eggs and juveniles rearing in the McCloud River year-round were once a major source of food for bull trout.

Bull trout grow slowly and are long-lived for salmonids, with lifespans up to 20 years. As a result, they are capable of achieving large sizes. They typically reach 5-8 cm TL in their first year, 10-14 cm in their second, and 15-20 cm in their third. Growth is slowest afterward in resident populations, and fastest in adfluvial populations, where individuals may reach 40-45 cm TL in 5-6 years. The largest bull trout on record, from Lake Pend Oreille, ID, measured 103 cm TL (14.5 kg). Bull trout from the McCloud River were purported to reach over 7.3 kg (ca. 70 cm TL), and the California angling record is a fish from McCloud Reservoir that weighed about 5.1 kg. A fish that lived for 19 years in the Mt. Shasta hatchery weighed around 6 kg at the time of death. The last two bull trout caught from the McCloud River (in 1975) measured 37 cm SL and 42 cm SL and were 4-6 years old (Sturgess and Moyle 1978).

Bull trout spawn for the first time in their fourth or fifth year at lengths of 40 cm TL or more. Fish from resident populations spawn at smaller sizes (25-30 cm TL) and presumably younger ages. They usually migrate upstream to spawn in gravel riffles of clear, cold streams. Migrations of 150-250 km are not unusual in adfluvial populations where passage facilitates such movement. Spawning migrations can begin in July or August, but spawning does not begin until water temperatures have dropped below 9-10°C in early fall (September and October) in the McCloud River. Female spawners choose sites that have relatively low gradients, expanses of loose gravel, groundwater or spring inflow, and nearby cover, such as pools. Spawning behavior is similar to that of brook trout, although males may spawn with multiple females. Small jack males are present among the spawners as well. Each female, depending on size, lays 1,000-12,000 eggs. Embryos are buried at a depth of 10-20 cm and hatch in 100-145 days. They remain in the gravel for another 65-90 days, absorbing their yolk sacs. They begin feeding while still in the interstices of the gravel and emerge at 23-28 mm TL to fill their air bladders, usually in April or May. Young-of-year bull trout spend much of their first summer along stream edges or in backwaters until they reach about 50 mm TL, when they move out into faster and deeper water to seek greater foraging opportunities.

Habitat Requirements: According to Moyle (2002) streams containing bull trout require exceptionally cold, clear water, often originating from springs. They are rarely found in streams that have maximum temperatures greater than 18°C, though Behnke (2002) suggests 15°C is the upper thermal tolerance. Optimum temperatures appear to be 12-14°C (10-12°C according to

Behnke 2002) for adults and juveniles, and 4-6°C for embryo incubation. Prior to the construction of McCloud Dam, the McCloud River provided near-ideal temperatures for bull trout, with its major source (Big Springs) flowing in at 7.5°C year-round and temperatures in the lower river rarely exceeding 13°C during the summer (Rode 1990). The river also had other characteristics favorable to bull trout: good spawning and rearing habitat below Lower Falls, deep pools in the lower river for adults, and abundant prey in the form of juvenile Chinook salmon.

Adult bull trout in rivers and smaller streams prefer to live on the bottom in deep pools. Adfluvial populations thrive in large coldwater lakes and reservoirs (e.g., Flathead Lake and Hungry Horse Reservoir, MT). In California, bull trout populations in both McCloud and Shasta reservoirs were unable to maintain themselves. Juvenile trout up to 20 cm TL are strongly bottom-oriented, hanging out near or under large rocks and large woody debris, in stream reaches with coarse, silt-free substrates. They seem to prefer pockets of slow water near faster-moving water that can deliver food. As they grow larger, they move into pools.

Distribution: In California, bull trout were found in nearly 100 km of the McCloud River (Shasta and Siskiyou counties), from its mouth to Lower Falls (Rode 1990). They may also have occurred in spring-fed tributaries to the Upper Sacramento and Pit Rivers, but records are lacking. According to Moyle (2002), this was historically the southernmost distribution of the species. Today the southernmost populations are found in the Jarbridge River, NV, and small streams in the upper Klamath Basin, OR. The northernmost populations are found in the headwaters of the Yukon River, British Columbia. The easternmost populations are found in Columbia River tributaries in Alberta and Montana. They are widely scattered in the Columbia River system, in the headwaters of coastal rivers of British Columbia, and in interior drainages of British Columbia and Alberta (Saskatchewan, Athabasca, and Peace Rivers). The presence of many disjointed populations at present indicates a wider distribution in the Pleistocene period, under wetter and cooler conditions than exist now.

Trends in Abundance: Bull trout are extinct in California. The last known bull trout caught in California was captured by University of California, Davis graduate student Jamie Sturgess in 1975, by hook and line. It was tagged and released but not seen again. Bull trout were apparently in decline throughout most of the 20th century, although in the 1930s they still supported a small fishery in the McCloud River (Wales 1939). By the 1950s, after the construction of Shasta Dam, they were scarce (Rode 1990). They became increasingly rare in the 1960s and were likely extirpated by the late 1970s.

Nature and Degree of Threats: Habitat degradation and fragmentation associated with the construction of Shasta and McCloud reservoirs, coupled with non-native trout introductions, were the principal causes of bull trout decline in California. According to Moyle (2002, p. 299-300) the factors that resulted in their extirpation from California are:

Depletion of salmon. In the 19th century, the McCloud River supported at least two runs of Chinook salmon, (probably) a small run of coho salmon, and a steelhead run. Salmon carcasses and juveniles likely supported fairly large bull trout populations at that time. The 19th-century Sacramento River commercial fishery, combined with sediments from hydraulic mining, severely depleted salmon runs in the McCloud River. The Baird Hatchery, established on the lower McCloud River in 1874 to take eggs from Chinook salmon in order to help restore depleted runs, may ironically have contributed to the decline of McCloud River salmon because the weir at the hatchery blocked much of the run at times. In the early 20th century, Chinook salmon recovered somewhat, but not to former levels. In 1942, Shasta Dam blocked access for all salmon upstream and passage of bull trout downstream. Salmon were a major driving force in the McCloud River ecosystem, so their depletion and loss undoubtedly had a major impact on the piscivores in the river, including bull trout.

Introduction of brook trout. Brook trout were established in the McCloud River watershed by about 1910. They are now present in small tributaries that juvenile bull trout may once have used for rearing. Brook trout will hybridize with bull trout, and this is a major cause of the decline of resident populations in Oregon and elsewhere. There is no evidence that hybridization took place in the McCloud River.

Introduction of brown trout. Brown trout probably entered the McCloud River in the 1920s, although they may not have been abundant until after Shasta Reservoir was created in the 1940s. The reservoir allowed a substantial migratory population of large brown trout to develop. Large brown trout are ecologically similar to bull trout, hanging out in large pools and preying on other fish. They may have contributed to bull trout decline through a combination of competition and predation. A recent study suggests that under increasing warming associated with climate change, brown trout are replacing bull trout in waters with temperatures greater than 12°C across Montana; perhaps under warmer temperature regimes, brown trout were better able to outcompete and displace the native bull trout in the McCloud River (Al-Chokhachy et al. 2016).

Shasta Dam and Reservoir. Construction of Shasta Dam in 1942 and the creation of Shasta Reservoir blocked access of major salmon runs, provided better habitat for migratory brown trout, and flooded about 26 km of the lower McCloud River (nearly a quarter of the bull trout's habitat). Although fluvial bull trout elsewhere have become adfluvial following construction of reservoirs, this did not happen with Shasta Reservoir. Small numbers of bull trout were documented in the reservoir fishery, but runs from the reservoir back into the McCloud River never developed. Presumably, the reservoir was too warm for growth and survival (Rode 1990).

McCloud Dam and Reservoir. McCloud Dam was completed in 1965 and blocked the river about 45 km upstream from Shasta Reservoir. This was the final blow to bull trout in California. The McCloud Reservoir flooded 8 km of prime habitat for bull trout, severed the connection between juvenile and adult habitats by blocking adult migrations to upstream areas, and altered conditions downstream of the dam, reducing flows, reducing recruitment of spawning gravel, reducing the frequency of flushing flows, increasing turbidity in the fall, and, most importantly, raising water temperatures in the river by 5-10°C (Rode 1990). After completion of the dam, long-lived bull trout survived for 10-12 years before becoming extirpated.

Status Score = 0.0 out of 5.0. Extinct in California. Bull trout are extinct in California and are listed by the USFWS (1999) as threatened throughout the rest of their range in the United States.

Effects of Climate Change: The persistence of bull trout in California despite warm summer temperatures was likely due to the springs that maintain cool flows in the McCloud River, even during severe drought. Increases in air temperature or reductions in snowpack during prolonged drought may reduce available wetted habitat, as was seen in during the 2012-16 drought in the McCloud/Mount Shasta region. Under a changing climate in California, shifts in precipitation patterns that favor more frequent and prolonged drought and more intense, if infrequent storms, are likely to be exacerbated in the future, to the detriment of coldwater fish species such as bull trout that rely on snowmelt and springs during warm and dry summer months. In a recent study, Kovach et al. (2015) found negative correlations between maximum summer temperatures and decreased allelic richness and diversity in bull trout across their range in the Columbia River basin, suggesting that those populations most at risk are the least able to deal with a changing climate due to low diversity and ability to shift their range. Therefore, remaining bull trout populations in North America are likely very susceptible to negative impacts of climate change, such as displacement by invasive brown trout and further restriction by thermal barriers (Al-Chokhachy et al. 2016).

Management Recommendations: CDFW had a plan for restoring bull trout through establishing resident populations in some tributaries upstream of McCloud Reservoir and in the lower river (Rode 1990). These populations would be supplemented by hatchery fish if they could not sustain themselves, which is likely. Attempts at introducing fish from the Klamath River Basin in Oregon to the McCloud River have failed, and additional attempts are unlikely unless the best source populations recover their former abundance (Rode 1990).

Presumably if McCloud Dam was removed or re-operated to produce colder water downstream through increased flows, a plan could be re-implemented for reintroduction of adfluvial bull trout. However, because Shasta Dam blocks access to spawning salmon, the abundance of prey is much lower than it was historically, so the river is likely not able to support a self-sustaining population of bull trout, especially in the face of competition and predation from brown trout. The Bureau of Reclamation and National Marine Fisheries Service (NMFS) are considering trapping and trucking adult and juvenile salmon around Shasta Dam to re-populate the McCloud River, though serious questions over the costs and effectiveness of such programs remain (Lusardi and Moyle *In review*). If this were to happen, perhaps talk of re-introducing bull trout to the McCloud River would resume.

CALIFORNIA GOLDEN TROUT
Oncorhynchus mykiss aguabonita (Jordan)

Critical Concern. Status Score = 1.9 out of 5.0. California Golden trout are likely to become extinct in the wild in the next 50-100 years. While the Golden Trout Creek (GTC) population is relatively secure, the South Fork Kern River (SFKR) population is threatened by introgression with rainbow trout and predation and competition from introduced brown trout.

Description: The California golden trout is named for its bright colors. Behnke (2002) describes their coloration as follows: "The color of the back is brassy or copper, becoming bright golden yellow just above the lateral line. A deep red stripe runs along the lateral line and the golden yellow body color intensifies below. A deep crimson color suffuses the ventral region from the anal fin to beneath the lower jaw… (p. 105)." Fish from GTC are particularly brightly colored. Young and most adults have about 10 parr marks centered along the lateral line. The parr marks on adults are considered to be a distinctive characteristic (Needham and Gard 1959), but they are not always present, especially in larger fish from introduced lake populations. Large spots are present, mostly on the dorsal and caudal fins and on the caudal peduncle. The pectoral, pelvic, and anal fins are orange to yellow. The anal, dorsal, and pelvic fins have white to yellow tips, preceded by a black band. Basibranchial teeth are absent and there are 17-21 gill rakers. Other characteristics include 175-210 scales along the lateral line, 34-45 scales above the lateral line, 8-10 pelvic rays, 25-40 pyloric caeca, and 58-61 vertebrae (Schreck and Behnke 1971).

Taxonomic Relationships: The complex history of golden trout taxonomy and nomenclature is reported in Behnke (2002) and is presented here in a simplified version. Originally, three species of golden trout were described from the upper Kern River basin: *Salmo aguabonita* from the SFKR, *S. whitei* from the Little Kern River, and *S. roosevelti* from GTC. However, the first two forms were eventually recognized as subspecies of *S. aguabonita*: *S. a. aguabonita* and *S. a. whitei*. *S. roosevelti* was shown to be a color variant of *S. a. aguabonita* (Moyle 2002). Berg (1987) concluded that the two recognized subspecies of golden trout are more closely related to the Kern River rainbow trout (*O. m. gilberti*) than either are to each other. However, Bagley and Gall (1998) and M. Stephens (2007), using improved genetic techniques, found that California golden trout and Little Kern golden trout represent two independent lineages derived from coastal rainbow trout. *O. m. aguabonita* is referred to in some lists as South Fork Kern golden trout or as Volcano Creek golden trout but California golden trout seems more appropriate, given its status as the official state freshwater fish of California.

The American Fisheries Society (Page et al. 2013) gives the official common name as just Golden Trout, which is considered a full species, *O. aguabonita.* The reasoning behind the species designation is that the golden trout is "recognized as a species by L. M. Page and B. M. Burr (2011) and herein because of lack of evidence of intergradation with *O. mykiss* (p. 206)." Presumably, Page et al. (2013) were thinking in terms of *natural* intergradation because interbreeding with introduced rainbow trout is a major problem for all golden trout (see below). However, natural interbreeding may have resulted in the Kern River rainbow trout, which has genetic characteristics of both species (Moyle 2002). An additional problem is that Little Kern golden trout should, under this classification scheme, be considered a separate species as well because they are a separate lineage from California golden trout. We therefore continue to favor treating the three forms as separate subspecies (lineages) of rainbow trout.

Life History: California golden trout live in cold, clear alpine streams. They have comparatively slow growth rates due to the truncated growing season and low productivity of high elevation streams in their native range (Knapp and Dudley 1990, Knapp and Matthews 1996). In streams, they are usually 3-4 cm SL at the end of their first summer of life, 7-8 cm SL at the end of their second summer, 10-11 cm SL at the end of their third summer, and grow 1-2 cm per year thereafter; they reach a maximum size of 19-20 cm SL and a maximum age of 9 years (Knapp and Dudley 1990). In alpine lakes, individuals from introduced populations grow to 4-5 cm FL, 10-15 cm FL, 13-23 cm FL, and 21-28 cm FL at the end of their first through fourth years, respectively (Curtis 1934); they can reach 35-43 cm FL by the seventh year. The largest on record from California weighed 4.5 kg, from Virginia Lake, Madera County, in 1952. However, most records of golden trout growth in lakes are suspect because populations were established from introductions that may have been hybridized with rainbow trout.

Golden trout spawn when they are three or four years old, when water temperatures exceed 10°C, with daily maximums of 16-18°C, in late June and July (Stefferud 1993; Knapp and Vredenburg 1996). Average daily temperatures for spawning are around 7-10°C and spawning occurs in gravel riffles in streams. Spawning behavior is typical of other members of the rainbow trout group, although golden trout spawn successfully in finer substrates (decomposed granite) more than most other trout (Knapp and Vredenburg 1996). Females produce 300-2,300 eggs, depending on body size (Curtis 1934). Embryos hatch within 20 days at an incubation temperature of 14°C. Fry emerge from the gravel two to three weeks after hatching, at which time they are about 25 mm TL. In introduced lake populations, fry move into lakes from spawning streams when they are about 45 mm TL.

In streams, golden trout are active at all times of day and night but tend to stay in the same areas for long periods of time (Matthews 1996a). They feed on both terrestrial and aquatic invertebrates, mostly adult and larval insects, taking whatever is most abundant. In lakes, they feed mainly on benthic invertebrates, especially midge pupae (*Chironomidae*) (T. Armstrong, UC Davis, unpubl. data). Although bright coloration makes them highly visible, there are very few natural predators in their range (Moyle 2002). Their tendency to be more active during the day than most trout also suggests low predation. Thus, their bright coloration may have evolved for reproductive advantage. However, bright coloration has also been implicated as providing camouflage against the bright colors of the volcanic substrates in the clear, shallow streams within their range (Needham and Gard 1959). When these trout are removed from mountainous streams and brought down to low elevation streams, they may lose their brightness and take on dull gray and red colors (Needham and Gard 1959). In lakes, they become paler in color, often appearing silvery.

Habitat Requirements: Golden trout evolved in streams of the southern Sierra Nevada, at elevations above 2,300 m. The valleys of the Kern Plateau are broad, flat, and filled with glacial alluvium, which results in wide meadows through which streams meander. These streams are small, shallow, and have only limited riparian vegetation along the edges. The exposed nature of the streams California golden trout inhabit is largely the result of heavy grazing of livestock on a fragile landscape, which began in the 1860s. Grazing causes compaction of soils, collapse of stream banks, and elimination of riparian plant cover (Odion et al. 1988, Knapp and Matthews 1996, Matthews 1996b). Stream bottoms are mostly volcanic sand and gravel, with some cobble. The water is clear and mostly cold, although summer temperatures can fluctuate from 3 to 20°C (Knapp and Dudley 1990). California golden trout generally prefer pool habitat and congregate near emergent sedges and undercut banks (Matthews 1996a). Environmental tolerances are presumably similar to those of coastal rainbow trout.

Distribution: California golden trout are native to the SFKR, which flows into Isabella Reservoir, and to GTC (including its tributary, Volcano Creek), which flows into the Kern River (Berg 1987). Initially (1909 and earlier), California golden trout were collected from GTC and transported north by pack train, extending their range by some 160 km by 1914 (Fisk 1969). They were also translocated into many other waters within and outside California, including Cottonwood Lakes, not far from the headwaters of GTC, and headwaters of the SFKR, such as Mulkey Creek (Stephens et al. 2004). Cottonwood Lakes served as a source of golden trout eggs for stocking other waters beginning in 1917. (Stephens et al. 2004). As a result of stocking in California, fish considered to be golden trout are now found in more than 300 high mountain lakes and 1,100 km of streams outside their native range (Fisk 1969), including Ash Meadows, Diaz Creek, and the Owens River drainage (Stephens et al. 2013). Many of these transplanted populations have hybridized with rainbow trout, including the golden trout from Cottonwood Lakes that have been used as brood stock for transplants (Moyle 2002, Stephens et al. 2004). California golden trout have also been introduced in lakes and streams in the Rocky Mountains, and in various ranges in Utah and Wyoming. However, most populations are also likely hybridized with either rainbow or cutthroat trout, although some populations in Wyoming generally show low levels of introgression (Stephens et al. 2013). Most out-of-basin transplants, however, show limited amounts of genetic diversity (Stephens et al. 2013). Some unhybridized populations apparently still exist from early transplants in the Sierra Nevada, but these too appear to have limited genetic diversity due to small numbers used to establish these populations (Stephens and May 2011).

Trends in Abundance: California golden trout populations suffered major declines during the 19^{th} and first half of the 20^{th} century from overfishing and heavy grazing. Invading brown trout displaced California golden trout, including hybrids, from all reaches below artificial barriers, so golden trout are now confined to a few kilometers of stream in the GTC watershed and in the South Fork Kern watershed. Within their native range, California golden trout occur at both low densities (0.02 - 0.17 fish per m^2 in streams) (Knapp and Dudley 1990) and at high densities (1.3-2.7 fish per m^2). Low densities are most likely to be in found in grazed reaches of stream with little cover and food, with some exceptions (see next paragraph). Presumably, densities were much higher, on average, before livestock began grazing the drainage. Although California golden trout were widely introduced outside their native range during the 19^{th} and 20^{th} centuries, the introduced populations should not be regarded as contributing to conservation because most (if not all) have hybridized with introduced coastal rainbow trout.

Knapp and Dudley (1990) estimated that golden trout streams typically support 8-52 fish/100 m of stream, although a recent estimate for Mulkey Creek, a tributary to the SFKR which supports an introduced population, was 472 fish/100m (Carmona-Catot and Weaver 2006). If the Knapp and Dudley figures are accurate, in 1965, when the first major CDFW habitat management plan was issued (CDFG 1965), there would have been 2,400-15,600 individuals in GTC (30 km) and 4,000-26,000 in the South Fork Kern (50 km). Curiously, the high numbers in the SFKR are found in reaches that have been degraded by grazing, presumably because the reaches contain decomposed granite substrates that are used for spawning (Knapp et al. 1998, S.

Stephens, CDFW, pers. comm. 2008). Lack of cover in these reaches selects for smaller fish, which are more numerous, but may have lower fecundity due to small body.

At present, if unhybridized fish exist only in 5 km of Volcano Creek, then there are only 400-2600 'pure' golden trout left in their native range, a decrease of at least 95% from historical numbers. The percentage of these fish that reproduce every year is unknown but likely to be small. A caveat on this very rough calculation is that it is based on genetic studies (Stephens et al. 2004) that show many fish that are counted as hybrids have a very low incidence of 'foreign' genes; thus it may not be necessary to eliminate all rainbow trout genes from introgressed populations through eradication, if there is no impact on phenotypes. If golden trout populations with phenotypes that show low introgression of rainbow trout genes are considered to have conservation value, then the numbers of golden trout would be considerably higher and might include fish both within and outside their native range as well. For example, the introduced population in Mulkey Creek may be as large as 40,000 fish (>75 mm FL) in roughly 10 km of habitat, with very low levels of introgression (2%; Stephens 2007). Nevertheless, because golden trout had been eliminated through hybridization and predation from most of the lower SFKR by 1965, where populations would have been most dense, the 95 percent decline figure for the native range may still be valid, even if populations with low introgression are counted.

California golden trout in the upper SFKR and GTC are introgressed with non-native rainbow trout, but introgression levels are markedly different in these two streams. Nearly all SFKR trout are introgressed with rainbow trout to some degree. There is also a cline of introgression from the lower Kennedy Meadows area (94%) upstream to the headwaters (2%). Such a pattern is reflective of stocking in the lower river with upstream movement of introgressed or rainbow trout through time (Stephens et al. 2013). Kennedy Meadows also contains dense populations of brown trout. In many reaches of GTC, however, levels of introgression are low. Recent work by Stephens et al. (2013) suggests that several populations within GTC show less than 5% introgression. Nevertheless, genetically 'pure' populations exist in only a few km of streams and will continue to do so for the short term (<5 yrs.).

Overall, unhybridized California golden trout are much less abundant than they have been in the past in their native range. In areas where they still persist, numbers may be higher than they were in the days of heavy harvest and grazing, but these numbers are still presumably less than historical highs (pre-1800s) because of the continued presence of hybridized fish, grazing, and other human impacts. More recently, drought in California has decreased habitat availability to the point that the CDFW determined a drought rescue was necessary. Forty-two California golden trout were taken from Volcano Creek by CDFW staff on September 19th, 2016 due to concerns about low population levels, the ongoing drought, and potential decrease in habitat quality. Fish were placed into separate raceways based on natal streams at the American River Hatchery to be reared through the winter and spring until their habitat is suitable for reintroduction, perhaps in the late summer or early fall, depending on habitat evaluation. Genetic samples will be taken from all fish to determine diversity in the population. Once CDFW completes a broad habitat assessment to determine adequate conditions exist for the species, the fish will be released back into the wild. Fortunately, these fish were kept in raceways that were unaffected by warm, turbid waters in the hatchery that forced hatchery staff to relocate and release tens of thousands of juvenile Chinook salmon and rainbow trout (J. Weaver, CDFW, pers. comm. 2016).

Factors Affecting Status: The principal threats to California golden trout are interactions with alien trout species, followed by grazing.

Alien species. The major threats from alien species are hybridization with rainbow trout and competition and predation from brown and rainbow trout. There is a long history of planting rainbow trout in the upper Kern River basin to improve recreational angling. The peak of stocking was probably 1931-1941, when 85,000-100,000 rainbows were planted every year (Gold and Gold 1976). Stocking of hatchery rainbows in the SFKR at Kennedy Meadow occurred until 2008 (B. Beal, CDFW, pers. comm. 2012). This portion of the SFKR also supports a fishery for wild brown trout. In addition, golden trout were introduced in Cottonwood Lakes in 1891, with a subsequent egg-taking station established by 1918; this population, the source of most golden trout transplants to other watersheds, was apparently contaminated with rainbow trout fairly early in its history. While Pister (2010) refers to the Cottonwood Lakes population as remaining "physically very attractive," a comment on their phenotypic similarly to California golden trout, the Cottonwood Lakes population is highly introgressed and likely offers little conservation value (Cordes et al. 2006).

In the SFKR, brown trout were eliminated, using piscicides, from headwaters in the early 1980s and Ramshaw, Templeton and Schaeffer barriers were constructed to prevent their reinvasion. Rainbow trout, however, were able to move upstream over the deteriorated Schaeffer Fish Barrier to the Templeton Fish Barrier. Brown trout still dominate about 780 km of stream in the SFKR basin (Stephens et al. 2004). Recent electrofishing data conducted by the Department of Fish and Wildlife at multiple reaches on the SFKR estimated an average of 775 brown trout/km, compared to 181 California golden trout/km within the same reaches (DFW 2009).

Recent efforts have focused on the removal of brown trout, but not introgressed California golden trout. Hybridized trout have been found upstream of the Templeton Barrier all the way to the headwaters of the SFKR. When these events occurred is not known because the original barriers have been replaced with improved structures. Improved structures, however, do not minimize the threat of introgression without the simultaneous removal of hybridized populations upstream of barriers. This combination of events has resulted in rainbow trout or rainbow trout-golden trout hybrids invading most streams in the native range of California golden trout in the SFKR and hybridizing with non-introgressed individuals (Cordes et al. 2006). By the early 1990s, both Templeton and Schaeffer fish barriers had deteriorated and the Schaeffer Barrier allowed upstream fish passage. Both barriers were replaced with substantial concrete structures in 1996 and 2003, respectively. In these reaches, golden-type trout (goldens of varying degrees of hybridization) coexist with both brown trout and native Sacramento sucker (Carmona-Catot and Weaver 2006), although the long-term viability of this assemblage is not known. In GTC, hybridization affects only a small percentage (about 5%) of the trout and many of these populations represent the highest conservation priority (Stephens et al. 2013). In the SFKR basin, only a few headwater populations may have escaped hybridization (Cordes et al. 2006), while in Volcano Creek and some smaller tributaries most populations show extremely low levels or no signature of introgression. Both GTC and SFKR populations show some reduction in genetic diversity, but Stephens et al. (2013) suggested that levels of genetic diversity in most native range populations is not of primary concern.

Most places where golden trout have been planted outside their native range have also likely been planted with rainbow trout, or the golden trout actually originated from hybridized stocks (i.e., Cottonwood Lakes). Hybridization with rainbow trout results in fish that are likely to be less brightly-colored than native golden trout. The rainbow trout phenotype eventually becomes dominant, so the fish look more like rainbow trout. This has been well demonstrated in the lower SFKR, where hatchery rainbow trout had been planted annually from the 1930s until the late 2000s. The few wild golden trout left are heavily hybridized, having a rainbow-like appearance. After 2004, only sterile triploid rainbow trout were stocked in the lower SFKR, with stocking entirely discontinued in 2008. Hybridization can ultimately result not only in the loss of the uniquely colored variety of trout, but also loss of genetic material that reflects adaptations to the distinctive environment of the upper Kern River basin. However, it is possible that populations with a low frequency of rainbow trout alleles (genes) may be able to retain golden trout coloration, a high degree of genetic fitness, and adaptability to their habitats.

In addition to threats from rainbow trout, predation and competition from introduced brown trout are a continuous threat. In 1993, CDFW biologists found a reproducing population of brown trout above the lowermost barrier (Schaeffer) and a population was also found in Strawberry Creek in 2003 (S. Stephens et al. 2004). How they arrived there is not known, but it would have been relatively easy for anglers to move fish over the barrier. While barriers that prevent fish from migrating upstream can eliminate or reduce gene flow among golden trout, they may be the only solution to preventing additional upstream movement of alien trout. Construction of an additional barrier is possible near Dutch John Flat, upstream of Kennedy Meadows, to create an additional isolated area (B. Beal, CDFW, pers. comm. 2012).

Grazing. Livestock grazing is permitted in designated Wilderness Areas, such as the Golden Trout Wilderness Area; grazing occurs around GTC and the SFKR where California golden trout reside. According to the USFWS (October 11, 2011, 76 FR 63094), about 95 percent of areas around golden trout streams have been grazed by livestock for 130 years. Not surprisingly, some sections of stream and entire meadows have been severely damaged. The negative effects of grazing at all levels in the fragile meadow systems of this region have been well documented (Knapp and Matthews 1996, Matthews 1996b). Grazing impacts to instream and riparian habitats include: reducing the amount of streamside vegetation, collapsing banks, making streams wider and shallower, reducing bank undercutting, polluting waters with feces and urine, increasing temperatures, increased siltation in spawning beds (smothering embryos), and generally making habitats less complex and suitable for trout. These impacts may result in declines in trout populations.

Levels of cattle grazing have been reduced in recent years and the USFS has adopted guidelines to allow heavily grazed areas to recover (USFWS October 11, 2011, 76 FR 63094). Two of the four grazing allotments on the Kern Plateau have been rested since 2001 (S. Stephens et al. 2004). Future management of grazing for the four allotments is being considered by the USFS with a decision concerning grazing yet to be determined (as of 2016). Herbst et al. (2012) showed that eliminating grazing in the Golden Trout Wilderness meadows resulted in improved streambank structure and macroinvertebrate diversity, while fencing short sections of stream did not have the same effect. The authors suggest that removal of grazing at broader spatial scales (i.e., meadow) was more effective in achieving ecosystem and biotic recovery than small, reach-scale improvements. Such broad spatial habitat improvements are likely to be reflected in larger, more robust golden trout populations. The grazing allotment decision and the future enforcement of improved grazing practices will have major impacts on the health of golden trout populations in their native range.

Recreation. Although California golden trout waters are entirely within Sequoia and Inyo National Forests and largely within the Golden Trout Wilderness, they are still impacted by human activities, including off-road vehicles (in the lower portions of the SFKR) and recreational damage by hikers, horse riders and pack stock. A particular threat is off-road vehicle use in the vicinity of Monache Meadows and the severe degradation of the lower SFKR due to multiple causes throughout that area.

Harvest. Recreational fishing within the Golden Trout Wilderness is allowed from the last Saturday in April through November 15, is restricted to artificial lures with barbless hooks, and a five fish daily bag and possession limit is allowed. Harvest rates are unknown, but are presumably low due to the remote nature of most golden trout-bearing streams, along with shifts in angler preference toward catch-and-release fishing, particularly for native or unique forms of trout with limited distributions.

Hatcheries. Golden trout, usually partially hybridized, are still raised in hatcheries for the purpose of supporting recreational fisheries, but these fish are not planted in the native range.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of California golden trout. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	n/a	All major dams outside native range of California golden trout.
Agriculture	n/a	
Grazing	Medium	Ongoing threat but greatly reduced from the past.
Rural /residential development	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	Historical mines are present but have no known impacts.
Transportation	Low	Trails and off-road vehicle routes can be a source of sediment and pollution input into streams; direct habitat impacts from wet route crossings.
Logging	Low	This is an important land use in the broader region but probably has no direct effect on golden trout streams.
Fire	Low	Because of fire suppression, headwater areas could be impacted by hot fires, although this is unlikely given sparse plant communities in region.
Estuary alteration	n/a	
Recreation	Low	Pure populations within the GTC watershed are entirely within designated wilderness; South Fork populations with conservation value are also within designated wilderness.
Harvest	Low	Potential impact; light pressure, mostly catch and release.
Hatcheries	Low	Residual effects of hybridization with hatchery fish.

Alien species	High	Major cause of limited distribution in South Fork Kern; however, very limited introgression with rainbow trout and no brown trout in waters within GTC watershed.

Effects of Climate Change: The major predicted impacts of climate change in the Sierra Nevada are reduction in snow pack, increased likelihood of rain-on-snow events, and shifts in peak runoff from late spring/early summer months to late winter/early spring months due to warmer temperatures. This will have the least effect in the southern Sierra Nevada because the mountain elevations are highest there and may continue to retain a great deal of snow. Thus, snow melt is likely to maintain flows in golden trout streams. Nevertheless, snow pack may not persist as long in the extensive meadows of the Kern Plateau and meadows are likely to become drier by the end of summer, with reduced base flows in streams. Elimination of grazing and other activities that compact meadows (reducing their ability to store water) and reduce riparian cover and shade can mitigate, in part, for the effects of climate change. Temperatures are likely to increase earlier in the season in golden trout streams and it is possible that spawning times may become earlier, with unknown consequences. Moyle et al. (2013) rated California golden trout as "critically vulnerable" to climate change, indicating that extirpation from its native range is likely by 2100 if present trends continue.

In addition, climate change is likely to increase the variability of precipitation patterns in California, and may increase the frequency and intensity of drought in California. For example, as a direct result of California's ongoing historic drought, golden trout in Volcano Creek and adjacent wetland meadows (Tulare Co.) were rescued from drying, isolated pool habitat. As predictions of more frequent drought become reality, such continues are likely to be necessary in the future to continue persistence of the species.

Status Score = 1.9 out of 5.0. Critical Concern. The California golden trout is listed as a Species of Special Concern by California Department of Fish and Wildlife and as a Sensitive Species by the USDA Forest Service. The American Fisheries Society lists it as threatened, while NatureServe lists it as "Critically Imperiled" (Jelks et al. 2008).

A petition to the USFWS to list California golden trout as federally endangered was submitted by Trout Unlimited in 2000 (Behnke 2002). The U.S. Fish and Wildlife Service determined in a 90-day finding that the proposal deserved additional consideration. After a 10 year review, the USFWS concluded (October 11, 2011, 76 FR 63094) that listing was not warranted because of the collaborative efforts taking place to protect the trout, particularly the ongoing and active implementation of the Conservation Assessment and Strategy for the California Golden Trout (1994). This cooperative conservation agreement, signed by state and federal agencies and concerned NGOs, indicated that listing the fish would provide few, if any, additional benefits to it. According the *Federal Register* (76 FR 63094): "The purposes of the Conservation Strategy are to: (1) Protect and restore California golden trout genetic integrity and distribution within its native range; (2) Improve riparian and instream habitat for the restoration of California golden trout populations; and (3) Expand educational efforts regarding California golden trout restoration and protection." Until recently, the California golden trout was perceived as secure because it had been widely introduced throughout the Sierra Nevada and the Rocky Mountains. However, these introduced populations are likely on a different evolutionary trajectory from the native populations (most are in lakes) and they have also largely hybridized with rainbow trout. Nonetheless, Stephens and May (2011) show a number of populations do exist outside the native range that are unhybridized or only slightly introgressed. As Stephens and May (2011) point out:

> "…it is possible that these populations could be preserved *in situ* as an insurance policy against the loss of CAGT [California golden trout] within their native range or possibly utilized in other conservation or restoration efforts. Any introduction of these fish into the native CAGT range should be considered with caution: 1) future genetic analysis may reveal introgression previously undetected, 2) they do not appear to contribute any unique allelic diversity not already represented in the extant native range populations, and 3) they may have experienced substantially different selection regimes in their watersheds, possibly rendering them less (or more) fit than extant CAGT (p. 12)."

Meanwhile, even slightly hybridized populations in the native range can only be maintained through constant intervention such as building and repairing barriers and eradication of non-native trout and golden-rainbow hybrids (Behnke 2002).

Table 2. Metrics for determining the status of California golden trout in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	1	Unhybridized California golden trout are confined to a few small tributaries in one watershed.
Estimated adult abundance	3	Volcano Creek populations may be <1,000 but, if other populations with conservation value within native range are counted, the numbers would be much higher, perhaps 50,000.
Intervention dependence	2	Annual monitoring of barrier performance required; continued implementation of Conservation Strategy is critical. Rescued individuals from Volcano Creek will need to be re-introduced based on genetic management strategy.
Tolerance	3	Generally tolerant of a wide range of conditions and habitats within their native range.
Genetic risk	1	Hybridization with rainbow trout is a constant high risk.
Climate change	1	Rated critically vulnerable in Moyle et al. (2013).
Anthropogenic threats	2	1 High, 1 Medium threat.
Average	1.9	13/7.
Certainty (1-4)	4	Well documented.

Management Recommendations: The overarching goal of California golden trout management should focus on the maintenance of self-sustaining populations in refuges that can persist through long periods of less intensive management and/or extended drought. Populations in their native range have persisted because of continuous, cooperative actions by the California Department of Fish and Wildlife, USFWS, and U.S. Forest Service, along with volunteers from multiple groups. Ever since it was realized in 1968 that California golden trout in the SFKR were threatened by alien trout, mainly brown trout, major efforts have been undertaken to create refuges for golden trout in the upper reaches of the SFKR by constructing three barriers (Ramshaw, Templeton, Schaeffer) and then applying rotenone and antimycin to eradicate unwanted fish above or between barriers. From 1969 through 2000, 10 treatments were carried out, with varying degrees of success (Stephens et al. 2004). In addition, gill netting of selected headwater lakes (e.g. Chicken Spring Lake, Rocky Basin lakes) to remove hybridized fish has been successful and these lakes are now fishless. The future focus of conservation should be protection of the original gene pools of golden trout in GTC and SFKR as: (1) a source for future fish transplants into restored streams, (2) stocks that can be genetically compared with introduced populations, and (3) an aesthetic measure. However, special protection should also be provided to demonstrably unhybridized populations outside the native range, as an insurance policy against complete loss of unhybridized fish from within the native range.

Implementation of the Conservation Strategy for California golden trout should reduce the threat of extinction through management of hybrids, maintenance of multiple barriers (redundancy in case one fails), improved management of watersheds, and elimination of non-native trout populations (S. Stephens et al. 2004). This strategy continues to be implemented and several key goals of this document have been met. These include the replacement of two failing fish barriers and increased genetic research to better understand the overall status of California golden trout. Construction of an additional barrier in the lower portions of the South Fork Kern drainage is being explored. Two of the four existing grazing allotments in the area have been rested since 2001. Additional management actions needed include: (1) repair or replacement of barriers, (2) eradication of all rainbow trout and brown trout populations that threaten California golden trout, (3) utilization of recent genetics techniques to refine management, (4) improved management of livestock grazing, (5) modified recreation management strategies, and (6) expanded efforts to further implement the Conservation Strategy.

Barrier improvement. Barriers to prevent alien trout from invading golden trout waters are important, if ultimately short-term, management measures. Templeton and Schaeffer barriers were replaced with major concrete structures in 1996 and 2003 respectively, and have reduced the probability of unwanted invasions. However, because accessible barriers that have golden trout on one side and brown trout on the other are inherently flawed (by the ease of moving fish over the barrier), other solutions are needed. D. Christensen and S. Stephens suggested (USFS, CDFW, pers. comm. 1995) that "It would seem appropriate to construct a bedrock barrier downstream of Monache Meadows in the gorge area or even further downstream in the drainage, and extend the [California golden trout] population. This would provide a permanent barrier with a great deal less public access." Such a structure at Dutch John Flat is in the early planning stages about 10 km upstream of Kennedy Meadows. Whether such a structure will ever be built in designated wilderness remains uncertain (S. Stephens, CDFW, pers. comm. 2016).

Eradication of alien species. Eradication of non-native trout continues to be a necessary and important measure. Unfortunately, such eradication generally requires the use of the controversial piscicide, rotenone. Alternate toxins (e.g., antimycin) have yet to be approved in California so are unavailable for use. A thorough risk analysis should be conducted for streams in which their use is proposed. The analysis should include risks entailed to the continued existence of the species if they are *not* used.

Use of genetic techniques. Increased use of new genetic techniques is occurring and necessary in order to allow for genetics-based management. A genetics management plan

(GMP) for California golden trout was completed in 2013 (Stephens et al. 2013). The best management approach in the GTC watershed is to monitor populations at intervals of five years or more to assess estimates of introgression from SNPs (single nucleotide polymorphism) and microsatellite analyses. After fully defining the genetic landscape, Stephens et al. (2013) recommend segregating SFKR and GTC populations into distinct management units and then ranking subpopulations within SFKR and GTC based on their conservation value. Conservation value should reflect some composite of genetic status, abundance, and likelihood of responding positively to recovery actions, for example (Stephens et al. 2013). Once the genetic landscape is defined and populations are ranked, recovery actions may proceed. Genetic recovery actions primarily focus on establishing refuge populations outside the native range, considering the use of a conservation hatchery where strict genetic protocols are in place to maximize genetic diversity, and translocating individuals of similar ancestry from one population to another (Stephens et al. 2013). Such actions should help to conserve important genetic diversity. There are risks associated with these actions, such as the potential for outbreeding depression or artificial selection associated with a hatchery environment, but there are also potential benefits regarding the conservation of important genetic diversity. As such, recovery actions should be conducted experimentally and within an adaptive management framework.

As mentioned, there is a cline of hybridization in the SFKR with levels of introgression with non-native rainbow trout increasing downstream (Stephens 2007). Plans to install a new fish barrier at Dutch John Flat should be pursued. Using the guidance of the GMP and the Conservation Strategy, managers should develop appropriate plans and take steps to fully eradicate brown trout and hybrid golden trout utilizing the system of SFKR barriers. These activities may take years to accomplish, but offer large rewards for golden trout in terms of greatly expanded range and protection from hybridization, competition and predation.

Grazing. Improvements have been made in livestock grazing management in the Golden Trout Wilderness Area in recent decades, but further refinement and restrictions may be necessary to protect golden trout populations and their habitat. Continued resting of grazing allotments (or elimination of allotments altogether) should result in recovery of riparian vegetation and associated shading, improved stream channel morphology, and increased abundance of invertebrate food supplies for fish (Herbst et al. 2012). According to the USFWS (2011, *Federal Register* 76 FR 63094), changes in grazing management practices for the past 10 years or so, including resting allotments, have removed grazing as a primary threat to golden trout but the practice may still cause degradation of streams. If complete elimination of grazing is infeasible, then intense management of grazing to reduce impacts on streams should be continued and expanded, including the use of allotment rotation, seasonal closures during periods when meadows are wet, herd size reduction, expanded fencing, and active herd management to keep cattle away from streams. Monitoring of grazing practices needs to continue in order to document compliance with appropriate U.S. Forest Service guidelines.

Recreation management. Improvement of recreation management is needed, which should include better enforcement of existing laws and increased public education programs. Forest Road (Route) closures should be implemented where needed (e.g., eliminate off-road vehicles from areas where they are currently directly impacting streams).

Integrated management. The CDFW performs regular monitoring of populations in the native range (Carmona-Catot and Weaver 2006, Weaver and Mehalick 2008, Weaver and Mehalick 2009), and these surveys should continue in order to determine population status and to document the presence and distribution of non-native trout. The CDFW plans greatly expanded genetics, population structure and abundance, and habitat monitoring in the near future which will include random stratified sampling of sites throughout the SFKR and GTC drainages (J. Weaver, CDFW, pers. comm. 2013). This level of sampling will provide scientifically rigorous and objective data to inform future management on a much broader spatial scale than ever performed. Beyond expanded monitoring, two kinds of refuges in the native range should also be established for managing California golden trout: (1) streams containing unhybridized populations and (2) streams containing populations with low levels of hybridization (S. Stephens et al. 2004). Defensible streams (by barriers) that do not meet these criteria should be converted to one or the other type of refuge as soon as possible. This type of intensive management requires periodic genetic assessments of refuge populations. In addition, populations of unhybridized California golden trout found outside the native range should also receive special protection and management, as described for populations in the native range. These would serve as additional refuge populations and could be used for experiments in management (e.g., modified grazing practices, introductions from other populations to increase genetic diversity) without compromising genetically 'pure' populations within the native range. For information on additional management measures, see Stephens et al. (2004) and Sims and McGuire (2006).

COASTAL CUTTHROAT TROUT
Oncorhynchus clarkii clarkii (Richardson)

Moderate Concern. Status Score = 2.7 out of 5.0. Coastal cutthroat trout populations in California are small, fragmented, and face multiple threats, including cumulative impacts from land use practices and predicted outcomes of climate change in their range. However, their numbers appear to be stable in the few watersheds they inhabit along the Northern California coast.

Description: Coastal cutthroat trout are similar in appearance to coastal rainbow trout (*O. mykiss*) but have heavier spotting, particularly below the lateral line, and heavy spots on ventral fins. Adults have spotting on the lower mandible and more pointed heads than coastal rainbow trout. The spots become nearly invisible when fish become silvery during smolting and migrations to and from the sea. Mature fish in fresh water have a dark coppery or brassy appearance, especially on the fins (Behnke 1992, Moyle 2002). Cutthroat trout are more slender than rainbow trout and possess characteristic red to orange to yellow slashes under the mandibles, though the slashes are rarely visible until the fish reach over 80 mm total length (TL) (Scott and Crossman 1973, Behnke 1992). Larger fish have long maxillary bones extending past the eye. Well-developed teeth are found on the jaws, vomer, palatines, tongue, and sometimes on the basibranchial bones (Rizza 2015). The dorsal fin has 9-11 rays, the anal fin 8-12 rays, the pelvic fins 9-10 rays, and the pectoral fins 12-15 rays. There are 15-28 gill rakers on each arch and 9-12 branchiostegal rays. The caudal fin is moderately forked and scales are smaller than those of rainbow trout, with 140-200 along the lateral line (Behnke 1992). Parr possess 9-10 widely spaced parr marks (vertical bars) along the lateral line and juveniles may be difficult to distinguish from rainbow trout parr by visual identification (Kennedy et al. 2009, Rizza 2015). Anadromous forms rarely exceed 40 cm fork length (FL) and 2 kg, but individuals reaching 70 cm and 8 kg have been recorded. It is uncommon for individuals from landlocked populations to exceed 30 cm FL. Tagged fish in California have been known to live up to seven years old (M. Sparkman, CDFW, pers. comm. 2016).

Taxonomic Relationships: The coastal cutthroat has long been recognized as distinct and was the first cutthroat trout described by John Richardson in 1836. He used the name "*Salmo clarkii*," so *clarkii* with a 'double-i' ending is the correct name (Trotter 2007). Behnke (1992, 1997) proposed that approximately one million years ago, cutthroat trout diverged into two major lineages, the coastal cutthroat (*O. c. clarkii*) and all the other interior subspecies (with complex evolutionary histories). Coastal cutthroat are characterized by having 68 chromosomes and interior cutthroat subspecies are characterized by possessing either 66 or 64 chromosomes. The 64-chromosome fish include Lahontan cutthroat (*O. c. henshawi*) and Paiute cutthroat (*O. c. seleneris*) in California (Trotter 2007). Coastal cutthroat have numerous populations across their range that spend their entire life cycle in fresh water but are genetically connected to sea-run populations (Trotter 2007). Coastal cutthroat colonized coastal rivers from the Eel River in northern California to Prince William Sound in Alaska (Johnson et al. 1999). California's populations are at the southern end of the coast range lineage and include both sea-run and freshwater populations, which are considered part of the Southern Oregon-California Coast DPS (Johnson et al. 1999; Trotter 2007). While separate species, coastal cutthroat and steelhead hybridize naturally, and specific processes and phenomena that have become well studied recently keep them distinct (Beuhrens et al. 2013, Rizza 2015).

Distribution: Coastal cutthroat trout are distributed from the Cowpen Creek, in Prince William Sound, Alaska, to tributaries to the Salt River, a southern tributary to the Eel River estuary (Humboldt Co.) in California. In California, coastal cutthroat trout have been observed in 182 named streams (approximately 71% of the 252 named streams within their range in California) and an additional 45 streams may support populations (Gerstung 1997, Figure 1).

Figure 1. Coastal cutthroat trout observations (white circles) in California showing the southern extent of the species range near the Eel River watershed. From: CCTIC 2015.

Coastal cutthroat trout persist in lower Salt River tributaries near Ferndale and lower Eel and Van Duzen river tributaries such as Stitz and Fox creeks (A. Renger, CDFW, pers. comm. 2016, CCTIC 2016). North of the Eel River, their range coincides closely with that of temperate coastal rainforest and spans most of the Smith River tributaries to the Oregon border through Humboldt, Del Norte, and Siskiyou counties (Trotter 2007, CCTIC 2016). The interior range of the subspecies in Washington, Oregon, and California is bounded by rain forests on the western slope of the Cascade Range; their range rarely extends inland more than 160 km and is usually less than 100 km (Johnson et al. 1999). In California, this band is to near the South Fork of the Eel River and nearly as far in the Van Duzen River, and is nearly 48 km wide at the Oregon border (Moyle 2002). However, a small resident population exists in Elliot Creek in Siskiyou County, about 120 km from the ocean. Elliot Creek is a tributary to Applegate River in Oregon, which drains into the Rogue River. Fish from Elliot Creek have been transplanted successfully to Twin Valley Creek in the Klamath River watershed (Moyle 2002), where they still persist (J. Weaver, CDFW, pers. comm. 2011). Cutthroat from other parts of their range have also been successfully transplanted to Indian Creek, also in the Klamath River watershed (M. McCain, USFS, pers. comm. 2011). Recent anecdotal evidence also places a coastal cutthroat population as far inland as Panther Creek, tributary to Redwood Creek (M. Sparkman, CDFW, pers. comm. 2016), and a smolt trap in mid-upper Redwood Creek (rkm 53) captures small numbers of coastal cutthroat each year (Sparkman 2016).

Self-sustaining populations apparently occur in many coastal basins, including Humboldt Bay tributaries, Little River, and Redwood Creek (Gerstung 1997). The principal large basins where coastal cutthroat trout occur are the Smith and lower Klamath rivers and Redwood Creek (M. Sparkman, CDFW pers. comm. 2016). Cutthroat trout also rear in approximately 1875 ha of habitat in several coastal lagoons and ponds: Big, Stone, Freshwater, and the Lake Earl-Talawa complex, though may no longer be present in Espa Lagoon due to desiccation (Gerstung 1997). The largest populations are currently in the Smith River, and to a lesser extent, the lower Klamath River and tributaries, though pristine Prairie Creek, with its intact old-growth forest is perhaps the most productive coastal cutthroat habitat remaining in the state (Gale and Randolph 2000, M. Sparkman, CDFW, pers. comm. 2016). Gerstung (1997) indicated that the lower Mad River is another area of high cutthroat occupancy, but more recent assessments indicate that it contains only a small population and does not produce as many fish as neighboring watersheds (M. Sparkman, pers. comm. 2016).

Historical coastal cutthroat trout distribution once encompassed all or nearly all Smith River tributaries, and may have once extended farther south to the Russian River in Sonoma County (DeWitt 1954). There are anecdotal reports of cutthroat trout in several streams from the Mattole River down to the Garcia River (Gerstung 1997); however, there are currently no known populations south of the Eel River. No coastal cutthroat trout have been observed in neighboring Mendocino County in recent memory (S. Gallagher, CDFW, pers. comm. 2016).

Life History: Coastal cutthroat trout are the least understood salmonid along the North Coast of California due to the dearth of studies directed at them; they are neither commercially harvested nor listed under the Endangered Species Act. In addition, they possess widely variable life history strategies (DeWitt 1954; Pauley et al. 1989, Moyle 2002, Beuhrens et al. 2013). This plasticity is among the most extreme in Pacific salmonids, and variations in migratory behavior are found both between and within populations. Trotter (2007) categorizes the diversity into four main life history groups: (1) amphidromous (sea-run), (2) lacustrine, (3) riverine (potadromous), and (4) stream-resident. These diverse life history strategies make habitat connectivity and access crucial to the continued persistence of this species in its native range. The amphidromous forms are not considered anadromous because they can move back and forth between fresh and salt water multiple times to feed, often on other juvenile salmonids, although they must also migrate into fresh water to spawn. Lacustrine coastal cutthroat use large lakes or lagoons like the ocean (such as Lake Earl (Talawa) in California) to attain their largest sizes (W. Duffy, HSU, pers. comm. 2016). Potadromous forms are found in rivers and make seasonal migrations up and down these systems for a variety of reasons. Resident populations are typically found above natural barriers, in small headwater streams (such as Jones Creek, tributary to the S. Fork Smith River). Offspring of fish with one life history can adopt any one of the four life histories (Beuhrens et al 2013). The Smith, Klamath, and Eel rivers in California have both amphidromous populations and resident populations isolated in small streams upstream of barriers (e.g., Little Jones and Tectah creeks).

Sea-run cutthroat trout generally make their first migrations at one to two years of age, although they can enter seawater as late as their fifth year (Wilzbach et al. 2016). When multiple forms coexist, temporal and spatial segregation may influence the genetic structure of the population and lead to genetic differentiation within a watershed. Environmental conditions that affect growth rate, such as food availability, water quality and quantity, and temperature influence migratory behavior and residency time (Hindar et al. 1991, Northcote 1992, Johnson et al. 1999, Sparkman et al. 2016). Johnson et al. (1999) noted that the large variability in migratory behavior may be due to habitat being most available for cutthroat trout at times when it is not being used by large numbers of other anadromous salmonids; this flexibility may release cutthroat trout from competition and predation pressures at certain times of year, while allowing them to track juvenile salmonids as prey (Trotter 2007, Sparkman et al. 2016).

Coastal cutthroat trout have ecological requirements analogous to those of resident rainbow trout and steelhead. When the two species co-occur, cutthroat trout occupy smaller tributary streams, while competitively dominant steelhead occupy larger tributaries and rivers. As a consequence, cutthroat trout tend to spawn and rear higher in watersheds in small tributaries than steelhead, as well as in estuaries (Wilzbach et al. 2016). While cutthroat and rainbow trout can naturally hybridize, this habitat partitioning and some degree of spatial/temporal segregation are likely important reproductive barriers where their distribution overlaps (Beuhrens et al. 2013). Age at first spawning ranges from 2 to 4 years, depending on migratory strategy and environmental conditions (Trotter 1991). Coastal cutthroat can live 4-7 years, with non-migratory fish often reaching sexual maturity earlier and at a smaller size than anadromous fish (Trotter 1991, Johnson et al. 1999, M. Sparkman, CDFW, pers. comm. 2016). Resident fish generally reach sexual maturity between the ages of 2 and 3 years, whereas sea-run fish rarely spawn before age 4 (Johnson et al. 1999). Sexually mature trout can demonstrate precise homing capabilities in their spawning migrations to natal streams. In northern California, coastal cutthroat trout migrate upstream to spawn after the first significant rain, beginning in fall. Peak spawning occurs in December in larger streams and January to February in smaller streams (Johnson et al. 1999). Ripe or nearly ripe females have been caught from September to April in California streams, indicating a prolonged spawning period.

Females dig redds in clean gravels with their tails, predominantly in the tails of pools in low gradient reaches, often with low flows (less than 0.3 m^3/second summer flows) (Johnston 1982, Johnson et al.1999, Trotter 2007). The completed redds average around 35 cm in diameter by 10-12 cm deep. After spawning is completed, the female covers her redd with about 15-20 cm of gravel. Each female may mate with numerous males. Fecundity ranges from 1,100 to 1,700 eggs for females between 20 and 40 cm TL. Coastal cutthroat trout are iteroparous with a higher incidence of repeat spawning than steelhead. They can spawn every year, but post-spawning mortality can be quite high. Maximum age recorded for coastal cutthroat is 14 years, from Sand Creek, Oregon, though fish this old have not been documented in California to date (Trotter 2007, M. Sparkman, CDFW, pers. comm. 2016).

Eggs hatch after incubating for 6-7 weeks, depending on water temperature. Alevins emerge as fry between March and June, with peak emergence during mid-April. Newly hatched fish spend the summer in backwaters and stream margins (Johnson et al. 1999). Juveniles remain in the upper watershed until they are approximately 1 year old, when they may begin to move extensively throughout the watershed. It is difficult to determine the difference between sea-bound smolts and silvery parr moving back up into the watershed at this age (Johnson et al. 1999). Smolts or adults entering the saltwater environment remain close to the shore and do not normally venture more than about 7 km from the edge of the coast (Johnson et al. 1999). Typically, they stay in or close to the plume of the river in which they were reared for feeding (Trotter 2007, W. Duffy, HSU, pers. comm. 2016). Individuals can spend months in estuaries, often moving in and out of fresh water, likely taking advantage of different feeding and rearing habitats in brackish estuaries and river plumes in the ocean. Cutthroat trout up to ~350 mm were captured in the Smith River estuary from May- October 1997-2001 (R. Quiñones, unpubl. obs). The largest recorded cutthroat in Prairie Creek, Humboldt Co., measured 522mm (M. Sparkman, CDFW, pers. comm. 2016). A similar pattern is observed in the Klamath River estuary (M. Wallace, CDFW, pers. comm. 2013). The largest coastal cutthroat in California are said to inhabit Lake Earl (Tolowa), Big Lagoon, and Stone Lagoon, though these claims from fishermen are unsubstantiated (M. Sparkman, CDFW, pers. comm. 2016).

Adults feed on benthic macroinvertebrates, terrestrial insects in drift and small fish, while juveniles feed primarily on zooplankton, macroinvertebrates, and microcrustaceans (Wilzbach 1985, Romero et al. 2005). White and Harvey (2007) found that cutthroat trout of all sizes in small creeks fed mainly on aquatic insects, but that earthworms washed in by winter storms may be the most important food item for overwintering. Cutthroat captured in Prairie Creek, a well-studied tributary to Redwood Creek, appear to feed opportunistically and voraciously on various prey, including migrating Chinook salmon and steelhead fry during peak outmigration periods, and they can act as an important piscivorous predators (M. Sparkman, CDFW, pers. comm. 2016), and cutthroat captured in the Klamath estuary regurgitated salmon eggs during late summer, when large numbers of adult salmon were being caught and cleaned (M. Wallace, CDFW, pers. comm. 2013). In the marine environment, cutthroat trout feed on various crustaceans and fishes, including Pacific sand lance (*Ammodytes hexapterus*), salmonids, herring and sculpins.

Habitat Requirements: Coastal cutthroat trout require cool, clean water with ample cover and deep pools for holding in summer. They prefer small, low-gradient coastal streams and estuarine habitats, including lagoons, but also persist in small headwater streams with historical passage to the Pacific (A. Renger, CDFW, pers. comm. 2016). Preferred water velocities for fry are less than 0.30 m/sec, with an optimal velocity of 0.08 m/sec (Pauley et al. 1989). Summer flows in natal streams are typically low, averaging 0.12 m^3/sec in Oregon, and less than .03 m^3/sec in California tributaries (Pauley et al. 1989, DeWitt 1954). Adults overwintering in streams prefer pools with fallen logs or undercut banks, but will also utilize boulders, depth, and turbulence as alternative forms of cover in the absence of woody debris (Gerstung 1998, Rosenfeld et al. 2000, Rosenfeld and Boss 2001). Juveniles generally rear in smaller streams with dense overhead cover and cool summer temperatures (Rosenfeld et al. 2000, 2002). Presence of large woody debris can provide refuge for juveniles during winter high flow events (Harvey et al. 1999). Spawning takes place in small streams with small to moderate sized gravel ranging from 0.16-10.2 cm in diameter. Spawning coastal cutthroat preferentially use riffles and the tails of pools with velocities of 0.3-0.9 m/sec, although they have been observed spawning in velocities as low as 0.01-0.03 in small streams in Oregon (Pauley et al. 1989).

Optimal stream temperatures for coastal cutthroat are less than 18°C, with preferred temperatures around 9-12°C. This may explain why they occur mainly in more northern streams in California, within the coastal fog belt. Perhaps the most productive coastal cutthroat trout stream in California, Prairie Creek, maintains temperatures less than 14°C in its tributaries throughout the summer (Duffy 2013). Interestingly, with only 38km of available anadromous habitat, relatively pristine Prairie Creek still manages to produce significantly more juvenile cutthroat per year than the more developed and degraded mainstem Redwood Creek and its 150km of accessible habitat (Sparkman et al. 2016). In Washington streams, most rapid growth occurred at 8-10°C, in early summer, with rates declining as temperatures rose to 12-14°C (Quinn 2005). Spawning has been recorded at temperatures of 6-17° C, with preferred temperatures of 9-12°C (Pauley et al. 1989, Moyle 2002). Coastal cutthroat require high dissolved oxygen levels and will avoid areas with less than 5 mg/L dissolved Oxygen in summer months (Pauley et al. 1989). Feeding and movement of adults are impaired at turbidities of greater than 35 ppm. Embryo survival is greatly reduced at turbidities greater than103 ppm and dissolved oxygen levels less than 6.9 mg/L.

Trends in Abundance: There are limited long-term data sets available to evaluate population trends in coastal cutthroat trout. Data from California are spotty, scattered, and typically unpublished, but suggest that coastal cutthroat populations are probably significantly less than they were historically, and are either currently stable or in decline. Records suggest that coastal cutthroat trout supported substantial fisheries and were even stocked in the Mad River dating back to at least the 1920s (Gerstung 1997, S. Van Kirk 2013). Current coastal cutthroat trout abundance is thought to generally be low in most waters, particularly where juvenile steelhead are present (Johnson et al. 1999, Griswold 2006). Effective population size in California streams is difficult to determine, but Gerstung (1997) estimated that there are likely less than 5,000 spawners each year in all of California. While small, the existing populations appear to be stable in some areas, such as the Redwood and Prairie Creek basin 2015/2016 (Sparkman et al. 2016).

The largest population apparently exists in the Smith River, where a local watershed group, the Smith River Alliance (SRA) and U.S. Forest Service conduct annual snorkel surveys for salmon and trout. Previous population and trend data collections from the Smith River have been intermittent and represent only a small portion of the range with inconsistent locations and methods over the years. The Yurok Tribe has conducted anadromous salmonid surveys on the lower Klamath River and tributaries and found cutthroat widely distributed in medium to high densities in nearly all lower Klamath tributaries downstream of Mettah Creek (Gale and Randolph 2000).

Quantitative measures of historical abundance are lacking, so it is difficult to determine whether populations are declining increasing, or stable (Johnson et al. 1999, Griswold 2006, Rizza 2015). Coastal cutthroat numbers have likely decreased due to extensive estuary, watershed, and stream alteration throughout their range in California over time. Fortunately, there is increasing protection in some areas (e.g., Smith River, streams in Del Norte Coast Redwoods State Park), in part to protect listed Coho salmon that also benefit coastal cutthroat trout. One long-standing CDFW smolt outmigration study on Prairie Creek (Figure 1) has gained valuable data on juvenile and some adult cutthroat trout in these drainages. In 2016, the population abundance of coastal cutthroat smolts through lower Prairie Creek was 63% less than average, which could be a response to ongoing historic drought conditions across the state (M. Sparkman, CDFW, pers. comm. 2016) and warrants continued monitoring.

	Year										
Species	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2011
Coho salmon 1+	29,606	5,616	23,398	13,962	3,590	1,381	2,114	4,255	4,352	2,503	8,446
Prod Est ± S.D.	(6,638)	(1,516)	(3,506)	(1,622)	(646)	(540)	(423)	(851)	(1,175)	(676)	(650)
	Raw Data										
Coho salmon 0+	2,702	2,690	1,677	230,745	17,648	15,765	9,410	4,282	408	2,209	778
Chinook salmon	56,615	22,409	18,026	110,148	11,038	45,358	2,772	1,401	598	288	7,743
Steelhead trout	218	284	505	172	68	8	57	160	216	188	2,289
Cutthroat trout	462	394	1,316	477	36	49	233	479	458	299	1,219
Unidentified trout	1,350	1,788	3,482	1,504	4,677	2,442	169	103	399	13	--
Pacific lamprey	56	30	25	113	20	0	9	18	46	40	43
Western brook lamprey	159	1	219	9	51	1	80	58	58	13	73
Lamprey ammocete	152	18	189	246	79	79	0	0	0	0	1,335
Coastrange sculpin	95	24	154	71	80	0	18	28	13	12	1,498
Prickly sculpin	124	147	344	97	18	3	248	382	134	146	1,693
Threespine stickleback	41	13	103	78	46	6	62	117	60	66	1,011
Sacramento sucker	13	0	6	3	1	0	0	6	2	4	120

Figure 2. Coastal cutthroat trout smolt estimates from Prairie Creek (Redwood Creek, Humboldt Co.), 1999-2011. From: Duffy 2013, Table 13, pg. 32).

Factors Affecting Status: Coastal cutthroat trout populations are affected differentially by one or more stressors, depending on location (Table 1). According to Gregory and Bisson (1997), degraded habitat is associated with more than 90% of documented extinctions or declines of Pacific salmonid stocks. Major anthropogenic land-use activities, including agriculture, forestry, grazing, water diversions, urban and industrial development, road construction, and mining have resulted in the alteration and loss of cutthroat trout habitat and a subsequent reduction in production (Johnson et al. 1999). Fish passage issues from faulty tidegates, road crossings, and diversions lead to loss of over-wintering habitat, changes in geomorphic processes and channel geometry, channelization and simplification of habitat in estuaries, loss of sediment and large wood in channels, and general habitat degradation in the coastal cutthroat trout's range (A. Renger, CDFW, pers. comm. 2016). While treated separately in this account, the causes of decline often interact synergistically.

A unique problem relates to the effects of habitat alteration on interactions between steelhead and cutthroat trout. The two species naturally co-occur and hybridize naturally. (Rizza 2015). However, habitat disturbance and anthropogenic disturbances such as logging, stocking fish, and land development increase rates of hybridization (Neillands 2001, Heath et al. 2010, Rizza 2015) but more study must be done on this subject. Recent genetics work from the Smith River basin indicates that perhaps as many as 19% of juvenile coastal cutthroat trout have some trace of steelhead ancestry in this stronghold, but this estimate may be high based on sampling design (Rizza 2015). Most of the resulting offspring are more closely related to coastal cutthroat trout than steelhead due to asymmetric introgression, whereby hybrids with intermediate characteristics are more likely to spawn with pure cutthroat rather than steelhead over time. It is posited that reduced fitness or ability to undertake lengthy ocean migrations keeps hybrid offspring from mating with larger, pure steelhead adults (Rizza 2015). Even while coastal cutthroat and steelhead overlap spatially and temporally somewhat while spawning, cutthroat are generally found spawning further upstream in smaller tributaries than the larger steelhead. Reproductive isolation, inherent genomic incompatibility, assortative mating, and recombination of DNA disrupting fitness links may all be acting on cutthroat and steelhead populations to allow them to remain distinct species in the watersheds where they overlap (Ostberg et al. 2004, Buehrens et al. 2013, Rizza 2015).

Major dams. Dams and diversions have altered flows in a number of coastal rivers, especially the Klamath and Mad rivers, within coastal cutthroat trout range. The impact of these dams on cutthroat trout is not known. Likewise, the effects of small diversions, common in coastal streams, are not known. Anecdotal evidence suggest that the Mad River in particular does not support the numbers of coastal cutthroat trout it once did, which may be related to significant development and impacts from dams in this watershed (W. Duffy, HSU, pers. comm. 2016, M. Sparkman, CDFW, pers. comm. 2016, Van Kirk 2013). Dams and diversions can isolate once-connected populations of cutthroat from one another and the ocean, with presumably negative consequences. However, Matthews Dam on the Mad River may provide important flows during summer months for salmonids, whereas in the past downstream reaches would dry up (M. Sparkman, CDFW, pers. comm. 2016).

Logging. Logging and associated road networks have caused tremendous impacts to coastal cutthroat trout habitats in California through landslides and erosion stemming from excessive tree removal and road construction on steep, unstable soils. Small streams favored by cutthroat trout are inherently more susceptible to such impacts and have been disproportionately damaged by timber harvest practices. Johnson et al. (1999) cite numerous studies highlighting the importance of riparian vegetation to fish production in California, and note approximately 89% of the state's riparian forest has been lost with associated declines in aquatic habitat. Removal of trees has starved watersheds of valuable large woody debris, which serves as important cover and holding habitat for coastal cutthroat trout (M. Sparkman, CDFW, pers. comm. 2016). Heavy erosion and sedimentation causes increased turbidity, as well as burying spawning gravel, altering rearing habitats, and filling pools. Additionally, clear cutting in headwater basins has decreased shading and reduced the absorption capacity of soils. In certain areas, this is likely to have increased stream temperatures and incidence of flash flooding, as well as reduced late summer and early fall base flows. It is worth pointing out that California's most productive coastal cutthroat stream, Prairie Creek, with only 38km of accessible stream to anadromy, lies entirely within a state park and retains intact old-growth forest along its tributaries. This watershed has been subject to minimal development, making it one of the few habitats available to the species that are neither sediment nor temperature-impaired left in the state (M. Sparkman, CDFW, pers. comm. 2016). Legacy impacts from industrial logging to coastal cutthroat trout have been substantially reduced in many watersheds thanks to strict timber harvest regulations and many restoration efforts. However, the cumulative impacts of past logging practices still likely reduce salmonid carrying capacity throughout the range of coastal cutthroat trout and warrant further study.

Agriculture. Agricultural practices that most likely impact cutthroat trout are reclamation of estuarine marshes, water diversions and associated dike building, damming, culverts, and runoff. These activities degrade water quality, increase water temperature, reduce in-stream flows, and eliminate estuarine rearing areas (Johnson et al. 1999). Increasingly, marijuana growing in the north coast region where cutthroat reside is a major threat to aquatic habitats because growers divert water from headwater streams, convert large areas of former timber lands to monoculture crop production, pollute streams with pesticides and other material, and degrade stream habitats. Unfortunately, there are no known studies that document such impacts specifically for coastal cutthroat trout, but the impacts to small tributaries where these fish spawn and rear can likely be significant.

Grazing. Grazing occurs in most of the former wetlands surrounding the Smith and Eel river estuaries and Humboldt Bay. Grazing and other agricultural uses of the land are prevalent in the lower Mad River, Little River, and Redwood Creek. These rivers have largely been isolated from their surrounding riparian habitat, to the detriment of cutthroat trout. In addition, complete blockage of access to estuarine marsh channels by faulty tide gates, diversions, and dikes to keep pastures from flooding greatly reduce rearing habitat and disrupt migrations to and from estuaries and the ocean for cutthroat feeding and other purposes.

Rural/residential development. A significant portion of cutthroat trout habitat in California lies within state or federal park or national forest boundaries. Residential areas are scattered throughout their range, and likely impact fish through habitat alteration, water diversion, and pollution from leaking septic tanks or surface runoff. These effects are mostly localized but cumulatively could pose significant threats during drought periods.

Urbanization. Urbanization plays an important role in reducing cutthroat trout habitat in urban streams in the Humboldt Bay region and around Crescent City (T. Weseloh, CalTrout, pers. comm. 2008). These streams generally have reduced overhead and in-stream cover, shallower pools, and poorer water quality than less disturbed streams, which reduce holding habitat for cutthroat.

Transportation. Roads or railroads line most cutthroat streams and most date back to past eras of heavy exploitation of natural resources. They continue to be a major source of habitat loss for cutthroat trout through continued bleeding of sediment into streams. Poorly constructed or placed culverts and crossings prevent access to headwater areas and significantly hamper migration of this species, resulting in loss of gene flow to isolated headwater populations. In addition, roads, railroads, and other infrastructure associated with transportation and urbanization limit habitat restoration projects because 'hardened' banks are very difficult and expensive to restructure into viable habitat for fish.

Harvest. Gerstung (1997) indicated that historical runs of coastal cutthroat trout were quite large and that, in some areas, substantial commercial and sport fisheries existed for them, such as in the Mad River (Van Kirk 2013). Today, fisheries for coastal cutthroat occur mainly in coastal lagoons, where populations tend to be largest. Fisheries elsewhere are small and largely catch-and-release, although impacts from harvest on coastal cutthroat trout populations are unknown. In general, coastal cutthroat trout perhaps luckily receive considerably less attention from anglers than salmon and steelhead fisheries of the north coast, but this general lack of targeted effort makes their study and collection of anecdotal evidence difficult.

Hatcheries. Coastal cutthroat trout are generally competitively subordinate to all other species of salmonids (Johnson et al. 1999) and hatchery steelhead, in particular, are likely to affect their numbers through predation and competition, as well as disease (Johnson et al. 1999). Direct competition and hybridization with hatchery steelhead remain possible, with relatively unknown consequences for the health of the species (Heath et al. 2010).

Estuarine alteration. Estuaries are important for cutthroat trout rearing and passage, yet most in California have been severely altered (and simplified) by agriculture, rural and urban development, and associated channelization. In general, there is much less habitat available in the larger estuaries (e.g. Eel and Smith rivers, Humboldt Bay) than in the past, and the remaining habitats are less well connected, causing isolation of populations in headwater streams to the Salt, Eel, and Van Duzen rivers.

Alien species. Alien species occur throughout the range of coastal cutthroat trout but impacts appear to be small. Aliens with potential impacts to coastal cutthroat trout include: (1) New Zealand mud snail in lower Klamath, Big and Stone lagoons, Lake Earl, lower Smith River, and lower Redwood Creek (below the US 101 Highway bridge); (2) largemouth bass (*Micropterus salmoides*) in the Big Lagoon watershed; (3) striped bass (*Morone saxatilis*) in several estuaries; and (4) Sacramento pikeminnow (*Ptychocheilus grandis*) in the Eel River (M. Gilroy, CDFW, pers. comm. 2011). Pikeminnow may pose an increased threat if they spread beyond the Eel River; CDFW has captured small numbers in Martin Slough, a tributary to Elk River, which flows into Humboldt Bay.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of coastal cutthroat trout in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate based on peer reviewed and gray literature, direct observation, expert judgment, and anecdotal information. See methods for explanation.

Factor	Rating	Explanation
Major dams	Medium	Dams present on some streams.
Agriculture	Medium	Conversion of estuarine wetlands to agricultural lands, diversions, influx of fertilizers and other pollutants into estuaries.
Grazing	Medium	Some impacts in lowland areas, especially where estuary marshes have been converted to pasture.
Rural/ residential development	Medium	Effects localized, but increasingly an issue in Humboldt Bay tributaries and the Crescent City area.
Urbanization	Low	Increasingly an issue in Humboldt Bay tributaries.
Instream mining	Low	No known impact but occurs in some streams.
Mining	n/a	
Transportation	Medium	Roads are an ongoing source of sediment input, habitat fragmentation, and channel alteration.
Logging	Medium	Major activity in many watersheds; dramatic historical impacts in many areas, resulting in lack of large woody debris as cover.
Fire	Low	Increased stream temperatures and sediment input may be a factor in some inland watersheds.
Estuary alteration	High	Estuaries are vitally important rearing, feeding, and migrating habitat and have been significantly altered in most watersheds in California. They are crucial to allowing expression of the full range of coastal cutthroat life histories.

Recreation	Low	Probably minor but may affect populations in heavily used streams.
Harvest	Low	Harvest is generally light but not widely monitored; data mostly from CDFW angler survey boxes at lagoons. There is potential for cutthroat-steelhead hybrids to be misidentified and retained.
Hatcheries	Medium	Hybridization or competition with hatchery steelhead is possible but not well studied.
Alien species	Low	Alien species are common throughout range; impacts to coastal cutthroat are unknown but assumed to be minimal at present.

Effects of Climate Change: Climate change will further stress coastal cutthroat trout populations in California; existing numbers suggest that the overall population in California is low, but perhaps stable in some watersheds such as Redwood Creek (Johnson et al. 1999, Wilzbach et al. 2016). Recent drought conditions have caused shifts in timing of the peak juvenile coastal cutthroat migration from June-July to May in this watershed, indicating migrations from headwater streams with unsuitable habitat may be undertaken out of necessity (Sparkman et al. 2016). These impacts could be exacerbated by likely future climate change scenarios. Their requirements for exceptionally cool water (<18ºC) may mean that even small temperature increases could reduce growth rates and survival. In small headwater streams with no or only occasional influxes of migrating spawners, these impacts may be important.

The suitability of estuarine habitats may also decrease as sea levels rise and more extreme tides and storm surges alter salinity profiles that define food webs. Barriers and diversions in and near estuaries are likely the first places to completely dry up during drought, causing isolation and potential die-offs in certain streams. This past August, low water in Francis Creek, an estuarine tributary to the Salt River, necessitated a fish rescue, where three coastal cutthroat trout and other species were moved away from the Port Kenyon area upstream to available habitat (CDFW Northern Region 2015). If drought conditions persist, these types of interventions are likely to continue to be necessary to help the species persist in some habitats. Sea level rise will also move estuarine conditions farther upstream, potentially causing more competition and/or hybridization between cutthroat trout and steelhead (M. Wallace, CDFW, pers. comm. 2013). For these reasons, Moyle et al. (2013) regarded coastal cutthroat trout as "critically vulnerable" to extinction in California as the result of the added effects of climate change, and they remain vulnerable to habitat loss and degradation impacts from the ongoing drought. If future climate change scenarios make longer and more severe drought more likely, these threats to coastal cutthroat will increase as well.

Status Score = 2.7 out of 5.0. Moderate Concern. Coastal cutthroat trout are in no immediate risk of extinction throughout their range in California. There is still significant uncertainty about their status in the state, and most populations can decline rapidly in response to environmental change such as drought (Sparkman 2016, Table 2). Coastal cutthroat trout persist in many streams on the northern California coast, although many populations are only opportunistically monitored. Coastal cutthroat trout are listed as a Sensitive Species in California by the U.S. Forest Service. Their populations are now entirely dependent on natural reproduction. This makes them unique among the more abundant north coast salmonids, and therefore a good indicator of condition of streams in their range. Nevertheless, coastal cutthroat trout are a non-commercial, non-listed, widely distributed, and somewhat cryptic salmonid that support a very minor sport fishery. California populations appear stable for now, but significant habitat alteration throughout their range, coupled with their fairly narrow environmental tolerances and ongoing drought mean cutthroat populations may face extirpation one at a time. Monitoring coastal cutthroat populations to document the effects of climate change on North Coast rivers is of particular value because of their lack of hatchery influence, dependence upon intact estuarine conditions, low exploitation rates, wide distribution, intolerance of warm temperatures, and preference for smaller streams.

Table 2. Metrics for determining the status of coastal cutthroat trout in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	5	Found in many watersheds from Salt River tributaries in the south north to Oregon.
Estimated adult abundance	3	Many populations are genetically isolated by barriers; most appear to be small and fragmented within California.
Intervention dependence	2	Persistence requires improved management of heavily logged watersheds and extensively altered estuaries. At least one tributary to the Salt River required a fish rescue in August 2015, and more may be required if the current drought continues.
Tolerance	2	Prefer water temperatures below 12ºC.
Genetic risk	2	Recent focus on genetics research on coastal cutthroat hybridization with steelhead indicates that hybridization, while natural, may increase in degraded watersheds or in the presence of hatchery steelhead; may affect populations in most streams in California.
Climate change	2	Most populations exist in small streams or depend upon existing estuary conditions; considerable range-wide vulnerability to climate change.
Anthropogenic threats	3	1 High, 7 Medium threats.
Average	2.7	19/7.
Certainty (1-4)	3	Information compiled from expert judgment, published and gray literature, and anecdotal evidence.

Management Recommendations: The greatest conservation need for coastal cutthroat trout is updated information on their status across their documented distribution in California so appropriate management measures to reconnect isolated populations to the coast can be taken. NOAA Fisheries denied a petition for listing coastal cutthroat trout under the ESA in 1999 due to lack of clear evidence. In 2005-2006 symposia on coastal cutthroat trout were held in Port Townsend, Washington to develop a framework to guide and prioritize conservation, management, research, and restoration of coastal cutthroat trout. This group, the Coastal Cutthroat Trout Interagency Committee, found the state of coastal cutthroat trout research and monitoring remained virtually unchanged and lacking (Griswold 2006). The Committee found in California that virtually all aspects of coastal cutthroat biology, status, and distribution needed updating. Since then, many sources of data have been identified and geo-referenced in a database across the species' range (K. Griswold, CCTIC, pers. comm. 2013, CCTIC 2016).

In California, research and monitoring of coastal cutthroat trout is being performed in concert by multiple partners, including: the California Department of Fish and Wildlife in partnership with Pacific States Marine Fisheries Commission, Humboldt State University, the Yurok Tribe, Green Diamond Resource Company, U.S. Forest Service (Six Rivers National Forest, Redwood Sciences Laboratory), Smith River Alliance and other agencies and groups. The interagency committee has completed a coastal cutthroat trout range-wide status assessment, with data entered from many partners of observations spanning the Salt River tributaries in the South to tributaries to the Applegate River, OR. A long-running smolt outmigration study on Redwood and Prairie Creeks provides perhaps the best data on juveniles and PIT-tagged adult fish in the state, though this program is in jeopardy of ceasing due to funding cuts. Annual snorkel surveys are conducted on the Smith River by the Smith River Alliance, and the Pacific States Marine Fisheries Commission and California Department of Fish and Wildlife collaborate to identify coastal cutthroat trout encountered during sampling for listed salmonids such as Coho salmon. Identifying the full range of coastal cutthroat in California is an important first step toward conservation of this species, but development of a long-term management strategy for the species is heavily dependent on improved dedicated monitoring and assessment and not relying on opportunistic sampling from studies focused on other salmonids.

Griswold (2006) noted "it should be recognized that a voluntary effort that tackles difficult scientific and monitoring issues for a non-listed non-commercial subspecies requires considerable leadership and good will from Federal and State agencies." The multi-agency team with resources from state and federal agencies has helped to fill data gaps and provide the framework for future coastal cutthroat trout conservation. The many measures, both local and regional, taken (or proposed) to protect steelhead and salmon populations should benefit coastal cutthroat trout, although direct benefits remain largely unstudied. Continued management of the Smith River as a free-flowing, wild river that is a refuge for all salmonids, including the seemingly abundant cutthroat trout, is of particular importance. Recent conservation measures have included acquisition and protection of much of the Goose Creek, Mill Creek, Hurdygurdy Creek, Little Jones Creek, and Siskiyou Fork watersheds. Mill Creek has benefited from numerous habitat restoration projects (M. McCain, USFS, pers. comm. 2011). Other targeted restoration efforts include: Lake Earl, Jordan Creek, Stone Lagoon, tributaries to Lake Earl, Big Lagoon, the Eel River estuary, Prairie and Redwood Creeks, and many creeks in Humboldt and Del Norte counties, including Blue Creek in the Klamath basin (M. Gilroy, CDFW, pers. comm. 2011). Blue Creek has become a Salmon Sanctuary of the Yurok Tribe, to protect its diverse salmonids. Based on current evidence, it is clear that Prairie Creek is also an important juvenile production site and should continue to be studied and protected for coastal cutthroat trout as well as threatened Southern Oregon/Northern California Coast Coho salmon (Sparkman et al. 2016, Wilzbach et al. 2016).

COASTAL RAINBOW TROUT
Oncorhynchus mykiss irideus

Low Concern. Status Score = 4.7 out of 5.0. Coastal rainbow trout are the least concern among native trout species due to their life history plasticity, environmental tolerance, large natural and expanded range (through introductions) and resilience.

Description: Coastal rainbow trout are typically silvery in color, with a white belly, black spots on the tail, adipose fin, dorsal fin, and back; spots on the tail radiate in lines (NMFS 2016). There is a pink to rosy lateral band on each side, and the gill covers are usually also pink to purple. Color is highly variable, however, so trout from small streams may be fairly dark on the back with a yellowish belly and orange tips on the fins, while lake-dwelling fish tend to be more silver in color. Coastal rainbow trout rarely exhibit the chrome coloration characteristic of fish that have reared in the ocean. The mouth is large, with the main bone of the upper jaw (maxillary) extending behind the eye; small teeth line the jaws, tongue, and roof of mouth. The tail is only slightly forked, with rounded tips. Other fins are pointed, with a white leading edge and translucent rays. Fin ray counts are as follows: dorsal, 10-12; anal, 8-12; pelvics, 9-10; pectorals, 11-17. Scales are small and highly variable in number: lateral line 110-160, rows above 18-35, and rows below 14-29. Generally, rainbow trout that undertake ocean migration attain larger sizes than inland coastal rainbow trout as a result of feeding opportunities in rich marine waters in the California Current (NMFS 2016). In California's anadromous waters, any rainbow trout greater than 41 cm (16 in.) in length are usually considered to be "steelhead" for management purposes and catch limits, although in productive reaches of river, resident coastal rainbows this size and larger are common. See Moyle (2002) for a more detailed description.

Taxonomic Relationships: The taxon "coastal rainbow trout" in this account refers to all wild rainbow trout that spend their entire life cycle in fresh water and are not part of some other taxon; they are often referred to as resident rainbow trout. Non-migratory rainbow trout in anadromous waters, which are capable of undertaking ocean migrations or of producing offspring that do, are technically included in the relevant steelhead Distinct Population Segment (DPS) accounts elsewhere in this report. However, the large resident rainbow trout populations in tailwaters of dams in the Sacramento and San Joaquin rivers are not included as part of the Central Valley steelhead DPS, even though at least some of them are capable of anadromy (Zimmerman and Reeves 2000).

The National Marine Fisheries Service (NMFS) summarizes these DPSs in California in Williams et al. (2016):

> "The Klamath Mountains Province Steelhead DPS begins at the Elk River in Oregon and extends to the Klamath/Trinity basin in California, inclusive. The Northern California Steelhead DPS extends from Redwood Creek in the north to the Gualala River in the south, inclusive. The Central California Coast Steelhead DPS begins at the Russian River, contains populations in streams tributary to the San Francisco/San Pablo Bay system, and stretches south to Aptos Creek, inclusive. The South-Central California Coast Steelhead DPS starts at the Pajaro River in the Monterey Bay Region and continues to Arroyo Grande in San Luis Obispo Bay. The Southern California Steelhead DPS begins at the Santa Maria River, inclusive, and stretches to the border with Mexico. The

> California Central Valley Steelhead DPS includes all populations in the Sacramento/San Joaquin River system and its delta. *All of these DPSs include only potentially anadromous fish below definitive natural or manmade barriers to anadromy* (pg. 4)."

Rainbow trout upstream of migration barriers are excluded from California's six steelhead DPSs (Pearse and Garza 2015), although many should probably not be because, if isolation was recent, they may still be contributing to downstream populations. Therefore coastal rainbow trout can be defined as self-sustaining rainbow trout populations that are (a) isolated above natural barriers as the result of geologic activity (landslides etc.), (b) isolated above anthropogenic barriers, such as dams, and (c) the result of introductions by people into isolated areas, such as the historically fishless region of the Sierra Nevada. Many of these populations are derived from stocking hatchery fish of diverse strains. Presumably all rainbow trout had steelhead as ancestors, but developed as resident fish as the result of strong natural selection against anadromy.

We follow Behnke (1992, 2002) in using *O. mykiss irideus* to refer to non-redband trout under this designation. In general, "coastal rainbow trout" as used here is a catch-all term to refer to resident rainbow trout that cannot access the ocean and thus are not included in a steelhead DPS. As indicated above, the boundary between steelhead and resident coastal rainbow trout is fuzzy because it is not biologically based, but a distinction of convenience for management. Many different populations of resident rainbow trout presumably had independent origins from steelhead, as well as populations established through introductions. These populations include (a) those in upstream areas above natural barriers in coastal watersheds, (b) those in Central Valley streams upstream of dams and other manmade barriers, and (c) those established through introductions above natural barriers (e.g., in lakes and streams in the Sierra Nevada, Cascades, and Trinity Alps) in native watersheds, and (d) those established in non-native watersheds as a result of introductions.

The distinction between coastal rainbow trout and steelhead is particularly hard to sustain in the Central Valley, especially the Sacramento River. Large populations of rainbow trout exist in reaches such as the Sacramento River between Keswick Dam and Red Bluff, where anadromy is only weakly supported by conditions that favor the resident life history in wild fish, and most steelhead presumably originate in hatcheries (see Central Valley steelhead account).

Another difficult distinction exists in populations of rainbow trout in reservoirs above dams that engage in steelhead-like behavior in tributary streams. Before dams were constructed across Central Valley watersheds, most rainbow trout there were genetically part of a DPS that also included steelhead. Not surprisingly, rainbow trout upstream of these dams have been found to be more similar genetically to one another than they are to fish downstream of the dams in the same watersheds, while also showing watershed-specific traits. This suggests that they are remnants of the original steelhead populations that adapted their life history to reservoir conditions, where reservoirs replace the ocean as the place where fish can grow rapidly to a large size (see next section). It is also possible that at least some of these populations are the result of decades of stocking various strains of rainbow trout, although locally adapted trout/steelhead are remarkably resilient. Pearse and Garza (2015) surprisingly found little evidence that out-of-basin coastal rainbow trout contributed to the genomes of above-reservoir trout, despite a century of introductions of hatchery strains of domesticated fish of mixed geographic sources from rim of the Central Valley (California Hatchery Scientific Review Group 2012). Traits of wild native fish have presumably been selected for under natural conditions and contributed to this result.

In contrast, rainbow trout of all persuasions below Central Valley dams are genetically fairly uniform, showing a strong influence of steelhead introduced from the Eel River in the 1950s (Pearse and Garza 2015). Although individuals from above dam populations may occasionally be washed downstream, these populations are presumably on their own evolutionary trajectories, providing reason not to include them in the relevant steelhead DPS downstream. In addition, the remaining upstream populations may better represent the pre-dam steelhead genome than the introgressed populations present below the dams today, especially in the American River (Pearse and Garza 2015). For further discussion, see Moyle (2002).

Life History: Coastal rainbow trout have a high diversity of life history strategies, which is the principal reason for their success. Strategies vary widely. For example, the classic pattern for resident fish is to spend most of their lives in a short section of stream, perhaps making a short migration (a few meters to a few kilometers) for spawning. Rainbow trout that display a non-anadromous life history but maintain a connection to downstream areas are often capable of producing anadromous offspring; this fact is suspected to be the a major reason for persistence of very small populations of returning adult steelhead to small, coastal streams in South-Central and Southern California (Courter et al. 2013).

In contrast, reservoirs often develop steelhead-like runs of fish that grow in lakes and spawn in tributary streams, using the large reservoir to rapidly increase growth and fecundity (adfluvial life history). Such runs may or may not have been derived from steelhead that became trapped behind dams after their construction over the last century. Additional research is needed on the genomes and life histories of migratory fish in and above reservoirs, including their relationships with resident forms.

Coastal rainbow trout mature in their second or third year of life, spawn 1-3 times, and rarely live more than five or six years. Spawning takes place in spring (February to June, depending on flows and temperatures). However, under artificial flow conditions, such as those found in the tailwaters below dams (e.g. Putah Creek, Solano/Yolo counties), coastal rainbow trout can spawn in winter months (late November-January, K. Davis, pers. comm. 2016). In spring-fed systems, such as the Fall River (Shasta Co.), which lack strong seasonal cues of flow and temperature, spawning can occur from September through June (R. Lusardi, unpubl. obs.). Each female digs a series of redds and buries the fertilized embryos. Embryos hatch in 3-4 weeks (at 10-15°C) and the fry emerge 2-3 weeks later. Fry aggregate in shallow water along shore, and gradually move into deeper water as they grow larger. If they live in riffles or shallow runs, fish may be territorial or partially so, but fish in pools tend to congregate in groups, albeit with some sorting by size.

Diets of stream-dwelling trout are primarily aquatic and terrestrial insects that drift in the water column, although frogs and fish may be consumed on occasion. Benthic feeding also occurs and may be a dominant mode in some rivers, such as the McCloud (Tippets and Moyle 1978). In lakes and reservoirs, they frequently feed heavily on planktivorous fish, such as threadfin shad. Moyle (2002) provides more information on the diversity of life history strategies.

Habitat Requirements: Resident rainbow trout are found primarily in cool, clear, fast-flowing waters. They typically thrive in tailwaters of large dams, but also can easily adapt to inhabiting lakes and reservoirs with ample food. Rainbow trout are among the most physiologically tolerant of salmonids, which is why they are often the only salmonid found in streams that are thermally marginal. For most studies of thermal tolerances of domesticated coastal rainbow trout, a maximum critical thermal threshold of 26°C is assumed (Robinson et al. 2008). However, coastal rainbow trout in California can live in waters that temporarily reach 22.3-33.1°C (Sloat and Osterback 2013) in summer for short periods of time, provided there is sufficient acclimation time and food available to offset higher energetic costs and stress associated with warm water (see Box 1 on bioenergetics in SONCC coho salmon account). Thermal refuges (e.g. upwelling ground water, cool tributary stream mouths, springs) and cooling temperatures at night are also important in marginal thermal habitats. For the most part, rainbow trout in summer prefer temperatures between 15 and 18° C (Moyle 2002) At low temperatures, rainbows can survive under relatively low dissolved oxygen concentrations, although saturation is needed for embryo development and most activities. They also can survive and grow in a wide range of water chemistry, including water with a pH between 6 and 9.

Different life stages have different habitat requirements as defined by depth, water velocity, and substrate (Moyle 2002). Smaller fish generally require shallower water, lower velocities, and less coarse substrates than larger fish. Given a choice, trout in streams live in areas where they can hold in place with minimal effort, while food is delivered to them in nearby fast water. They also require nearby cover, such as downed trees or overhanging vegetation, to protect them from predators.

Distribution: Coastal rainbow trout were originally present in virtually all permanent coastal streams from San Diego north to the Smith River, although almost all resident fish are/were closely related to the local steelhead DPS than to resident fish in other regions. Likewise, coastal rainbow trout were found in most rivers in the Central Valley from the Kern River north to the Pit River system. Resident forms were typically found above barriers difficult to pass by steelhead. Today, due to numerous official and unofficial introductions across California, resident trout with coastal rainbow origins are found in virtually all streams where suitable habitat exists. Their expanded range includes most of the lakes and streams in the once-fishless Sierra Nevada north of the Upper Kern basin as well as lakes and streams in the Cascades and Trinity Alps. For more details, see Moyle (2002).

Trends in Abundance: Wild, naturally spawning resident coastal rainbow trout are undoubtedly more abundant today than they were historically in California. They have been introduced into most suitable waters, including reservoirs, where they are often the dominant species. They are abundant in tailwaters below large dams in valley flow reaches. Starting roughly in the 1950s, increasing emphasis was placed on supporting fisheries with domesticated trout from hatcheries for put-and-take fisheries. In recent times, maintaining such fisheries has been an important activity of the California Department of Fish and Wildlife (CDFW). However, the growing popularity of catch-and-release fisheries for wild trout has resulted in improved management of many streams, by reducing grazing and road impacts, protecting riparian corridors, improving flow regimes below dams, and other actions. Hybridization of locally adapted strains of coastal rainbow trout with hatchery origin fish is often regarded as a problem, but most hatchery strains today survive poorly in the wild, especially in streams, and have limited opportunities to reproduce. "Although the genetic identities of distinct local populations may have been lost in many instances as the result of planting hatchery fish, wild strains adapted to local conditions may persist" (Moyle 2002, p. 280). Therefore, many of the remaining fish above man-made barriers have limited introgression with hatchery or domesticated strains from out-of-basin (see Central Valley steelhead account).

While local populations in urban and heavily agricultural areas may be diminished or even eliminated, total abundance statewide is generally high. However, coastal rainbow trout abundance across California was likely depressed during the ongoing (2012-16) drought. While it is impossible to detail every single population's abundance trends statewide, a smattering of drought rescue information from CDFW and other management partners suggest that less coldwater habitat was available for all species across most of the state for the last five years of drought (2012-2016).

Factors Affecting Status: At one time or another, every factor discussed for other salmonids in this report has reduced local resident rainbow trout populations, including: over-exploitation, water diversions, dams, pollution, poor watershed management (through logging, agriculture, over-grazing, road building), mining, channelization of streams, introductions of alien species, etc. Despite many possible causes of decline, rainbow trout overall continue to thrive throughout California.

Part of the success story of resident rainbow trout is their wide introduction outside their native range across California, North America, and the world. Most of these populations are at least partially, if not wholly, derived from California coastal rainbow stocks. Where introduced, rainbow trout are alien species responsible for the depletion and even extinction of native fishes, especially other trout species (e.g., Lahontan cutthroat trout [*Oncorhynchus clarkii henshawi*] in the eastern Sierra Nevada). They are considered one of the hundred worst invaders in the world by the International Union for the Conservation of Nature (Lowe et al. 2000).

Dams. Dams are a major cause of resident coastal rainbow trout replacing steelhead in many waters, including in cold tailwaters below dams. Above dams, tributary streams at higher elevations fed by snowmelt support resident coastal rainbow trout, while reservoirs themselves often support populations of adfluvial trout. Overall, it can be argued that resident or adfluvial coastal rainbow trout have benefited from dams, despite isolation and fragmentations of populations.

Agriculture. Water diversions, especially for agriculture and municipal use, often reduce the quantity and quality of water available to coastal rainbow trout. Countless pumps, levees, dikes, and other water infrastructure to support drawing water from reservoirs and streams remove cold water inputs to watersheds and often contribute to conditions that are better suited to alien fishes than coastal rainbow trout. Return flows to watersheds are often warmer and include pollutants and sediment that degrade water quality for salmonids, limiting their growth and abundance.

Hatcheries. Most populations of coastal rainbow trout in California were introduced to through stocking hatchery-reared fish, so from the perspective of coastal rainbow trout as a taxon, hatcheries have been beneficial. However, these widespread introductions have displaced and negatively affected native fishes and other aquatic species, such as amphibians, throughout California.

Logging. Both private and public forest lands across California have been heavily logged in the past century, often reducing the quality of rainbow trout habitat but rarely eliminating populations completely. In some parts of the state, current logging practices are well managed, but legacy effects from past unregulated timber harvest may continue to limit coastal rainbow trout populations in some areas. Contemporary logging, along with associated roads and widespread legacy effects from extensive historical timber harvest, has increased erosion rates, increasing sediment loads in streams (Lewis et al. 2004). Increased sediment loads cause pools to fill, embed spawning gravels in fine materials, and create shallower runs and riffles, decreasing the amount of usable spawning and rearing habitat and increasing vulnerability of fish to poachers and predators. Sedimentation is also known to significantly reduce eyed egg survival in coastal rainbow trout at concentrations of fines >30% (Jensen et al. 2009), while removal of trees can reduce canopy cover and shading and indirectly increase water temperature.

Grazing. Heavy livestock grazing throughout the coastal rainbow trout range, particularly cattle in riparian zones in the Sierra Nevada, has degraded stream habitat with broad effects including loss of riparian vegetation, channel incision, and siltation. Loss of riparian vegetation has resulted in higher water temperatures and reduced cover, leaving fish more vulnerable to reduced water quality, disease, and predators. Much coastal rainbow trout habitat exists on Bureau of Land Management and National Forest lands, where grazing has historically or is still permitted. Public lands are far less heavily grazed today, but in areas where grazing still occurs, canyon areas should be prioritized for cattle exclusion to protect areas of hyporheic flow that provide refuge for salmonids during warm summer and fall months with low flows (Boxall et al. 2008). In general, while grazing has probably reduced trout populations in many areas, it has rarely eliminated them, so recovery can be rapid once grazing management is improved. For a more complete account of grazing on trout populations at high elevations, see the California Golden Trout account.

Rural/residential development. Much of the coastal rainbow trout range in California is impacted by rural and/or residential development, especially at lower elevations, reducing carrying capacity of streams for rainbow trout. Residential and municipal users have placed high demands on limited water supplies through diversions and groundwater pumping, which have led to habitat reductions, degradation, fragmentation, and non-point pollution exposure. Water diversions for human uses in rural areas throughout California generally reduce the supply of cold, subsurface flows, which support coastal rainbow trout oversummering habitat. In addition, numerous tributaries are currently listed as impaired water bodies under the Clean Water Act due to high levels of sedimentation, elevated water temperatures, presence of pathogens or contaminants, pharmaceutical products, and poor water quality. Urban development throughout the coastal rainbow trout range has greatly reduced riparian habitat, contributing to reductions in water quantity and quality.

Fire. Most of the coastal rainbow trout range lies within forested areas from moderate to high elevations. Populations can therefore be affected by wildfires. Fires can increase water temperatures in holding and rearing headwater streams, cause landslides, increase sediment loading, and remove shading canopy cover. Large rainfall events can quickly mobilize debris from steep slopes and bury spawning and rearing habitats in headwater reaches. In addition, fire can fragment populations by creating barriers (physical, thermal, etc.).

Transportation. Roads cross most streams containing coastal rainbow trout. Unsurfaced and unimproved roads (mining, logging, and rural/residential access) are abundant in the Sierra Nevada foothills, and Forest Service access roads bisect National Forests throughout California. Many of these roads are sources of fine sediment and pollutant runoff (oil, gasoline) and can significantly degrade water and habitat quality. Culverts associated with road crossings over streams block access to and fragment habitat in many streams, reducing population connectivity and resilience.

Alien species. Alien species, especially non-native salmonid species, such as brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*) can out-compete and prey directly upon juvenile coastal rainbow trout, although the species also frequently coexist because of different reproductive strategies (Kiernan and Moyle 2012). Across the native range of rainbow trout in California, stocking practices have likely reduced populations, in some cases considerably. Predation on coastal rainbow trout, especially by basses (*Micropterus spp.*) and other predators can also cause direct mortality, as can predation by native predators including fish, birds, and sea lions. However, rainbow trout are remarkably adaptable and predation is likely to be a cause of decline only if also associated with habitat alteration.

Harvest. Harvest demands drove California's management of coastal rainbow trout for decades (CHSRG 2012), whereby domesticated strains of rainbow trout from hatcheries were stocked into waters to provide fishing opportunities for the public. More recently, angler preferences have shifted towards catch-and-release fishing for wild trout on most streams, although put-and-take fisheries are favored in urban areas and in reservoirs. For streams, changes in regulations over time have seen reductions in daily bag limits (e.g. 2 fish instead of 5 per angler, per day), or zero retention altogether, to protect the quality of the angling experience and satisfaction. Put-and-take fisheries will likely continue to have a prominent place in fisheries management for rainbow trout in California; this results in some incidental take of wild fish but likely has small impact on wild coastal rainbow trout populations. However, legal catch-and-release fishing for wild fish also results in some low levels of mortality, but the population level impacts of managing a fishery in this way are poorly understood and require further study.

Mining. Nearly every watershed containing coastal rainbow trout in California underwent hardrock, hydraulic, and dredge mining for gold or other metals in the past, which has no doubt reduced carrying capacity for wild trout of many streams. While dredge mining has been banned in California since 2005 (www.wildlife.ca.gov/licensing/suction-dredge-permits) and the days of prospecting in California for gold are past, legacy impacts persist. Mine tailing piles remain throughout many of the Sierra Nevada watersheds, bearing witness to complete turnover of streambeds that occurred on huge scales across the landscape. These alterations have been shown to have fundamentally reduced productivity of watersheds in California downstream of historical mining operations, and likely have reduced spawning habitat for coastal rainbow trout. In addition, old mines continue to leach pollutants and heavy metals into waterways, reducing water quality.

Recreation. Recreational activities in streams and lakes supporting coastal rainbow trout include: angling, boating, gold panning, swimming, hiking, and other outdoor activities. These impacts, especially at the population level, are likely minimal. Intensive motorized boating (e.g., California's major reservoirs such as Lake Shasta, Shasta Co.) may disrupt movement patterns and, potentially, temporary habitat utilization, but this has not been substantiated.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of coastal rainbow trout. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	Low	Dams alter habitat quantity and quality for coastal rainbow trout both positively and negatively and may fragment populations.
Agriculture	Low	Agricultural diversions reduce habitat and water quality.
Grazing	Low	Grazing impacts riparian habitat, but impacts are low.
Rural/ residential development	Low	Demand for limited surface water and groundwater pumping for municipal and other uses reduce available habitat.
Urbanization	Low	Much of coastal rainbow trout range is in rural areas.
Instream mining	Low	Mostly legacy impacts after dredge mining ban in 2006.
Mining	Low	Historical impacts throughout coastal rainbow trout range, legacy impacts may have small impacts in recent times.
Transportation	Low	Dirt roads across range may increase sedimentation.
Logging	Low	Legacy impacts associated with timber harvest.
Fire	Low	More frequent and intense wildfires may reduce local habitat availability.
Estuary alteration	n/a	Coastal rainbow trout, by definition, are separate from steelhead that have access to the ocean.
Recreation	Low	Minimal impacts.
Harvest	Low	Harvest and legal catch-and-release fishing may reduce survival in some areas.
Hatcheries	Low	Many populations of coastal rainbow trout started by hatchery strains (e.g. Sierra Nevada lakes); introgression of above-dam populations and hatchery fish is low.
Alien species	Low	Possibly a limiting factor in some populations in reservoirs or degraded stream habitats suitable for alien species; alien trout species (brook and brown) have most impact across range.

Effects of Climate Change: Similar to species of inland native trout in California, climate change is likely to negatively impact coastal rainbow trout by reducing cold-water habitat. Moyle et al (2013) rated coastal rainbow trout as "highly vulnerable" to climate change impacts. However, coastal rainbow trout are less vulnerable than other species of trout due to their inherent tolerance for high water temperatures and poor water quality, competitive behavior, abundance, and access to diverse habitats that increase their resilience as a species. In general, climate change is likely to reduce the total amount of habitat available to rainbow trout but there will be sufficient cold water to support many self-sustaining, if reduced, populations.

Status Score = 4.7 out of 5.0. Low Concern. Coastal rainbow trout are very widespread in California, occupy a variety of habitats, enjoy significant gene flow among populations, and are highly adaptable. As a result, they face no danger of extinction at this time. Despite the damage to trout streams in California over the past 150 years, coastal rainbow trout continue to thrive in many areas. Climate change will reduce populations but not drive the species to extinction in California.

Table 2. Metrics for determining the status of coastal rainbow trout, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	5	Abundant in California and widely distributed around the world.
Estimated adult abundance	5	Many fish in many populations. Adult abundance probably greater than 500,000 statewide.
Intervention dependence	5	While stream improvements and other activities greatly improve habitat for native and introduced populations, most populations can persist on their own with existing protective laws and regulations.
Environmental tolerance	4	Broad physiological tolerance.
Genetic risk	5	Lots of gene flow among populations.
Climate change	4	Low vulnerability due to widespread populations in diverse habitats.
Anthropogenic threats	5	All threats low except estuarine alteration, which is n/a.
Average	4.7	33/7.
Certainty (1-4)	4	Well-documented.

Management Recommendations: Most ongoing conservation efforts in California for coastal rainbow trout center around increasing reliable quantities and quality of cold water habitat. An increasing focus is to support healthy populations of wild trout for catch-and-release recreational fisheries. In fact, increasing the number of stream miles and lakes devoted to thriving wild trout populations is now a major goal of the California Department of Fish and Wildlife, mandated by state law and implemented by the Wild and Heritage Trout Program. As climate change impacts continue to result in warmer water, reduced summer flows, and increased frequency of large floods throughout California, maintaining existing populations will become harder over time. In addition, there will be continuing conflicts with protecting endangered and native fishes and other aquatic species, such as listed red-legged frog (*Rana draytonii*) and mountain yellow-legged frog (*Rana muscosa*) in California. With climate change likely to reduce the consistency of precipitation patterns across the state and available snowpack in the Sierra Nevada, efforts that protect cool source waters such as springs, meadows, and hyporheic flows into streams should be prioritized and expanded where possible.

Increasing public fishing opportunities for trout around California is and will remain a priority for CDFW, California Trout, and others into the future, and domesticated strains of hatchery coastal rainbow trout will play a role in supporting this priority. However, domesticated hatchery-strain coastal rainbow trout should only be stocked in areas where their existence does not threaten listed species or native fish species. The benefits of increasing access to trout for the public should be carefully balanced and conservatively managed against providing angling for California's other native trout species wherever possible. Common sense approaches to stocking, such as utilizing native species wherever possible in their native range, should be expanded.

Management of rainbow trout in California is complex, and is made even more difficult by the changes humans have wrought on the landscape (such as dams) to support agriculture and urbanization. While steelhead DPSs were created to conserve fish with anadromous life histories, which were in severe decline, genetic information was insufficient at the time to identify the important link between above- and below-barrier populations for conservation (Courter et al. 2013, Pearse and Garza 2015, Williams et al. 2016). However, the latest information indicates that rainbow trout upstream of barriers have a critical role in supporting the anadromous life history of DPSs, with the notable exception of the heavily altered Central Valley populations (Pearse and Garza 2015).

This fact has led to a shift in management and restoration at NMFS that focuses on reconnecting populations of rainbow trout that are currently separated by barriers promoting access to diverse habitats to restore genetic diversity and aid in recovery (e.g. South-Central and Southern California steelhead DPSs, Jacobson et al. 2014, Abadia-Cardoso et al. 2016, Williams et al. 2016). Under such an approach, if adopted by CDFW, there would be no need to differentiate between above- and below-barrier populations of rainbow trout. Rather, the term "coastal rainbow trout" would simply apply to fish that are resident upstream of natural barriers and in lakes and reservoirs with no access to the ocean. Rainbow trout in anadromous waters could be managed as a unit, albeit with significant ESA-listing implications, based on the new information on genetics and life history interactions.

EAGLE LAKE RAINBOW TROUT
Oncorhynchus mykiss aquilarum (Snyder)

High Concern. Status Score = 2.3 out of 5.0. While recent progress has been made, the Eagle Lake rainbow trout (ELRT) does not exist as a self-sustaining wild population because of dependence on hatchery propagation.

Description: This subspecies is similar to other rainbow trout in gross morphology (see Moyle 2002), but differs slightly in meristic counts, especially in having finer scales than coastal rainbow trout. It is also distinctive in possessing 58 chromosomes, rather than the 60 typical of other rainbow trout (Busack et al. 1980). Eagle Lake rainbow trout have many small, irregular black spots along their dorsal surface, especially on the caudal, dorsal, and adipose fins.

Taxonomic Relationships: Snyder (1917) described this trout as a subspecies of rainbow trout, *Salmo gairdneri aquilarum*. However, Hubbs and Miller (1948) examined Snyder's specimens and concluded that ELRT were derived from hybridization between native Lahontan cutthroat trout (presumed to have occupied Eagle Lake prehistorically) and introduced rainbow trout. Miller (1950) later retracted the hybridization theory. Needham and Gard (1959) then suggested that ELRT were descended from introduced or immigrant rainbow trout from the Feather or Pit River drainages. Behnke (1965, 1972) proposed a redband-rainbow hybrid origin, although redband trout are now considered to be rainbow trout subspecies. Busack et al. (1980), in an extensive electrophoretic, karyotypic and meristic analysis, suggested that ELRT were derived either from immigration or an unrecorded introduction of a rainbow trout with 58 chromosomes. The distinctive morphology, ecology, and physiology of this form all point to ELRT being derived from natural colonization from the Sacramento River drainage. Behnke and Tomelleri (2002) speculated that Lahontan cutthroat trout were the original inhabitants of Eagle Lake but that they disappeared during the Pleistocene during an extended period of drought. During a wetter period, rainbow trout managed to invade through an unspecified headwater connection (Behnke and Tomelleri 2002). Recent genetic studies using amplified fragment length polymorphisms (AFLP) DNA techniques suggest that the closest relatives of ELRT are rainbow trout from the Feather River (Stephens 2007, Simmons 2011). Given the relatively recent volcanism and resulting uplift and mountain building in the vicinity of Lassen National Park (near the headwaters of the Feather River), it is plausible that historical wetted connectivity existed between the Feather River and Pine Creek, Eagle Lake's main tributary (R. Bloom, CDFW, pers. comm. 2012).

Life History: Eagle Lake rainbow trout are late maturing (usually in their third year for females) and were historically long-lived, up to 11 years (McAfee 1966). Trout older than five years are rare in the lake today, although individuals as old as 8-9 years have been caught (CDFW, unpubl. data). Historically, the trout spawned primarily in Pine Creek, which flows into the lake on the western shore and, presumably, on occasion, in the much smaller Papoose and Merrill creeks, which feed the southern end of Eagle Lake. Upstream migrations took place in response to snowmelt-fed high flows in March, April, or May. In the Pine Creek drainage, principal spawning areas were presumably gravel-bottomed, spring-fed creeks, such as Bogard Spring Creek, and headwaters in meadows, especially Stephens Meadows, about 45 km from the lake. In the past, it is likely that the trout spent at least their first 1-2 years of life in these stream habitats before migrating to the lake, much like coastal steelhead. However, it is possible some became stream-resident, while retaining the capability of producing migratory progeny, similar to steelhead and other lake-dwelling trout populations, such as Goose Lake redband trout (Moyle 2002). In recent years, progeny of adults transported to the upper basin have been found to be as old as four years. It is also possible that ELRT spawned successfully in the lower reaches of Pine Creek, with fry washing into the lake. In 2010 and 2011, 26 (21 male and 5 female) and 150 adult spawners (60 male and 40 female PIT tagged fish, along with 50 others), respectively, were released above the weir in lower Pine Creek in April. In June, fry (30-40 mm TL) were collected from the trap downstream (P. Divine, CDFW, pers. comm. 2012). It is not known if these fish can survive in the lake.

Yearling ELRT from hatchery plantings grew to about 40 cm by the end of their first year in the lake, 45-55 cm in the third, and up to 60 cm in the fifth year (McAfee 1966). These fish could (at least in the past) apparently reach 3-4 kg and 65-70 cm FL (McAfee 1966). Data from the last 10 years shows that mature females produce an average of 3,300 eggs (Crystal Lake Hatchery, CDFW, unpubl. data, 2009). Rapid growth is the result of abundant forage in Eagle Lake, combined with a delay in maturity until 2-3 years of age. This latter trait has made them highly desirable as a hatchery fish (Dean and Chappell 2005).

The life history of these fish has been significantly altered because access to spawning grounds in Pine Creek has been obstructed since the late 1950s, until recently when the trap was modified to allow passage. Currently, most fish move up Pine Creek in the spring, are trapped at a permanent weir and artificially spawned. The number of adults passing the trap largely depends on if sufficient flow exists in lower Pine Creek. Typically, fertilized eggs are taken to Crystal Lake and Darrah Springs hatcheries where they are hatched and the young reared for 14-18 months. The first generation fish that originate from parents captured in the trap are planted in Eagle Lake at 30-40 cm FL (CDFW, unpubl. data). Since 2011, between 80,000 and 140,000 fish are planted in the south basin of the lake each year, where access for planting trucks is still feasible due to drought conditions. The stocking of greater numbers of trout and 1+ kg "bonus fish" halted in 2011 due to persistence of drought conditions, lower fishing pressure, and to improve the Department's ability to assess fish growth and condition over time. Progeny of the fish captured in the Pine Creek trap are also reared in other hatcheries in California and planted widely in reservoirs (Carmona-Catot et al. 2011). During high water years between 1959 and 1994, ELRT were periodically able to pass the fish weir near the mouth of Pine Creek volitionally but only during extreme high water events (CDFW 2015). The weir, however, was redesigned in 1995 ceasing all passage. In 2012, the weir and fish trap were again redesigned to promote passage when sufficient flows exist in Pine Creek and a recent conservation agreement strategy between USFWS, CDFW, and Lassen National Forest aims to improve fish passage to Pine Creek and restore natural spawning and rearing of Eagle Lake rainbow trout (CDFW 2015). A stated goal of the conservation strategy is to operate the Pine Creek fish ladder to encourage upstream passage of ELRT and reestablish a stream population.

All trapped fish are marked in order to prevent sibling crosses (reduce inbreeding), avoid using fish that have spent more than one generation in the hatchery, and to select for longer-lived fish to compensate for longevity reductions that may have been caused by past hatchery practices (R. Elliott, CDFW, pers. comm. 1998). Beginning in 2001, no ELRT were planted that experienced more than one generation in the hatchery (P. Divine, CDFW, pers. comm. 2012). In 2014, CDFW began collecting and genetically testing all trapped fish in order to further reduce the potential for inbreeding and to maximize effective population size, while also stocking the lake with fish exhibiting differences in run-timing in order to maximize run-timing diversity (P. Divine, CDFW, pers. com. 2016). Analysis will begin on these samples in 2017. Formerly, a hatchery program for rearing ELRT was maintained at Mt. Shasta Hatchery by using wild-caught fish as broodstock for one generation. The progeny of these fish were originally planted widely in reservoirs of the state and used as a source for broodstock in other hatcheries in California, as well as elsewhere in the western U.S. Eagle Lake rainbow trout are prized because of their delayed maturity, rapid growth and longevity. As noted, all fish reared in hatcheries for planting in Eagle Lake are first generation ELRT from the Pine Creek trap, although fish from hatchery broodstock were planted in combination with first generation fish from the Pine Creek trap into Eagle Lake in the past (P. Divine, CDFW, pers. comm. 2009).

Despite this long (60+ year) history of hatchery selection, there is evidence that ELRT can still spawn successfully in Pine Creek. Fish that were trucked to the upper reaches of Pine Creek in the 2000s produced young, which survived and grew for two years. A thorough survey of Bogard Spring Creek revealed the presence of at least 170 ELRT in 2007, with most fish lengths between 105 and 150 mm FL; in 2008, only 25 ELRT were captured with lengths between 130-165 mm FL, while 34 ELRT were captured in 2009 (Figure 1; Carmona-Catot et al. 2010, 2011). These fish survived and grew despite the presence of about 5,300 brook trout in the same reach of stream in which they were found (see management for details). There is some evidence that two year old fish will try to migrate downstream to the lake during periods of high spring flow (P. Moyle, unpubl. observations, 2006). In spring 2009, an ELRT was captured in Pine Creek at 800 meters downstream from the confluence with Bogard Spring Creek. This fish was fin clipped in September 2008 in Bogard Spring Creek (Moyle and Carmona, unpubl. data). In 2011, a single male ELRT managed to migrate the entire distance from the weir to the upstream spawning areas (T. Pustejovsky, UC Davis, pers. comm. 2011).

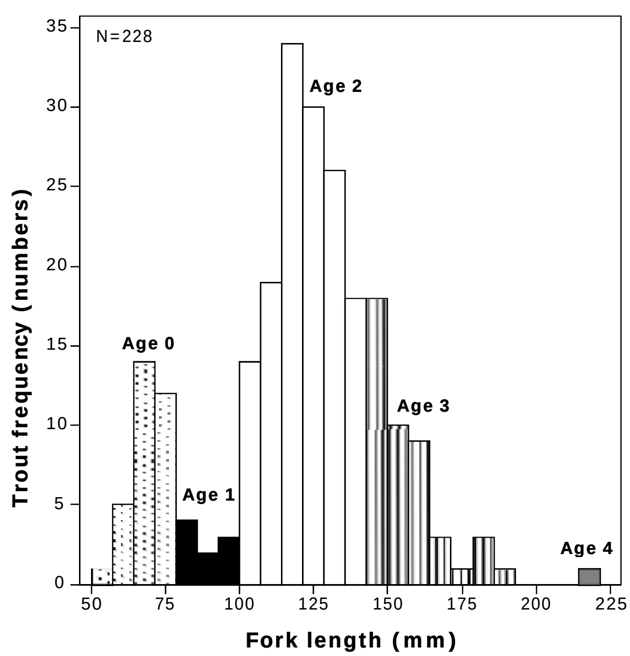


Figure 1. Fork lengths and ages of Eagle Lake rainbow trout in Bogard Spring Creek in 2007, 2008, and 2009; age distributions inferred from scales of 71 fish. From: Carmona-Catot et al. 2011, Fig. 2 pg. 9.

The diet of ELRT varies with age and season. Newly planted trout in their first year in the lake feed mainly on zooplankton, including *Daphnia* spp. and *Leptodora kindti*, as well as on benthic invertebrates, especially leeches and amphipods. By August, most of the trout switch to feeding on young-of-year tui chubs (*Gila bicolor*) (King 1963, Moyle 2002, Eagles-Smith 2006).

Habitat Requirements: Eagle Lake rainbow trout spend most of their life in Eagle Lake, a large (24 km long by 3-4 km wide), highly alkaline lake. The lake consists of three basins: two of them average 5-6 m deep in most years, but drop to 2-3 m during severe drought and the third averages 10-20 m, with a maximum depth of about 30 m. The shallow basins are uniform in their limnology and water temperatures may exceed 20°C in the summer. The deep basin stratifies so, in late summer, most of the trout are in the deeper, cooler water of this basin. Otherwise, they are found throughout the lake. In 2015, the lake reached record low elevations of less than 1552 m (5,092 feet) above sea level, with the northern two basins averaging less than one meter in depth. In 2016, the lake levels remained historically low, but depths increased about .1 m in the two northern basins. The effects on the ELRT population in the lake are unknown.

During the summer, upper Pine Creek is a cold, spring-fed stream, flowing at .03-0.14 m^3/s through meadows and open forest, with modest gradients. Bogard Spring Creek is also a spring-fed creek, with flows of 0.01-0.02 m^3/s. The meadow streams have deep pools and glides with deeply undercut banks, providing abundant cover for trout. The Pine Creek watershed is described in detail by Pustejovsky (2007). Unfortunately, the trout present today in the Pine Creek watershed are almost entirely alien brook trout (Carmona-Catot et al. 2010).

Environmental tolerances of ELRT are high for a trout. In Eagle Lake, they live in highly alkaline water (pH 8.4-9.6), in which dissolved oxygen is usually at or close to saturation (except in the hypolimnion of the south basin during months of thermal stratification). ELRT have evolved to tolerate the high alkalinity water of the lake as well as the spring-fed waters of Pine Creek, their only accessible spawning tributary. They have been observed foraging in shallow water at temperatures of 22-23°C but generally retreat to deeper, cooler areas (< 20°C) as lake temperatures increase. The requirements of spawners and juveniles in streams have not been well studied but are presumably similar to those of other rainbow trout (see Moyle 2002).

Distribution: Eagle Lake rainbow trout are endemic to Eagle Lake (Lassen Co.) and its main tributary, Pine Creek. They have been planted in numerous waters throughout California, where they are maintained from hatchery stocks originating from trout captured at the weir at the mouth of Pine Creek. In the past, hatchery trout have been exported to other states and Canada. It is unlikely that naturally reproducing populations of genetically 'pure' Eagle Lake trout are present in any of these waters, although supporting data are largely absent.

Trends in Abundance: Naturally spawned ELRT were once abundant in the lake. According to Purdy (1988), "In the spring months of the 1870s and 1880s, when trout were spawning, huge quantities were being caught. It was not unusual to hear that wagon loads of trout, some weighing as much as 600 pounds, were being brought into Susanville where they were sold at local markets for twenty-five cents a pound" (p. 14). This exploitation occurred at the same time as extensive logging in the drainage, heavy grazing of the basin's meadows, and the first construction of railroad grades and roads across meadows and streams, all of which altered stream hydrology and morphology. When the ELRT was described by Snyder (1917), he noted numbers were low. Although commercial fishing for trout was banned in California in 1917, ELRT populations remained low, presumably because of the poor condition of Pine Creek and the establishment of predatory largemouth bass and brown bullheads in the lake. By 1931, trout were scarce in the lake and Pine Creek (Snyder 1940).

During the 1930s, trout populations were further stressed as lake levels dropped dramatically when diversion of water through Bly Tunnel combined with prolonged drought to reduce spawning access to Pine Creek. In 1939, biologists with the Lassen National Forest expressed concern over impoundments further reducing flows of drought-stricken Pine Creek (Pustjevoksy 2007). Meanwhile, logging, railroad construction, and other human alterations to the basin further degraded the Pine Creek watershed. Fortunately, high alkalinities brought on by dropping lake levels also eliminated bass from the lake, although bullheads persisted into the 1970s. Even with the return of wetter conditions, the trout population showed little sign of recovery. In 1949 and 1950, CDFW collected 35 and 75 adult ELRT, respectively, from the mouth of Pine Creek, spawning them for hatchery rearing (Dean and Chappell 2005). The 258 progeny from the 1949 fish were planted in Pine Creek, where brook trout had recently become established, but probably did not survive. The spawning of fish in 1950 was more successful and the hatchery-reared progeny were planted in the embayment at the mouth of Pine Creek. From 1951-1958, some artificial propagation also took place, although the records are not clear as to how many fish were produced (Dean and Chappell 2005). Prior to hatchery propagation, trout presumably persisted only because occasional wet years permitted successful spawning despite degraded stream channels and the presence of brook trout in the spawning reaches of Pine Creek (McAffee 1966). It is possible that these actions by CDFW biologists prevented extinction of ELRT, although, based on recent genetic evidence, a small component of the population may have been able to migrate upstream during larger flow events until all access to upstream areas was blocked in 1995 (Carmona-Catot et al. 2011).

In 1959, an egg taking station was built at the mouth of Pine Creek, including a wooden weir/dam to block upstream passage of most fish (Dean and Chappell 2005). Regular trapping operations began in 1959, when 16 trout were captured and spawned; in the next five years the numbers captured varied from 45 to 391 (McAfee 1966). From 1959 through 1994, a few trout were able to make it over the barrier during wet years, allowing some potential for natural spawning (Pustejovsky 2007, Moyle, unpubl. data). It is unknown, however, if spawning was successful, if progeny survived in degraded stream habitats and in the presence of abundant brook trout, or if any outmigrants during this period were able to return to the lake.

In 1995, the weir was rebuilt to prevent erosion and upstream movement of all ELRT (Pustjevoksy 2007) based on the assumption that adults migrating up Pine Creek would become stranded as the lower portions dried and would be lost to the lake population and fishery. The spawning of ELRT then became entirely under human control; at present, eggs and milt are stripped from the fish at the egg taking station. The embryos are then transported to Crystal Lake Hatchery and distributed to other hatcheries across California (Carmona-Catot et al. 2011). To provide fish for planting, hundreds of trout are trapped each year and between 1 and 6 million fertilized eggs per year are taken for hatchery rearing. In 2009, 1,737 females were spawned, producing 5,985,880 eggs for the hatchery, while in 2008 the take was 2,757,420 eggs, and in 2007 1,113,980 eggs were taken (P. Divine, CDFW, pers. comm. 2009).

Passage of California Assembly Bill 7 (AB-7) in 2005 required the CDFW to increase production of native trout forms in hatcheries, thus the incremental increase in egg take from 2007-2009. The egg quotas are developed every year by CDFW hatchery personnel in order to achieve the broodstock hatchery and statewide goals (Carmona-Catot et al. 2011). There is no recent evidence (without studies) of natural reproduction contributing to the lake population; the fish captured by anglers usually show signs of time in a hatchery environment, such as fins with distorted rays or missing and/or eroded fins. The trap was modified in 2012 in order to allow passage of adults, a significant stride toward restoring some level of natural reproduction in the population (P. Divine, CDFW, pers. comm. 2012). The CDFW stocked ca. 1,000 "half pound" fish in Pine Creek intermittently prior to 2006, ostensibly for the purpose of experimentally reducing brook trout abundance through predation (Dean and Chappell 2005). However, no studies were conducted to confirm that this practice had the desired effect. Subsequent sampling suggests that few of these fish persisted for long in the creek (Carmona-Catot et al. 2011).

Actual population size of trout in Eagle Lake has not been studied, but it is presumably dependent on the stocking allotments every year. Creel censuses indicate that catch per hour from 1983 through 2007 ranged from 0.2 to 0.6, with a mean of 0.3, while average length of fish caught increased over the years (Carmona-Catot et al. 2011). The number of mature females captured at the trap while migrating and spawned by the CDFW ranged from ca. 600 to 1,700, although no estimates were made of size of the entire spawning run.

Genetic studies provide some insights into minimum population sizes in the lake. Simmons (2011) found individuals in the lake population had an F_{IS}, or inbreeding value, of 0.064, significantly higher than zero, although no genetic evidence of a bottleneck was detected. The effective population size (size of breeding population) was estimated at 1,125 fish, with indications in all years of a fairly large population contributing to reproduction. Given the presumed small number of fish used to establish the original hatchery-based population, it is interesting that no genetic bottleneck was detected. A supposed bottleneck could have been masked by the number of generations that have passed since the bottleneck and/or efforts of the hatchery breeding program to maximize genetic diversity (by breeding as many individuals as possible), as seen in the population's now high effective population size. It is also possible that the population left in the lake in the 1950s was larger than trapping efforts on Pine Creek indicated and multiple years of naturally spawned fish contributed to the initial hatchery stock. The slight inbreeding value is still of concern and worth monitoring, although it is comparable to levels found in other lake-stream systems in the region such as Goose Lake (Simmons 2011).

Overall, the population appears to be stable because it is maintained by hatchery production, which may be selecting against fish capable of reproducing naturally. For example, Chilcote et al. (2011) show that wild populations of three species of anadromous salmonids from the Pacific Northwest have greatly reduced ability to remain self-sustaining when fish of hatchery origin are also present. There is ample evidence that hatchery rearing has an impact on the genetics and behavior of fish released into the wild, affecting their ability to persist (e.g., Waples 1999, Araki et al. 2007, 2008, Kostow 2008). Recent evidence suggests that fitness reductions may not just be limited to fish raised in the hatchery but, instead, continues into subsequent generations (Araki et al. 2009).

Factors Affecting Status: Degradation of the Pine Creek watershed and the establishment of brook trout in historical spawning areas are the greatest historical causes of the near-extinction of ELRT. The watershed was severely altered as the combined result of intensive harvest, logging, grazing, diversions, and railroad and road building among other threats (Carmona-Catot et al. 2011). These factors do not operate independently but, instead, must be viewed in aggregate, along with other less pressing threats (Table 1), as cumulative and synergistic watershed impacts.

Agriculture. In the past, Eagle Lake was viewed as a potential source of water for the otherwise arid agricultural region around Susanville and the Honey Lake Basin. This resulted in the construction of Bly Tunnel, which was completed in 1923, to send Eagle Lake water into Willow Creek for use in crop irrigation. This project largely failed to deliver the water promised. During the 1930s, lake levels dropped as the result of diversion of water through the tunnel in combination with a severe, prolonged drought. Although it was blocked off with a concrete plug in 1986, the tunnel continued to passively leak, through an eight-inch bypass pipe in the plug, 0.34 cubic m/s of Eagle Lake water into Willow Creek for downstream water users. Due to lack of surface flow diversion, some questions remained as to whether the water was coming directly from Eagle Lake or was, instead, percolating from groundwater into the tunnel. Water chemistry analysis revealed that most of the leakage was Eagle Lake water because of its unique chemical similarity to water sampled directly from Eagle Lake (Moyle et al. 1991). Based upon a position paper issued by the California Department of Fish and Wildlife to the State Water Resources Control Board in late 2011, the Bureau of Land Management, who administers the lands surrounding Bly Tunnel, closed the pipe in February 2012, thus eliminating direct discharge of Eagle Lake water via Bly Tunnel.

Grazing. Livestock grazing in the Eagle Lake basin started in the mid-1800s and was unregulated until 1905. Past grazing impacts to the Pine Creek watershed were substantial but are now greatly reduced because of improved management (Pustejovsky 2007; Carmona-Catot et al. 2011). Currently, about 35% of the Pine Creek watershed, and most of the perennial reaches, are protected from grazing through fencing and land management (CDFW 2015). However, the legacy effects of past grazing continue, especially in the lower 40 km of Pine Creek, where the streambed has downcut and become enlarged in places, much of the riparian vegetation has been removed, and riparian meadows have presumably become drier, making them more likely to be invaded by sagebrush and similar xeric vegetation. Although streamflow records are lacking, it is likely that Pine Creek flows have also become more intermittent during summer, with spring flows decreasing more rapidly after snowmelt. At present, the lower creek (below Highway 44) usually stops flowing in late May or early June. The legacy effects of past grazing practices may have contributed to this altered hydrological regime; however, habitat conditions in recent years have been steadily improving (Pustejovsky 2007).

Rural/residential development. Eagle Lake has a number of residential tracts on its shores that depend on groundwater pumping (connected to lake levels) for water supplies. Although the potential connection between aquifer pumping and lake levels is poorly understood, the impacts may be substantial (especially during drought periods). Leakage of septic tank effluent into the lake is also a potential water quality problem. This was resolved in 2007 at Spalding Tract, with the development of a wastewater treatment facility. Wastewater is now diverted to evaporation ponds in Spalding Tract, Stones Landing, and South Shore campgrounds, which may result in significant loss of ground water in the basin, potentially exacerbating low lake levels during drought periods.

Transportation. Past road and railroad building to support historical and ongoing logging activities (see below) negatively affected habitat conditions and fish passage in Pine Creek. Culverts created barriers to upstream fish migration and road or railroad crossings created constriction points, which may have altered stream hydrology. Wet road crossings contributed to stream bank erosion and sediment input. The more recent construction of State Highway 44, parallel to the railroad, forced Pine Creek through several culverts. The combination of culverts and channelized stream created a nearly impassible velocity barrier for spawning ELRT. All potential barriers created by roads or other infrastructure have been removed or modified in lower Pine Creek. The U.S. Forest Service's Eagle Lake Ranger District (ELRD) is planning on implementing road restoration actions adjacent to Pine Creek in order to restore and promote watershed processes through treatment of unauthorized roads or decommissioned roads that have historically contributed to degraded aquatic conditions (CDFW 2015). Remaining barriers in upper Pine Creek include an abandoned USGS gauging weir near highway 44 and the fish ladder at the 31N25 forest road crossing. Additionally, a ford near road 33N33 may also prevent passage, particularly during low flow periods. The California Department of Transportation (CalTrans) historically diverted a portion of water from Bogard Springs to a rest stop along highway 44. However, in 2014, CalTrans in cooperation with Lassen National Forest and the Eagle Lake Ranger District ended their diversion and now rely on a groundwater for municipal use at the rest stop. Returning the diversion back to Bogard Springs Creek will likely improve aquatic habitat for rearing ELRT.

Logging. Timber harvesting officially began in the Lassen National Forest in 1909, although the highest production took place in the 1970s and 1980s. The direct effects of timber harvest on stream habitats and flows may have been minimal because of the rapid infiltration capacity of the volcanic soils of the region, which reduces erosion rates (Platts and Jensen 1991). However, the roads constructed to facilitate logging were (and generally still are) very erosion-prone. Railroad lines were constructed across the Pine Creek drainage in the 1930s and 1940s to support logging activities, which restricted instream flows and led to channelized streambeds. Timber harvest is still very common in the area and the road networks utilized to support logging may serve as source inputs of sediments into streams.

Fire. Fires are common in the dry, heavily altered forests of the Eagle Lake watershed. The effects of fire on Pine Creek and its fishes have not been documented, but the potential exists for severe damage to the upper watershed, with subsequent erosion and perhaps direct mortality of fish in small streams. Historical photos and surveys documenting stand densities and sizes of the area show open stands of large conifers, with little understory or ladder fuels prior to fire suppression and logging in the Eagle Lake basin (P. Divine, CDFW, pers. comm. 2012). Current forest conditions are quite different, with increased stand densities and widespread growth of firs, which are not well adapted to fire and serve as ladder fuels (J. Weaver, CDFW, unpubl. obs.). This change in forest structure may increase the risk of high intensity, catastrophic fires, especially when coupled with predicted climate change outcomes, which may have dramatic impacts on riparian habitats and stream hydrodynamics.

Recreation. The major use of Eagle Lake and its watershed is increasingly for recreation, much of which is focused on the widely popular recreational fishery ELRT support. The impacts from recreational angling, other than from direct harvest, which is closely regulated by a two-fish bag limit per day, are likely minimal. Other recreational impacts may include off-road vehicle use.

Harvest. As noted, in the 19th century, ELRT were once heavily exploited by a commercial fishery, which probably contributed to their initial decline. Since the 1950s, however, demand to support the lake sport fishery has been the principal reason its population has been maintained. However, a high percentage of the trout produced are planted in places other than Eagle Lake and the actual carrying capacity of the lake for rainbow trout is not known. It is possible that planting fewer fish would result in higher survival rates and more rapid growth rates. If a run becomes re-established in Pine Creek, the trout fishery in the creek will have to be managed in ways that do not negatively affect recruitment to the lake. In 2012 and 2013, the number of ELRT stocked into Eagle Lake was reduced by 20,000 to improve quality/condition of ELRT in the lake (P. Divine, CDFW, pers. comm. 2013).

Hatcheries. Eagle Lake rainbow trout continue to be dependent on hatchery production for survival (Moyle 2002). However, the recent fish passage improvements and the conservation strategy (2015) produced by CDFW, USFWS, and the USFS aims to improve natural spawning and rearing of ELRT in Pine Creek by promoting passage at the fish trap and a self-sustaining ELRT stream population. Successive dry years, however, have hampered recent attempts, as sufficient flow for upstream adult passage in lower Pine Creek has only occurred for short time periods or has been non-existent (e.g., 2014) (P. Divine, CDFW, pers. comm. 2016). Prior to the 1950s, ELRT presumably persisted only because occasional wet years permitted access to upstream spawning areas through degraded stream channels and because ELRT were exceptionally long-lived. A potentially negative outcome of hatchery reliance is that fish are being selected for survival in the early life history stages in a hatchery environment, rather than in the wild, perhaps for early spawning (as has happened in steelhead, Araki et al. 2007). In addition, fish may have been directly selected for large sizes for planting the lake (Carmona-Catot et al. 2011) or disproportionally selected for early run-timing to ensure hatcheries could meet production quotas. However, sizes of angler-caught fish appear to be fairly static or slightly increasing over time (with average size over 43cm, Figure 2) and maintaining run-timing diversity has recently been prioritized (P. Divine, CDFW, pers. comm. 2016). Eggs taken from spawned fish at the Pine Creek Trap are sent to several hatcheries for rearing and then stocking into recreational waters. Crystal Lake Hatchery and Darrah Springs Hatchery rear fish to stock back into Eagle Lake. Darrah Springs also has a broodstock select program and rear these selected fish for 1.5 to 2 years. They are then transferred to Mt. Shasta Hatchery where they are used for production broodstock for statewide hatchery programs.

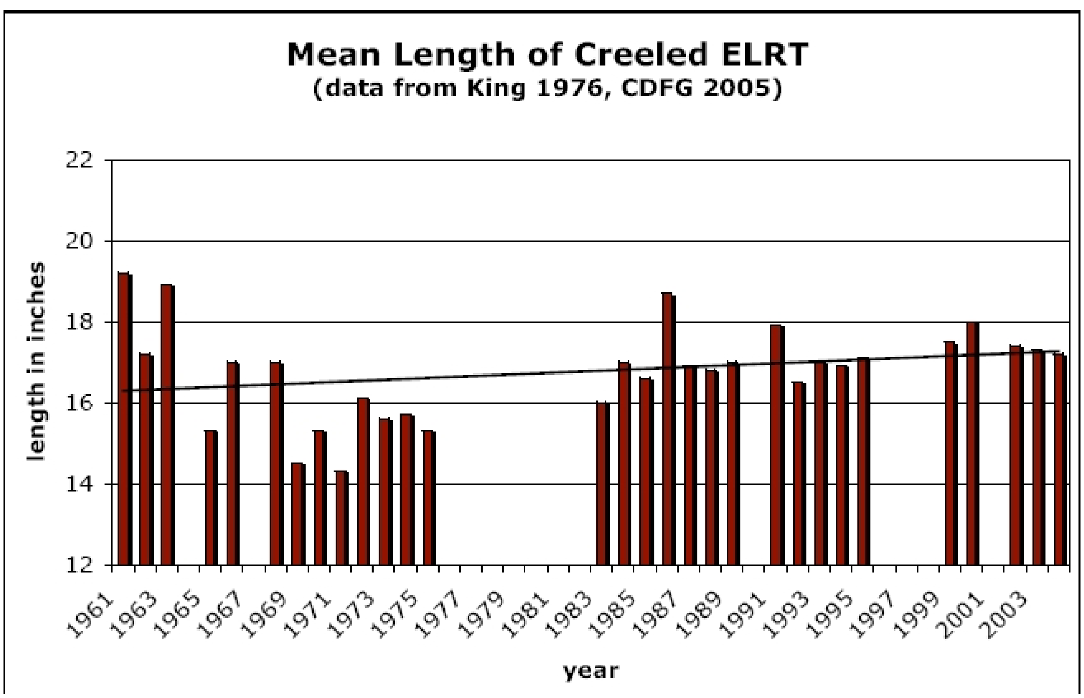


Figure 2. Mean lengths (in inches) of Eagle Lake rainbow trout caught by anglers, 1961-2005. From: Pustejovsky 2007, adapted from King 1976, CDFG 2005, Fig. 4. pg. 17.

Genetic changes to ELRT have likely occurred as the result of continued hatchery selection, which may reduce the ability of trout planted in the lake to spawn naturally and produce young that can survive in streams or retain the predisposition to outmigrate back to the lake. Complete dependence on hatcheries for maintaining the species is undesirable because survival of the species then becomes dependent on vagaries of hatchery funding and management. Disease in hatcheries, loss of adaptation for life in the wild, loss of life history diversity, and potential inbreeding further threaten survival. Hatchery impacts may be particularly detrimental to a species with notable longevity (e.g., possibly eliminating the adaptation of ELRT toward a 10+ year life span, which has likely served as a buffer against extended periods of drought and periodic lack of access to spawning grounds). National Marine Fisheries Service guidelines indicate that a salmonid population dependent on hatchery production cannot be regarded as viable in the long-term (McElhany et al. 2000), a policy supported by recent studies (e.g., Chilocote et al. 2011). Recent fish passage improvements at the Pine Creek fish trap should help improve genetic integrity by allowing for expression of the adfluvial life history, and eventually restore a stream population to counter some of the potential negative effects associated with hatcheries.

The Pine Creek Coordinated Resource Management and Planning (CRMP) group (Pustejovsky 2007) has focused on restoration actions to provide for natural spawning of ELRT in Pine Creek for the past 25 years. In addition, the ELRT conservation strategy (CDFW 2015) aims to restore natural spawning and rearing ELRT in Pine Creek through various means, including habitat improvements, passage past the Pine Creek fish trap, and eradication of introduced brook trout. These efforts, if carried out completely, will result in a stream again capable of supporting a self-sustaining, wild population of ELRT. While hatchery production to sustain the trophy fishery has historically been regarded as a higher priority than re-establishment of a wild population (Dean and Chappell 2005), management shifts in recent years are increasingly focused on restoring a wild population, which is likely to happen only if brook trout are eliminated from Pine Creek so high production and survival of ELRT juveniles can be assured (P. Divine, CDFW, pers. comm. 2012).

Exotic diseases, which could be introduced in hatcheries or the lake by hatchery-reared fish, potentially threaten the survival of the lake's ELRT population. However, hatchery protocols require routine examination of fish and water quality to reduce the threat of disease, and ELRT are reared at two separate facilities in a redundant system in the event that disease outbreak affects one hatchery (P. Divine, CDFW, pers. comm. 2013).

Alien species. Many different species have been introduced into Eagle Lake in the past but none have persisted because of the lake's alkalinity. Nonetheless, the lake's large size and accessibility make it possible that other species will be introduced illegally and eventually alter its ecology. Ironically, introduced species are most likely to become a problem if lake levels rise and alkalinity decreases, as happened in the early 1900s, when largemouth bass and brown bullhead became abundant in the lake. The only alien species that persists in the drainage is brook trout, which was introduced into Pine Creek in the 1930s to 1949 to increase angling opportunities (Dean and Chappell 2005), and stocking of brook trout ceased in 1950 (CDFW 2015). Currently brook trout exhibit extremely high densities in Pine Creek on the order of 30,000 fish/ha, which are some of the highest densities recorded in California. Predation and competition by brook trout in Pine Creek may prevent reestablishment of ELRT, so a program to eliminate this species from the watershed is needed. UC Davis has conducted significant electrofishing over several years with CDFW and the US Forest Service (2006-2011) in an attempt to reduce the brook trout population in upper Pine Creek, and, more recently, (2012 onward) CDFW conducted electrofishing removals in the headwaters of Pine Creek (2013-2014) with assistance from USFS, Trout Unlimited, and Coordinated Resource Management Partners (CRMP) members. However, high densities of brook trout remain, with recent estimates of 4,000 fish/mile (P. Divine, CDFW, pers. comm. 2016). Plans to use piscicides to eradicate brook trout from the upper Pine Creek watershed (upstream of highway 44) are currently under development with the goal of conducting such actions during the fall of 2018 (A. Jensen, CDFW, pers. comm., 2016). The high densities and biomass of brook trout in upper Pine Creek indicates good capacity for rearing ELRT in large numbers in the absence of brook trout (Carmona-Catot et al. 2010, 2011), with the potential for contributing wild fish back into the lake population.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Eagle Lake rainbow trout in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	n/a	
Agriculture	Low	Bly Tunnel once diverted water for agriculture; it closed in 2012.
Grazing	Medium	This was a major historical cause of watershed degradation but recent actions have substantially reduced impacts from grazing.
Rural /residential development	Low	Septic tank effluent and groundwater removal may pose ongoing threats; many septic issues were resolved with recent construction of wastewater treatment plants; however, diversion to evaporation ponds may negatively affect lake levels. CalTrans ended their diversion from Bogard Springs in 2014.
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Culverts (now fixed) have been past barriers to migration but roads continue to affect Pine Creek and lake (sedimentation, erosion, pollutants, etc.).
Logging	Medium	Major activity in watershed.
Fire	Low	Has potential to negatively impact entire Eagle Lake basin, especially with risk of more frequent and severe fires.
Estuary alteration	n/a	
Recreation	Low	Recreation is a major human use of the basin; impacts (other than the recreational fishery) to ELRT are unknown.
Harvest	Medium	Major impact in past; trophy fishery drives management; current fishing regulations in place to manage harvest rates.

Hatcheries	Medium	Almost all fish have been produced in hatcheries for 60+ years; however, ELRT hatchery operations focus on minimizing artificial selection processes and recent modifications of Pine Creek weir should help improve passage and promote a naturally reproducing population; hatchery diseases possible threat.
Alien species	High	Brook trout dominance in Pine Creek watershed is a major barrier to restoration and establishment of self-sustaining wild ELRT population; alien diseases are a possible threat.

Effects of Climate Change: Climate change is likely to have two major impacts on the Eagle Lake watershed: decreased stream flows and changing lake conditions. Reduced snowpack in the mountains surrounding the Pine Creek watershed will presumably reduce the output of springs that feed Pine Creek. The magnitude of this effect, however, will depend on the timing and amount of rain and snowfall and how well meadows are managed to increase their ability to retain water and release it during summer months. Reduced inflow into the lake could potentially increase alkalinities to lethal levels for trout although, if average precipitation remains roughly the same, the lake should maintain itself. Unfortunately, the lake is at near-record low levels and has been receding for several years, so changing water chemistry is an increasing concern as the ongoing drought continues. Surface temperatures of the lake could potentially increase 2-3°C but, presumably, a cold water refuge for trout will continue to exist in the deepest basin of the lake. If climate change produces extended droughts that dry Pine Creek early or for longer periods of time, resulting in increased lake alkalinity and temperatures, ELRT could be driven to extinction in its native range, relegating it to a hatchery fish. Fires, coupled with predicted climate change impacts, may become more frequent and catastrophic, especially in the dry headwaters of the basin and may interfere with ongoing and planned restoration efforts in the Pine Creek watershed. For these reasons, Moyle et al. (2013) scored the species as "critically vulnerable" to climate change and threatened with extinction by 2100 without human intervention.

Status Score = 2.3 out of 5.0. High Concern. While this score reflects improved understanding of ELRT genetics, the subspecies is likely to experience further genetic change and become a semi-domestic hatchery fish if actions to restore a naturally spawning population are not fully implemented. Recent passage improvements at the Pine Creek fish trap are significant and will greatly aid in the re-colonization of ELRT to the upper watershed. Remarkable progress has also been made in restoring stream habitats and natural spawning in the past 5-10 years with numerous other conservation oriented plans outlined in the ELRT conservation strategy. However, continued restoration is needed, particularly regarding the elimination of brook trout from the Pine Creek watershed. Despite genetic monitoring, marking individual fish and efforts to select ELRT from different run-timing intervals, continued hatchery selection is likely to select against the ability of ELRT to maintain a natural life history. Stochastic events, such as elimination of hatchery or lake stocks through a disease epidemic, severe drought, and illegal introductions of invasive species, parasites, or other factors put ELRT at high risk in its native habitat given that they are endemic to only one watershed. Eradication of brook trout coupled with successful re-colonization of ELRT in the Pine Creek watershed will significantly improve ELRT life history diversity and improve species resilience.

A petition for federal listing as a threatened species was rejected by the USFWS in 1994 [*Federal Register* 60 (151) 401: 49-40150]. Two similar petitions were filed with the USFWS in 2004, but in both cases, the petitions were rejected due to insufficient information. However, the USFWS issued a 90-day finding in 2012 [*Federal Register* 77 (172) 54548-54553] indicating listing may be warranted and performed a 12-month review to gather additional information and make a status determination. A 12-month finding decision of "listing is not warranted at this time" was issued in 2016 [*Federal Register* 81 (129) 43972-43979]. The ELRT is regarded as a Species of Special Concern by the California Department of Fish and Wildlife, and as an R5 Sensitive Species by the U.S. Forest Service. The American Fisheries Society lists it as "Threatened," while NatureServe lists it as "Critically Imperiled" (Jelks et al. 2008). Eagle Lake is a designated Heritage Trout Water (one which supports a fishery for native trout forms in their historical range), managed under CDFW's Heritage and Wild Trout Program.

Listing under either federal or state ESA, while potentially justifiable, is not desirable because progress is being made toward their conservation and management. Listing could inhibit the ability of agencies or local conservation groups to efficiently implement restoration tasks by increasing permitting delays or disallowing certain activities intended to benefit the species. Nevertheless, it is important to underscore the need to connect habitat restoration with re-establishment of a wild population, provide additional incentives to eradicate brook trout, and continue to address other stressors to avoid their listing.

Table 2. Metrics for determining the status of Eagle Lake rainbow trout in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	1	Endemic only to Eagle Lake watershed.
Estimated adult abundance	4	Includes hatchery fish.
Intervention dependence	1	Persistence depends on trapping fish for hatchery spawning, rearing and restocking annually, and allowing passage on Pine Creek.
Environmental tolerance	4	One of most tolerant, long-lived forms of rainbow trout.
Genetic risk	3	Although operated to maximize diversity and minimize artificial selection processes, hatchery rearing has presumably altered genetics; possible selection against longevity and fitness in the wild is of concern, particularly for stream forms; accidental hybridization in hatcheries possible.
Anthropogenic threats	2	One High and five Medium threats.
Climate change	1	Rated critically vulnerable in Moyle et al. (2013).
Average	2.3	16/7.
Certainty (1-4)	4	Well documented.

Management Recommendations: The management of ELRT is an ideal opportunity to institute principles of adaptive management, where management actions are treated as experiments to inform future management (Carmona-Catot et al. 2011). The first step in the adaptive management process is to continue efforts to restore a wild, naturally-spawning population, rather than relying on maximizing egg 'take' for hatchery reproduction and maintenance of the recreational fishery. Substantial take of eggs to meet hatchery goals and targets can likely take place even if 10-20% of the adult fish are diverted for natural spawning and for experimental migration studies. Progress towards restoring a naturally spawning population on Pine Creek was initiated in 2012 when the fish trap was modified to include a weir panel fishway to improve passage (CDFW 2015). The modification enables managers to actively promote fish passage, particularly during wet spring periods when sufficient flow exists in lower Pine Creek (CDFW 2015). Recently, a conservation strategy agreement was also produced by CDFW, USFS, and USFWS (CDFW 2015). Full implementation of the goals and objectives of the *Conservation Strategy for the Eagle Lake Rainbow Trout* (CDFW et al. 2015) will vastly improve the status of the Eagle Lake rainbow trout. The agreement aims to "restore natural spawning and rearing of ELRT, preserve the uniqueness of the subspecies, and restore stream habitat" (CDFW 2015). Similar strategies have been used on California golden trout with some success. Specifically, the ELRT conservation agreement focuses on six specific conservation strategies as noted in CDFW (2015):

1. Improve passage into and through Pine Creek for migration and spawning of ELRT. This specifically includes providing improved passage through the trap/weir structure at the mouth of Pine Creek as well as effective coordination with hatchery operations.
2. Remove the brook trout population in the headwater reaches of Pine Creek and the subsequent establishment and management of a stream population of ELRT.
3. Implement an artificial spawning program and monitor genetic integrity to ensure retention of adequate genetic diversity to maintain lake and creek populations.
4. Implement effective habitat restoration projects and management strategies to improve watershed function and riparian/aquatic habitat conditions. Utilize adaptive management and monitor land use activities in coordination with ELRT conservation objectives.
5. Develop and support research projects to inform adaptive management and success criteria of conservation actions.
6. Expand outreach and education program for ELRT and the conservation of its habitats.

CDFW, in collaboration with the CRMP, released a conservation strategy for ELRT in 2015 that focuses on restoration actions in the Pine Creek watershed, including a subcomponent addressing strategies to eradicate brook trout and options for enhancing spawning success and improving natural recruitment in Pine Creek. These recent developments indicate that natural spawning and recruitment of wild stocks into the population have been identified as priorities for the recovery and management of ELRT. As studies are developed and actions identified, three basic questions should be considered:

1: Can ELRT successfully migrate upstream in most years and successfully spawn?
2: Does re-establishment of a self-sustaining population of ELRT require complete eradication of brook trout from Pine Creek?
3: Can progeny from natural spawning survive to contribute to the fishery?

Given that ELRT have undergone more than 60 years of artificial selection for reproduction and survival under hatchery conditions for a significant part of their life cycle, it is imperative to reverse that process as soon as possible. This underlying issue has long been recognized and was one of the justifications for the formation of the CRMP group in 1987, followed by many projects on Pine Creek to improve flow and remove passage barriers (Pustejovsky 2007). In order to implement adaptive management and begin the process of restoring natural spawning of ELRT, it is likely that 1) continuing a program of adult fish passage above the Pine Creek weir via the fishway is necessary, 2) aiding passage of outmigrating juveniles back to the lake is required, and 3) significant reductions or eradication of brook trout in upper Pine Creek must occur. During below average water years, it may also be necessary to trap and truck juveniles downstream past low flow portions of the creek until the stream population is self-sustaining. Recent research has demonstrated that trapping and trucking may be a viable option for helping to recreate a naturally reproducing ELRT population (Carmona-Catot et al. 2010, 2011). The costs of this type of alternative management would presumably be comparable to costs of rearing hatchery fish but with fewer genetic consequences (e.g., Waples 1999, Araki et al. 2007, Kostow 2008). Focusing on volitional passage past the Pine Creek fish trap and a natural re-colonization process, however, is preferred when possible.

Prior to modifying the fish trap to promote adult passage, evidence suggests that ELRT, at least during wet years, could migrate to the upper reaches of Pine Creek and spawn successfully. In the 1980s, a few juvenile rainbow trout were found below Stephens Meadow, suggesting adults made it over the weir, migrated upstream and successfully spawned (Moyle, unpubl. data). In 1999-2005, biologists from CDFW, USFS and UC Davis placed radio transmitters in a small number of adult fish, which were then released above the weir (L. Thompson, UC Davis, pers. comm. 2008). In 1999, one of these fish apparently made it to the Pine Creek headwaters, as its transmitter was recovered in Bogard Springs Creek, a tributary to Pine Creek above Highway 44 (T. Pustejovsky, UC Davis, pers. comm. 2008). From 2002-2006, CDFW biologists released about 500 unspawned trout into Pine creek above Highway 44.

In September 2006, a crew from UC Davis, CDFW, and the USFS sampled Pine Creek to document the presences of ELRT (Carmona-Catot et al. 2011). They found evidence that ELRT had spawned successfully in the creek in the past two years because small numbers of juvenile rainbow trout were found at several locations in Pine Creek. About 100 m of Bogard Spring Creek were electrofished and 10 juvenile rainbow trout (76-90 mm FL) were captured, along with about 170 brook trout of varying sizes. Presumably, the rainbow trout were YOY or yearlings. The rainbow trout tended to be in faster water than brook trout, in reaches with deep overhanging cover. The UC Davis crew also found 3-4 small rainbows in Pine Creek, below the Bogard Spring Creek confluence, as well as a couple of rainbow trout in the 145 mm range in a creek filled with brook trout of all sizes, speckled dace, Lahontan redside, and Tahoe sucker. Curiously, several large trout from the lake that had been planted in the spring were still surviving in the pool below the culvert under Highway 44. Likewise, three spawners were found alive in a culvert about 5 km below the highway, in a largely dry section (no surface flow) along with a rainbow trout that was 142 mm. In 2007, at least 10 large ELRT (40-50 cm FL) were found downstream from the gauging station weir on Pine Creek (G. Carmona-Catot, UC Davis, pers. comm. 2008). Successful spawning and migration was observed in 2010 and 2011, with juveniles reaching the trap and one tagged adult migrating from the weir to upper Pine Creek (T. Pustejovsky, UC Davis, pers. comm. 2008).

During 2012, the fish trap on Pine Creek was modified to include a fishway, and in the spring of 2013, 40 adult ELRT were captured at the trap and 20 individuals were PIT tagged. Tagged fish were then released downstream of the trap. Within six hours, all tagged individuals successfully navigated the trap and migrated at least two miles upstream (CDFW 2015). During 2016, 50 adult ELRT were tagged over two days and 32 of those individuals successfully navigated at least two miles upstream (14 individuals died shortly after initial release, presumably because they resided in the trap for too long prior to tagging) (P. Divine, CDFW, pers. comm. 2016). From 2013to 2015, insufficient flows in lower Pine Creek caused the fishway to remain closed. Successful migration of adult ELRT both prior to and after fish trap modifications suggests that adults are capable of successfully re-colonizing Pine Creek. The presence of brook trout in the upper watershed, however, may complicate natural ELRT re-colonization efforts. Strong reductions or eradication of brook trout from upper Pine Creek are required. From 2007-2012, Bogard Spring Creek was electrofished to remove brook trout to determine if spawning success of transplanted adult rainbow trout could be improved and to assess whether three-pass electrofishing removal can successfully depress brook trout populations. In 2007, 4,887 brook trout were removed from the 2.5 km long creek (ca. 2,000 fish /km), which is remarkable considering the creek is less than 1 meter wide and mostly less than 40 cm deep. During 2007, 170 juvenile ELRT were captured and returned to the creek; most fish were less than 150mm FL, which indicates that they were not hatchery fish planted by CDFW at larger sizes (Carmona-Catot et al. 2010). Similar results were obtained in following years, along with evidence of a greatly diminished brook trout population. This evidence strongly indicates that a wild spawning population of ELRT can be reestablished, especially if brook trout populations are largely eliminated (Carmona-Catot et al. 2011).

As a result of the work of the CRMP and CDFW, USFS, and USFWS partners, sections of Pine Creek have been fenced to exclude livestock, off-stream watering stations have been provided, an impassible culvert under Highway 44 has been replaced with a passable one, a structure to divert water from Pine Creek near the Bogard Campground has been removed (and the meadow fenced), and most importantly, the fish trap was modified to include a fishway enabling passage when it is open. However, the meadows along lower Pine Creek and Bogard Spring Creek are still grazed by cattle, potentially affecting instream habitat and reducing the capacity for meadows to store and slowly release water into streams. However, more coordinated work is needed as part of an adaptive management strategy for ELRT, which should:

- Develop a genetics management plan that can be incorporated into the broader conservation strategy.
- Continue efforts to ensure that restoration of a wild, naturally spawning ELRT population remains a priority. This includes promoting passage of adult ELRT past the fish trap, particularly during wet spring periods, and possibly assisting juveniles during outmigration.
- Develop an eradication strategy for brook trout in Pine Creek using either piscicides or other means (e.g., installation of artificial barriers and manual removal via electrofishing). Currently, CDFW is in the planning phases of a brook trout eradication strategy with the intention of releasing a draft an EIR/EIS in January 2018 (CDFW 2015). The EIR/EIS is anticipated to examine several potential brook trout removal options including the use of rotenone. Data collection on endemic or rare invertebrate taxa within the watershed is currently proceeding and anticipated to be completed in 2016

(CDFW 2015; A. Jensen, CDFW, pers. comm., 2016). Adaptive management and experimentation should be at the core of any proposed eradication efforts.

- Implement the 2015 conservation strategy to allow adult passage above the modified Pine Creek trap to maximize potential for natural migration and spawning. Formalize a plan that specifies how many adults will be allowed to pass under different water-year type conditions and how those individuals might contribute to a natural re-colonization process. Continue and expand upon existing instream movement monitoring studies (e.g., PIT tagging, radio telemetry) and incorporate assessments of passage improvement using these technologies, where applicable.
- Depending on water year type, develop plans to potentially establish trapping and trucking operations for both adults (if natural migration of adults released above the weir does not occur) and out-migrating juveniles until there are signs the population is self-sustaining.
- Continue habitat improvements in the Pine Creek watershed with the goal of improving the quantity and duration of flow, following the recommendations in Pustejovsky (2007) and CDFW (2015). Continue improvements in grazing practices and other activities that may affect stream habitat conditions. Remove artificial barriers that may inhibit adult passage including the USGS weir, the fish ladder at the 31N25 forest road crossing and the ford at the 33N33 road crossing.
- Develop a comprehensive monitoring plan to assess habitat conditions, brook trout abundance, adult ELRT instream movement, spawning success, and juvenile abundance and outmigration success.
- Determine the feasibility of using Papoose Creek for establishment of a small spawning population.
- Conduct a thorough study of the survival and growth of trout planted in Eagle Lake to determine its actual carrying capacity for ELRT. Planting of trout in the lake (150,000+ per year) is based on maintaining catches of at least 0.4 fish per hour (Dean and Chappell 2005), rather than on biological constraints. It is possible that planting fewer trout may improve trophy angling and this option should be explored further.

GOOSE LAKE REDBAND TROUT
Oncorhynchus mykiss newberrii

Moderate Concern. Status Score = 3.1 out of 5.0. Goose Lake Redband trout do not face immediate extinction risk. However, California populations are not secure because they are largely isolated from one other, most are small, and, during drought periods, the lake population disappears and stream populations contract.

Description: Goose Lake Redband trout are similar in appearance to other rainbow/redband trout. Their bodies are a yellowish to orange color with a brick-red lateral stripe, with generous spotting along the body. The dorsal, anal, and pelvic fins are white-tipped. Stream-dwelling adults retain parr marks, while lake-dwelling adults become silvery-grey in color. The Goose Lake Redband trout has two ecological types: a lake-dwelling form that attains lengths of 45-50 cm TL and a stream-dwelling form that rarely grows larger than 25 cm TL. Behnke (1992) examined six specimens collected by J. O. Snyder in 1904 from Cottonwood Creek, in the Oregon portion of the basin. These fish had 21-24 (mean, 23) gill rakers, 61-64 (mean, 63) vertebrae, and averaged 30 scale rows above the lateral line and 139 scales in the lateral series. See Behnke (2002) for color plates of both lake and stream forms.

Taxonomic Relationships: Redband trout are inland forms of Rainbow trout (Behnke 1992, 2002) and the Goose Lake redband belongs in the group that Behnke (2002) calls "redband trout of the northern Great Basin, *O. m. newberrii*." The Goose Lake Redband trout is most similar to redband trout of two adjacent basins: the Warner Basin, California, Oregon and Nevada, and the Chewaucan Basin, Oregon (Behnke 2002). This conclusion is based on the lower vertebral counts and higher gill-raker counts of redband trout in these basins and distinct genetic markers.

The USFWS lumped Goose Lake Redband trout with five other Great Basin redband trout as one Distinct Population Segment when considering a petition for listing them as threatened under the Federal Endangered Species Act (*Federal Register* 65(54), March 20, 2000, 14932-14936). Although the Goose Lake watershed may have had connections to other Great Basin watersheds during wetter climatic periods, it is clearly isolated from other basins today and, presumably, has been for thousands of years. While *O. m. newberrii* is a reasonable taxonomic designation, the Goose Lake Redband trout is clearly also a distinct evolutionary unit/population segment, confined to the Goose Lake basin and nearby headwater streams in the upper Pit River.

Warner Lakes Redband trout are similar genetically to Goose Lake Redbands, and inhabit only a very small portion of water in an adjacent basin to Goose Lake in California (Dismal and Twelvemile creeks). Recent studies using DNA (amplified fragment length polymorphism AFLP techniques) indicate a close relationship between Goose Lake redbands with Warner Lakes Redband trout (M. Stephens 2007, Currens et al. 2009). Currens et al (2009), using both mitochondrial and nuclear DNA, posited that redband trout from the upper Pit watershed, Goose Lake, and the Warner Lakes basin form a distinctive lineage that is perhaps deserving of its own ESU under the criteria developed by the U.S. Fish and Wildlife Service. This has since been confirmed by more recent work from Simmons et al. (2011), Muhlfeld et al. (2015), and DeHaan et al. (2015). In addition the Goose Lake Redbands and Warner Lakes fish show very low levels of introgression with coastal rainbow trout, which is fortunate given past stocking in the Oregon portion of the watershed (DeHaan et al. 2015). While the California Department of Fish and

Wildlife consider the Warner Lakes Redband population a separate qualifying trout from the Goose Lake Redbands for the California Heritage Trout Challenge (CDFW 2016), we group them with the related Goose Lake Redbands in this account.

Life History: Goose Lake Redband trout have two life history strategies: a lake strategy and a headwater strategy. Lake strategy fish live in Goose Lake, where they grow to large size and spawn in tributary streams. Headwater strategy fish remain small and may spend their entire life cycle in streams. It is almost certain that the two forms represent one population because the aperiodic desiccation of Goose Lake presumably has eliminated the lake form repeatedly in the past. This was demonstrated in 1992 when the lake dried up entirely during a prolonged drought. In the next two years, the lake refilled and, about three years later, small runs of large trout again appeared in the streams. It is assumed that the lake dwelling form was reestablished from tributary stream-resident populations. In the small, cold streams of the Warner Mountains to the east of Goose Lake, scattered populations of resident trout persist, completing their entire life cycle in these streams. They look quite different from lake fish because of small size and more vibrant color patterns, reflecting responses to a stream environment. Many of these populations are above potential barriers to upstream movement of fish from the lake (Muhlfeld et al. 2015, M. Dege, CDFW, pers. comm. 2016). Presumably, small numbers of headwater redbands always move downstream, a natural mechanism for dispersing to new habitats or for recolonizing streams wiped out by drought or other natural disasters. Some of these fish reach the lake and, a few years later, they mature and spawn, renewing the cycle. It is also possible that progeny of lake-strategy fish can persist in some lower-elevation tributaries (e.g., Cold Creek).

In California, the lake-dwelling form ascends small tributaries to spawn. Landowners in the area have seen large adults ascending Cottonwood (Oregon) and Pine creeks in the spring, presumably migrating for this purpose, while CDFW have documented large spawning adults in Lassen, Cold, and Willow Creeks (CDFW unpubl. obs.). If sufficient flows are available, they also spawn in Buck Creek, a small tributary of Willow Creek. Upstream of its confluence with Cold Creek, a steep, rocky gorge apparently prevents spawners from ascending further up Lassen Creek. In Oregon, they formerly spawned in Thomas Creek and its tributaries and, possibly, in Cottonwood and Drews creeks. Spawning migrations occur following snow melt and rain in the spring, usually during late March or in April. Spawning fish are rather pale looking, perhaps as a result of time spent in Goose Lake's highly turbid waters. Adults return to the lake following spawning. Young trout apparently spend one or more years in streams before dispersing downstream (if they leave at all) into Goose Lake. In the lake, the trout likely feed on Goose Lake Tui chub, Tadpole shrimp, and other super-abundant food. Growth appears rapid; scales from 6 spawning fish (27-48 cm TL) taken in 1967 indicated that they were all 3 years old (CDFW unpubl. data).

The life history of the stream-dwelling form has not been studied but it is thought to be similar to other redband and rainbow trout that live in small, high-elevation streams. Surveys by CDFW (CDFG unpublished data; Hendricks 1995) indicate that headwater streams have 4-5 length classes of trout, with a maximum size around 240 mm TL, though about 80% of the population measured less than 150 mm (Weaver and Mehalick 2010). It appears that fish in their third summer are 9-12 cm TL. Lake fish were observed spawning May 14-15, 2007 (CDFW unpubl. data), though spawning time is highly dependent on variable water years and amount of runoff.

Habitat Requirements: Goose Lake is a large, alkaline lake that straddles the California border; it is shallow (mostly < 3 m when full), extremely turbid, and highly variable in area (about 500 km^2). Because of its high elevation (1430 m), the lake generally remains cool (<22°C) although summer temperatures in the lake may reach 24°C or higher during the day. During calm days, water temperatures stratify with warm water within the first 25-50 cm of the surface; on most days the wind causes temperatures to be uniformly cool (R. White and P. Moyle, unpublished data, 1989). Goose Lake Redbands nevertheless survive warm temperatures, high alkalinities, and high turbidity that exist in Goose Lake during summer months. Presumably, a major factor contributing to their survival is the extraordinarily high abundance of fish, tadpole shrimp (*Lepidurus lemmoni*) and other food in the lake (P. Moyle and R. White, unpubl. obs.).

Spawning takes places in March-May, whenever flows in Willow and Lassen creeks are high enough to attract trout for an upstream migration (M. Yamagiwa, USFS, and S. Reid, pers. comm. 2007). Most spawning areas are located in reaches and tributaries with permanent flows, such as Cold Creek, a tributary to Lassen Creek about 15 km upstream from the lake. Spawning sites are reaches with clean gravels and riparian cover that maintain cool water temperatures. Goose Lake Redbands have been observed to spawn in the lower reaches of Willow and Lassen creeks when access to upstream areas is blocked (P. Chappell, USFS, pers. comm. 1995), but most spawning areas are upstream of the Highway 395 crossing. However, spawning migrations and behavior of Goose Lake Redband Trout has been poorly recorded in California.

Tate et al. (2005) evaluated temperatures in the two largest California tributaries to Goose Lake, Lassen and Willow creeks. Lassen Creek, the larger of the two (1-2 cfs flows in late summer), became progressively warmer from headwaters to mouth, so that headwater reaches were typically <16°C in summer, while lower reaches typically averaged 18-21°C, all reasonable temperatures for trout. However, in the summer of 2007, temperatures in some reaches supporting trout regularly reached 24-26°C (S. Purdy, unpublished data). Likewise, Tate et al. (2005) found temperatures in Willow Creek of 24°C on occasion, although intermediate reaches in a shaded canyon were considerably cooler.

The habitat requirements of the stream-dwelling form are similar to other populations of redband trout that occupy small, cool, high-elevation streams. Forested, headwater streams with complex substrates provide habitat for most Goose Lake Redbands (Scheerer et al. 2010). Streams in the Warner Mountains are generally dominated by riffles with undercut banks. Pools in meadow areas provide habitat for larger fish. Dense overhanging vegetation, especially willows, provide essential cover.

The environmental tolerances of Goose Lake Redband Trout have not been measured, but it can be inferred that they can survive temperatures of 24°C for short periods on a regular basis, highly turbid, alkaline water (pH 8-9), and dissolved oxygen levels at <50% saturation, although growth may be inhibited under more extreme conditions.

Distribution: Goose Lake Redband Trout are endemic to Goose Lake and its major tributaries and a few tributaries to the upper Pit River. In California, Lassen and Willow creeks are their principal streams although they are also present in smaller streams (Pine, Cottonwood, Davis, Turner, and Corral creeks). In Oregon, they inhabit the extensive Thomas-Bauers Creek system as well as 12 smaller streams (Fall, Dry, Upper Drews, Lower Drews, Antelope, Muddy, Cottonwood, Deadman, Crane, Cogswell, Tandy, and Kelley creeks) (Oregon Department of Fish and Wildlife 2005). Berg (1987) reported that Joseph and Parker and creeks, tributaries of the North Fork Pit River in California, and East Creek, tributary to Mill Creek and the South Fork Pit River, contained trout genetically similar to Goose Lake Redbands. Similar results for upper Pit River Redbands were found by M. Stephens (2007). Simmons (2011) identified genetically similar fish in North Fork Fitzhugh Creek, tributary to South Fork Pit River and in Parker Creek, tributary to North Fork Pit River, south of Goose Lake. In addition, two populations, from Crump Lakes (Deep and Twentymile creeks) and Honey Creek (Hart Lake) in the eastern Warner Mountains above Surprise Valley, OR, seem to be Goose Lake Redbands, perhaps as the result of historical introductions (Stephens 2007, DeHaan et al. 2015). Goose Lake Redbands have been reported from Pine Creek (1993) and Turner Creek (1986) during electrofishing surveys, although they could have been misidentified (CDFW 2015 electrofishing data).

Trends in Abundance: According to local history, in the 19th century these trout were once abundant enough in the lake that they were harvested commercially and sold to logging camps. Conversations with local residents (P.B. Moyle 1989) indicated that both sport and commercial fisheries existed for Goose Lake Redband trout and that large runs occurred in local creeks, especially Thomas Creek in Oregon. The Goose Lake Redband trout population historically has undergone major fluctuations, being depleted during series of dry years and recovering in wet periods. The lacustrine population was severely depleted during the 1976-1977 drought, recovered during the wet early 1980s, and dropped precipitously during the 1986-1992 drought. Most recently, the lake was dry in 2009, nearly dry in 2010, extremely low and most likely dry in 2013, and completely dry in both 2014 and 2015 (M. Dege, CDFW, pers. comm. 2016). As a result, there is likely no lacustrine population at the moment, but the potential for that life history expression still exists in trout in tributaries to the lake.

In California, Lassen Creek and its tributary, Cold Creek, have been the principal spawning streams. Numbers of spawning fish have fluctuated from ten or so individuals to several hundred, but the creek appears to have the potential to support perhaps 1,000 spawning fish under optimal flow conditions (E. Gerstung, CDFW, pers. comm. 1995). The only large run documented in recent years in Lassen Creek (1988) was comprised of several hundred spawners (J. Williams, unpubl. data), which suggests that there were fewer than 1,000 adults from California streams in Goose Lake, assuming many of the lake fish were immature one and two year old fish. In 1989, in the middle of a drought, only about a dozen fish appeared in the creek and there was no evidence of successful spawning.

Goose Lake dried up in 1992 but, by March, 1997, a run was reported in Lassen Creek and spawning was reported in April in Cold Creek (M. Yamagiwa, USFS, pers. comm. 2007). In May, 1999, (S. Reid, CDFW, pers. comm. 2007) observed "…big fish (40-70 cm) stacked four deep (literally) in the pools (estimated 75 at Hwy. 395)." This suggests that runs of several hundred fish had redeveloped in these tributaries and others in a relatively short period of time.

The stream form of Goose Lake Redband trout apparently exists in about 20 small headwater streams. ODFW (2005) estimated that about 102,000 trout (+/-32%) age 1+ and older ($0.14/m^2$) live in 13 Oregon streams under typical conditions; this number is presumably low compared to numbers that existed before streams were degraded by grazing and other activities. Surveys of California streams made in 1993 and 1999, showed 600-1,600 trout per km in Lassen Creek, which suggests that densities/numbers in California and Oregon streams are roughly comparable (CDFW unpubl. data). More recent CDFW multiple-pass electrofishing surveys (Weaver and Mehalick 2010) estimated 114-747 trout per km in Lassen Creek and 313-451 trout per km in Cold Creek, considerably lower than previous estimates from surveys in 1986 and 1999, but with the caveat that section lengths were estimated in 1999 (J. Weaver, CDFW, pers. comm. 2013), so abundance estimates may or may not be accurate for that year.

ODFW (2005) indicated that most Oregon redband trout streams are impaired to some degree by cumulative effects from irrigation diversion dams, dewatering of streams, and generally poor habitat (from grazing, mining, and roads). Most of the streams also suffer from loss of connectivity to each other and to Goose Lake, and therefore it is likely that these habitats no longer support their potential productive capacity of native fishes (Scheerer et al. 2010). Streams in California face similar challenges although the largest stream, Lassen Creek, seems to be in better condition than most, largely due to extensive habitat restoration efforts such as installing juniper revetments, weirs and boulders, culvert baffles, fish screens, and removal of debris (CDFW 2010). Since 1995, conditions for Goose Lake Redband trout in California have steadily improved because large sections of Lassen Creek and other streams have increased protection from grazing due to changes in USFS allotments and otherwise been restored. These conservation measures have likely improved habitat conditions, which can benefit runs of lake fish to re-establish themselves when hydrologic conditions are favorable. Presumably, headwater populations have increased as well, thanks to better management. Recent habitat improvements in Oregon actually led to an expansion of the distribution of the species from 1995 to 2007, according to ODFW surveys (Scheerer et al. 2010).

Factors Affecting Status: Goose Lake Redband trout populations face many stressors, but habitat degradation and diversions have historically been and remain the greatest threats (Table 1). ODFW (2005) indicated that these two factors, combined, put Goose Lake Redband trout "at risk" in 80% of Oregon streams. Overexploitation and introduced species are, at present, minor problems. However, all threats are exacerbated during periods of severe drought such as the current one. Goose Lake dried up in the 1420s, 1630s, 1926 (with low lake levels from 1925 to 1939), 1992, and 2009, and 2014-2015-2016. As of Fall 2016, Goose Lake is filling as a result of an above normal water year in northern California (P. Divine, CDFW, pers. comm. 2017). The key to long-term survival of Goose Lake Redband trout (and other Goose Lake fishes) is maintenance of populations in tributaries that may have severely reduced habitat during drier periods.

Agriculture. Populations of the lake-dwelling form were reduced because access to spawning areas was blocked by dams, diversions, culverts, and channelization in the lower reaches of many streams but, since 1995, most of these impacts have been mitigated or eliminated. Much of the critical stream habitat for Goose Lake Redband Trout is on private land and, at times, large volumes of water are diverted to irrigate fields. On some streams, small diversion dams are barriers to fish movement (ODFW 2005). Diversions may have disproportionate impacts in dry years because they have the potential to dry longer stream reaches that are refuges for trout and other fishes when the lake is dry. Many fish screens have been installed on Goose Lake tributaries and are helping to reduce some impacts of agricultural diversions (CDFW 2010).

Grazing. Headwater streams containing redband trout have been heavily grazed, resulting in reduced riparian cover and, in places, down-cutting to bedrock. The impact of grazing has been reduced in recent years through a combination of fencing, rotational grazing, installation of erosion control structures, and planting of willows.

Transportation. All streams in the watershed have been degraded by roads to some degree. Highway 395 crosses all tributaries to the east side of the lake and culverts under the highway were once a partial barrier to migration, an issue which has largely been fixed. Roads also impact headwater streams, especially where culverts may be barriers to fish movement or where the road-cuts are a source of silt. Some streams face multiple threats from poor water quality as the result of road building, channelization, and waste materials from uranium mines.

Logging. Timber harvest is a prominent use of the watershed's forests and has contributed to habitat degradation in streams through siltation, road-crossings, and other factors. Logging impacts were more severe historically; many regulations exist today to protect stream habitats from the effects of timber harvest operations.

Harvest. When lake-dwelling fish are moving upstream to spawn, they are vulnerable to poaching, especially when confined below culverts or other partial barriers. This may have been a factor in the decline of the Lassen and Willow Creek populations. At present, only catch-and-release angling for redband trout is permitted in Goose Lake's California tributaries and legal fishing pressure remains light (CDFW 2010).

Alien species. Brook, Brown, and Rainbow trout have been introduced into streams of the Goose Lake drainage and Brown trout are known to persist in Davis and Pine creeks in California (Hendricks 1995, S. Purdy, unpubl. obs. 2006, P. Divine, CDFW, pers. comm. 2012). Brook trout are still present in at least one Oregon stream (ODFW 2005, Scheerer et al. 2010). California has not stocked any Rainbow trout in the drainage since 1980, when electrofishing studies indicated that the native redbands were distinct; planting of hatchery Rainbow trout apparently was discontinued in Oregon tributaries in 1961, although Cottonwood Meadows Reservoir, on Cottonwood Creek, is still planted with hatchery Rainbow trout (ODFW 2005). Behnke (1992) thought that some Goose Lake Redband trout populations in California showed evidence of past hybridization with Rainbow trout, based on meristic measurements, but there is no biochemical evidence of this. Recent information suggests there is little risk of hybridization with rainbow or Redband trout in the Goose Lake Basin (ODFW 2008).

The potential for future unauthorized, illegal introductions to impact native trout and other sensitive Goose Lake fishes remains although is unlikely. Possible effects to native fishes could occur through disease, hybridization, predation, or competition; however, but not hybridizing. Past introductions of warm-water fishes were largely unsuccessful because of the lake's extreme environment, though Brown trout do persist in some tributaries to Goose Lake and compete with and may consume Goose Lake Redbands (P. Divine, CDFW, pers. comm. 2013).

Beavers were historically distributed in the Goose Lake basin and likely provided ecological riparian benefits (CDFW 2013). Beaver dams in Lassen Creek's middle reaches have created intermittent dams and may have blocked lake fish runs from reaching preferred spawning habitat (J. Weaver, unpubl. obs. 2012). The California Department of Fish and Wildlife has, in the past, periodically used explosives to remove beaver dam complexes in Lassen and Willow creeks in order to improve upstream passage for Goose Lake Redbands, although this practice is no longer utilized (P. Divine, CDFW, pers. comm. 2012). Beaver dams may need to be evaluated in the future to determine if fish passage is being impeded and the overall ecological benefits they can provide. For example, their impediments to passage of large adults may be offset by their provision of back flooded habitats and water storage, especially as seen during recent drought years (M. Dege, CDFW, pers. comm. 2016).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Goose Lake Redband trout in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate. See methods for explanation.

Factor	Rating	Explanation
Major dams	n/a	
Agriculture	High	Water diversion and return flows from irrigation lower base flow and increase water temperatures; dams may block migration of lake fish, and these impacts are compounded in drought.
Grazing	Medium	Pervasive in the area, especially in meadows with redband streams; reduced impacts in recent decades with improved management but still an important factor.
Rural /residential development	Low	Rural development may impact a few streams in California through diversions, but is mostly an issue on the Oregon side of Goose Lake.
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Old uranium mines in watershed; unknown impacts.
Transportation	Low	Roads are a source of erosion and sediment input into streams and culverts have blocked access in the past.
Logging	Low	Logging and associated roads likely contributed to stream degradation, increased water temperatures and reductions in water quality, though greater impacts were in the past.
Fire	Low	Fire suppression, coupled with increasing aridity predicted with climate change, may contribute to increased fires.
Estuary alteration	n/a	
Recreation	Low	Off road vehicles a potential threat but not demonstrated.
Harvest	Medium	Poaching is potentially a problem during spawning; legal fishing pressure is light and limited to catch-and-release.
Hatcheries	n/a	
Alien species	Low	Trout introductions not regarded as a major threat, though competition and predation by brown trout may occur where both species overlap (P. Divine, CDFW, pers. comm. 2013).

Effects of Climate Change: Goose Lake is located in an arid, high desert region so any reduction in precipitation or increased frequency of droughts will further stress streams and the lake. Climate change models predict both in the future (Moyle et al. 2012). During low flow periods, streams in the Goose Lake basin already reach temperatures (24-26°C) that are nearly lethal to redband trout. Any increase in air temperature, combined with reductions in stream flow through diversions, could reduce or even eliminate most California populations, especially in lower reaches of streams where diversions are more common. Upstream reaches of streams, which are largely free from diversions and provide shaded habitat, cool springs, canyon reaches, and intact meadows, are likely to provide refuge for Goose Lake Redbands in the future (M. Dege, CDFW, pers. comm. 2016). An increase in fire frequency or intensity could reduce riparian shading, add sediment, and otherwise impair streams. Increased frequency of Goose Lake's known desiccation and lake temperatures could reduce the lake part of the population. Moyle et al. (2013) rated Goose Lake Redband trout as "critically vulnerable" to climate change, with extinction likely in California in the next 100 years if present climate change trends continue. Increasing likelihood of long-term drought will exacerbate these likely impacts.

Status Score = 3.1 out of 5.0. Moderate Concern. Goose Lake Redband trout face no immediate extinction risk, but their populations are not secure because: (a) of the 19 extant populations, only 6 are in California, (b) most stream populations are small, and (c) drought is predicted to increase over the century, which causes lake populations to disappear and stream populations to shrink. Warmer temperatures reduce the quantity and quality of stream refuges.

The Goose Lake Redband trout has been given various designations by state and federal agencies: (a) USFWS, Category 2 Candidate Species (now, Species of Concern); (b) USFS, Region 5, Management Indicator Species; (c) USFS, Region 6, Sensitive Species, (d) ODFW, Vulnerable or At Risk species and CDFW, Species of Special Concern (Moyle et al. 2015). The American Fisheries Society lists it as "Vulnerable," while NatureServe lists it as "Imperiled" (T2) (Jelks et al. 2008). In 1997, the USFWS was petitioned to list Great Basin Redband trout, which includes Goose Lake Redband trout, as threatened or endangered. In 2000, the petition was denied (Congressional Record, March 20, 2000:65 (54):14932-14936) because following a 1994 drought and reduction in Goose Lake Redband trout abundance, the population rebounded and the fish were estimated to inhabit 59% of their historical range. Since then, it is likely the population has shrunk due to drought, especially from 2010-2015.

USFWS analysis also cites the many successful restoration projects in the Goose Lake Basin as further reason for finding that listing was not justified. Goose Lake Redbands in California depend largely on just two streams, Lassen and Willow creeks, for survival. These populations could face extirpation from California even if there are viable populations in Oregon. Simmons (2011) indicated that there was substantial gene flow between the Lassen Creek population and others in the Goose Lake Basin, in support of its high genetic diversity. This may be supported by Passive Integrated Transponder (PIT) tagging studies by Oregon Department of Fish and Wildlife documenting lake fish moving to stream mouths in the spring (P. Divine, CDFW, pers. comm. 2017). More current information on California populations and better resolution of levels of movement (or lack thereof) of lake dwelling fish between tributaries, both in Oregon and California, would change their status. A fish rescue in 2014 and 2015 has given evidence that populations in Lassen and its tributary, Cold Creek, have declined since the last surveys in the 1990s and are susceptible to drought impacts (CDFW 2010).

Table 2. Metrics for determining the status of Goose Lake Redband trout in California, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	4	Present in six streams in California and 13 in Oregon.
Estimated adult abundance	3	Populations greatly reduced in drought years.
Intervention dependence	4	Long-term decline reversed by restoration actions; over 220 Redbands were rescued from Cold Creek during drought in 2014 and 2015; may require future rescue if drought conditions persist.
Tolerance	4	Indirect evidence suggests they are more tolerant than most salmonids of adverse water quality.
Genetic risk	3	Genetic risks are currently low; potential impacts from isolation of headwater populations need investigation.
Climate change	2	Distribution in isolated, small streams increases probability of extirpation due to prolonged drought.
Anthropogenic threats	2	1 High, 2 Medium threats.
Average	3.1	22/7.
Certainty (1-4)	2	Mostly 'grey' reports and expert opinion.

Management Recommendations: There has been interest in conserving populations of endemic fishes in the Goose Lake Basin. During the 1987-1992, 1994 drought, a proposal was developed to list the Goose Lake fish fauna as threatened under the federal ESA. In response, the Goose Lake Fishes Working Group was formed in 1991, made up of representatives from both California and Oregon, and comprised of private landowners, state and federal agencies, non-governmental organizations, and universities. The organization signed a Memorandum of Understanding in July, 1994, to protect and, where needed, reestablish native fishes in the Goose Lake basin. In 1995, the Goose Lake Fishes Conservation Strategy was completed. According to USFWS (Congressional Record, March 20, 2000:65 (54): 14936), the goal of the strategy is to conserve all native fishes in Goose Lake by: reducing threats, stabilizing populations, and maintaining an intact ecosystem. Since 1996, many improvements have occurred to remove barriers to migration, such as culvert replacement, passage improvement, diversion reparations, and various habitat surveys, as well as sediment reduction projects such as road improvement.

In the lower reaches of most streams, restoration actions included making road under-crossings passable to trout. A fish ladder was installed over a major diversion dam on Thomas Creek in 1992 by the Oregon Department of Fish and Wildlife. In Willow and Lassen creeks, the California Department of Fish and Wildlife has removed natural and artificial migration barriers. Headcut control, bank stabilization, stream fencing, planting of riparian vegetation, modified grazing practices and other protective measures have also been undertaken on a number of streams in recent years. These measures have greatly improved habitat and water quality in Goose Lake tributaries, including the lower reaches that flow through agricultural land. Monitoring of water quality, insects, and fish demonstrate the improvements (Tate et al. 2005); however, continued effort is needed to maintain (and ideally increase) the populations of trout and other fishes, especially during periods of severe drought (CDFW 2010). It is likely that populations have declined considerably during the drought years 2012-2016. Management recommendations (not in order of priority) include:

1. Identification and modification of barriers to fish movement, especially diversion dams. Restoring stream-lake connectivity would enable full expression of various life history potential in migratory fish like Goose Lake Redbands (Scheerer et al. 2010).
2. Identification, protection, and improvement of stream reaches that are critical for spawning, rearing, and refuge during drought. Cold Creek (tributary to Lassen Creek) and Buck Creek (tributary to Willow Creek) have already been identified as important habitats. At present, a diversion structure often diverts flows from lower Buck Creek. Lower Willow Creek habitat conditions are poor (bank sloughing, minimal riparian or instream cover, heavy sedimentation), along with multiple diversion dams. Although these dams were, at some point, improved with fish ladders, some of these structures appear badly deteriorated and fish passage needs to be reevaluated (J. Weaver, unpubl. obs., 2012). Implementation of water conservation measures could help increase streamflow and ensure habitat is available during low water years.
3. Regular quantitative monitoring (every 3-5 yrs) of fish populations in both upstream and downstream reaches of Lassen and Willow creeks, and at least qualitative monitoring of fishes in other streams. While no California-Oregon collaboration currently exists for basin-wide monitoring, CDFW and ODFW have been participating in range-wide assessments and collaboration under the recently-signed Interior Redband Trout Conservation Agreement (Redband Trout Conservation Agreement 2016). Invasive Brown trout should be removed to benefit the native fish assemblage (Goose Lake lamprey, Tui chub, and sucker) through reduced predation and competition. Together with Goose Lake Redbands, these four species are all endemic to the basin and are listed as state Species of Special Concern (Moyle et al. 2015).
4. Improved management of headwater areas to protect streams from livestock grazing and other stressors through the use of exclusion fencing, off-channel water sources for livestock, and working with landowners to improve riparian habitat through management changes.
5. Prevent the illegal importation/stocking of non-native fish in the Goose Lake Basin, and eradicate existing populations where possible. Bans on introductions of alien fishes or invertebrates that could alter the forage base or negatively impact native fishes should continue.
6. Adult lake-form trout in small streams are susceptible to poaching. Regular patrols by wardens and others should be conducted to prevent poaching adults in spawning areas.
7. The Goose Lake Fishes Conservation Strategy should be fully implemented and revisited periodically to ensure it is up to date. This is currently ongoing. The continued involvement of private landowners and public agencies is crucial to this effort, as is the continued involvement of University of California Cooperative Extension, which has provided coordination and scientific studies to support conservation efforts.

KERN RIVER RAINBOW TROUT
Oncorhynchus mykiss gilbertii (Jordan)

Critical Concern. Status Score = 1.4 out of 5.0. The Kern River rainbow trout has a high probability of disappearing as a distinct entity in the next 50-100 years, if not sooner. The greatest threat continues to be hybridization with coastal rainbow trout, but competition and predation from invasive brown trout and brook trout may also be contributing to its decline.

Description: This subspecies is similar to coastal rainbow trout but its coloration is brighter, with a slight tinge of gold; it has heavy, fine spotting over most of its body (Moyle 2002). The spots are more irregular in shape than those of the round spots of the other two Kern basin golden trouts. On many larger fish, there is a broad rosy-red band along the sides. There are also minor differences in meristics from the other two golden trouts (Schreck and Behnke 1971).

Taxonomic Relationships: The taxonomic status of this subspecies is controversial because of its complex evolutionary history and exposure to introduced varieties of rainbow trout. In 1894, D. S. Jordan designated this fish as a distinctive subspecies of rainbow trout; this analysis was accepted until Schreck and Behnke (1971) described it as a population of golden trout. Their decision was based mostly on comparisons of lateral scale counts and on aerial surveys that led them to believe that there were no effective barriers on the Kern River which might have served to isolate trout in the Kern River from those in the Little Kern River [in particular, barriers to downstream movement of golden trout into the Kern River, which also applies to Golden Trout Creek]. However, in a subsequent analysis, Gold and Gall (1975) determined that golden trout populations were effectively isolated genetically and physically. Meristic (Gold and Gall 1975) and genetic (Berg 1987) characteristics of *O. m. gilbertii* were regarded as sufficiently distinctive to warrant its subspecific status (Berg 1987). Bagley and Gall (1998), using mitochondrial and nuclear DNA, found that the Kern River rainbow was distinctive, but probably originated as the result of an early (natural) invasion of coastal rainbow trout that hybridized with Little Kern golden trout, creating a new genome. This has been more or less confirmed by analysis of genetic variation by Amplified Fragment Length Polymorphism (AFLP) markers for populations of rainbow trout statewide (M. Stephens 2007). The AFLP analysis indicated that Kern River rainbow trout represent a distinct lineage that is intermediate between coastal rainbow trout and Little Kern golden trout, although there was also some evidence of recent hybridization with coastal rainbows, presumably of hatchery origin. Most recently, Erickson (2013), using single nucleotide polymorphism (SNP) and microsatellite markers found Kern River rainbow trout to be most closely related to California golden and Little Kern golden trout, relative to numerous rainbow trout hatchery strains and a coastal rainbow trout reference population.

Life History: No life history studies have been performed on this subspecies, but its life history is assumed to be similar to other rainbow trout populations in large rivers (e.g., Moyle 2002). Historically, fish found in the mainstem Kern River grew to large sizes, as much as 71 cm TL and 3.6 kg (Behnke 2002), although fish over 25 cm TL are rare today (S. Stephens et al. 1995).

Habitat Requirements: Little information is available on Kern River rainbow trout but, in general, their habitat requirements are likely similar to other rainbow trout, with some modifications to reflect the distinctive environment of the upper Kern River (Moyle 2002). Environmental tolerances are presumably similar to those of coastal rainbow trout.

Distribution: This subspecies is endemic to the Kern River and its tributaries in Tulare County. It was once widely distributed in the system; in the mainstem it probably existed downstream well below where Isabella Dam is today and upstream in the South Fork as far as the town of Onyx (S. Stephens et al. 1995). It has been largely extirpated from the Kern River at least downstream from the Johnsondale Bridge (ca. 16 km above Isabella Reservoir). Today, remnant populations exist in the Kern River above Durrwood Creek, in Rattlesnake and Osa creeks, and possibly upper Peppermint Creek (S. Stephens et al. 1995). Bagley and Gall (1998), used a variety of genetic techniques to determine that several populations, mostly located in the middle section of the Kern River drainage, were relatively unhybridized Kern River rainbow trout: Rattlesnake Creek (in Sequoia National Park), Kern River at Kern Flat, Kern River above Rattlesnake Creek, Boreal Creek, Chagoopa Creek, Kern River at Upper Funston Meadow, Kern River above Redspur Creek, and Kern River at Junction Meadow. These populations are in the middle of the historical range and lack hybridization with either California golden trout (seen in the upper sections of the Kern) or with coastal rainbow trout (seen in the lower sections). While Behnke (2002) doubted that pure Kern River rainbow trout still exist in their native range, recent genetic analyses suggest that at least some unhybridized populations exist as indicated above. Erickson et al. (2010) and Erickson (2013) found seven populations that exhibit low introgression (estimates of less than 10%) and a few populations exhibit introgression of less than 2%. Nearly all of these populations occur in headwater tributaries that have been reproductively isolated from downstream, and in some cases, heavily introgressed populations. Much of their remaining habitat is in Sequoia National Forest (29+ km) and Sequoia National Park (40+ km). In addition, there are distinctive introduced populations in the Kern-Kaweah River and Chagoopa Creek, which have maintained their genetic identity (M. Stephens 2007, Erickson 2013).

Trends in Abundance: Kern River rainbow trout were once abundant and widespread in the upper Kern Basin. As a result, they were subject to intensive removal by angling. Since the 19th century, overexploitation, combined with habitat degradation and, most importantly, hybridization with other trout, has reduced populations to a small fraction of historical numbers. In 1992, a study of Kern River rainbow trout abundance in the Kern River in Sequoia National Park indicated there were about 360-840 trout per km (600-1,400 trout per mile) of all sizes (Stephens et al. 1995). Snorkel survey data collected by the Department of Fish and Wildlife in October of 2009 indicated there were about 32-2,145 trout per km (51-3,452 trout per mile) over multiple surveyed reaches on the Kern River (DFW Wild and Heritage Trout Snorkel Survey data set 1994-2012). However, the genetic status of these fish and potential estimates of introgression are unknown. There are no abundance data on unhybridized or minimally introgressed Kern River rainbow trout populations but, if it is assumed they currently persist in 20 km of small streams, with 200-1,500 trout per km, the total numbers would be 4,000-30,000 fish. These estimates are highly questionable given natural variation in numbers, difference in survey methods, small sample sizes upon which they are based, and uncertainties regarding the actual distribution of Kern River rainbow trout and their respective levels of introgression. The estimates do, however, suggest that absolute numbers in the wild are low and vulnerable to reduction by natural and human-caused events. The majority of the least hybridized populations are isolated from other populations, as shown in recent genetic assays (Erickson 2013). Thus, the status of Kern River rainbow trout could deteriorate rapidly due to their lack of population connectivity as populations disappear or become heavily hybridized.

Factors Affecting Status: The largest threat to Kern River rainbow trout is the introduction of other trout strains (coastal rainbow, California golden, Little Kern golden trout) and the loss of Kern River rainbow trout genetic material due to hybridization (Erickson et al. 2010). Erickson (2013) performed a detailed genetic analysis of upper Kern Basin trout in the historical range of Kern River rainbow trout using single nucleotide polymorphism (SNP) and microsatellite markers to evaluate the extent of introgression, and found that introgression with coastal rainbow trout and California golden trout was prevalent throughout the basin, but much less so for Little Kern golden trout. Many of the lower basin tributary populations were heavily introgressed with coastal rainbow trout. California golden trout introgression was also apparent in lower portions of the Kern basin and some sites in the upper basin. Erickson et al. (2010) and Erickson (2013) attributed high levels of hybridization with both coastal rainbow and California golden trout to well documented hatchery stocking in the past. Of the populations studied, Erickson et al. (2010) and Erickson (2013) found seven Kern River rainbow trout populations that showed low or extremely limited hybridization, scattered among creeks and lakes in the upper Kern Basin or nearby basins (from introductions). These populations generally exhibited less than 10% introgression estimates and showed strong similarities to the Kern River rainbow trout genetic reference. The Nine Lakes North population, in addition to Kern-Kaweah River and Picket Creek populations, represents the best examples of Kern River rainbow trout due to extremely low introgression estimates and a high number of Kern River rainbow trout-specific alleles (Erickson et al. 2010, Erickson 2013). While reproductive isolation has played an important role in minimizing hybridization in a few populations within the basin, those same populations show limited genetic diversity because of a lack of population connectivity and from genetic bottlenecks associated with founder effects (Erickson 2013). In addition to the aforementioned populations, Erickson (2013) also found another 14 populations that showed a distinct Kern River rainbow trout genetic signature with varying degrees of hybridization.

The primary threats to remaining populations are identical to those facing other endemic trout of the southern Sierra, which center on interactions with non-native trout: (1) hybridization with hatchery rainbow trout, which are still planted in the upper Kern Basin, though not in Sequoia National Park, (2) hybridization with golden trout historically planted (particularly California golden trout), that may continue moving into their waters, (3) low genetic diversity affecting the ability of relatively unhybridized populations to adapt to changing conditions and (4) competition from brown, brook, and hatchery rainbow trout. Invasions by hatchery rainbow trout or by brown or brook trout into the remaining small, isolated streams are possible, especially through angler-assisted introductions. In addition, habitat loss from the region's long history of grazing, logging and roads, as well as stochastic events such as floods, drought and fire can degrade habitats, negatively affecting already isolated populations and their persistence (Moyle 2002). For a full discussion of these shared regional stressors in the Kern Basin, see the California golden trout account.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Kern River rainbow trout. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years, whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate due to limited studies of the species in California. See methods for explanation.

Factor	Rating	Explanation
Major dams	Medium	Isabella Reservoir has fragmented the species' range and allowed for introduction of alien species.
Agriculture	n/a	
Grazing	Medium	Pervasive in the area, although less severe than in the past.
Rural /residential development	Low	Few residences; most of the subspecies' range is within Sequoia National Forest or Sequoia National Park lands.
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Trails and off-road vehicle routes can be a source of sediment influx into streams; however, most of range is in areas with minimal transportation impacts.
Logging	Low	This is an important land use in the region but probably has little direct effect on local streams.
Fire	Low	Fish-killing fires are unlikely given the sparse plant communities in the Kern Basin; fires are generally allowed to burn in national parks with unknown impacts to fish populations.
Estuary alteration	n/a	
Recreation	Medium	Off road vehicles a potential threat, but more so in past.
Harvest	Medium	Heavily harvested in past; present harvest, legal and illegal, may affect some populations.
Hatcheries	High	Constant threats of introgression, competition and predation from hatchery fish.
Alien species	Critical	Non-native trout limit distribution via hybridization, competition, predation and possible disease transfer.

Effects of Climate Change: The major predicted impacts from climate change in the range of the Kern River rainbow trout are a reduction in snow pack due to warmer temperatures, as well as a seasonal shift in peak runoff to the early spring. However, the southern Sierra Nevada is the highest part of the mountain range, and this may offset substantial reductions in snowpack, as is predicted in the northern Sierra Nevada and other regions of the state. Thus, snowmelt is likely to maintain flows in Kern River rainbow trout streams, but the timing and volume of these flows may shift over time. Nevertheless, more precipitation may come as rain, potentially earlier in the season, which may lead to increased 'rain on snow events' and corresponding flash flooding. This may be particularly acute in the Kern River, which drains a large watershed area and may suffer substantial habitat alteration or degradation associated with flood events. Since snowpack is predicted to melt earlier in the season, meadows and forests surrounding Kern River rainbow habitats are likely to become drier by the end of summer, with reduced streamflows. Elimination of grazing and other activities that compact meadows (reducing their ability to store water) and reduce riparian cover and shade may mitigate, in part, the predicted effects of climate change. Temperatures in streams are likely to increase, and it is possible that spawning may occur earlier, with unknown consequences. For these reasons, Moyle et al. (2013) list wild populations of Kern River rainbow trout as "critically vulnerable" to extinction via climate change, assuming the small, isolated, first- and second-order streams that support most populations with little introgression would be subject to increased frequency and extent of drying and warmer temperatures. Kern River rainbow trout occupying the main stem Kern may be less subject to threats of habitat loss due to drying but may be negatively affected by flood-based habitat degradation, warmer water temperatures, lower flows, and other factors.

Status Score = 1.4 out of 5.0. Critical Concern. The Kern River rainbow trout has a high probability of disappearing as a distinct entity in the next 50-100 years, if not sooner (Table 2). It is listed as a Special Concern (formerly Category 2) species by the USFWS, indicating that it is a candidate for listing but that there is inadequate information to make the determination. The American Fisheries Society considers it to be "Threatened" (Jelks et al. 2008), CDFW labels it as a "Critical Concern" species in their Fish Species of Special Concern Report (CDFW 2015), while NatureServe considers it as "Critically Imperiled."

Kern River rainbow trout are confined to a handful of streams that are subject to natural and human-caused disturbance, such as landslides and fire, even through most are in protected areas including Sequoia National Park. The greatest single threat to the continued persistence of the species continues to be invasions of alien rainbow trout, brown trout, and brook trout into their remaining streams, either through natural invasions, stocking programs, or through angler-assisted introductions. Protection of remaining populations, therefore, requires constant vigilance and the ability to react quickly to counter new threats.

Table 2. Metrics for determining the status of Kern River rainbow trout in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	1	Found in 4-6 small tributaries and short reaches of the Kern River.
Estimated abundance	2	High uncertainty about size of populations.
Intervention dependence	1	Barriers must be maintained, planting of hatchery fish managed (preferably eliminated), grazing managed, and other actions.
Tolerance	3	Presumably fairly tolerant as most rainbow trout are but not tested.
Genetic risk	1	Hybridization with introduced rainbow trout and California golden trout a constant high risk to its distinctiveness.
Climate change	1	Rated critically vulnerable in Moyle et al. (2013).

Anthropogenic threats	1	1 Critical, 1 High, and 4 Medium threats.
Average	1.4	10/7.
Certainty (1-4)	3	Least-studied of three native trout found in the Kern River Basin.

Management Recommendations: A multi-agency management plan for the upper Kern River basin, written in 1995, has as its goal to "restore, protect, and enhance the native Kern River rainbow trout populations so that threatened or endangered listing does not become necessary" (S. Stephens et al. 1995, p. 9). While this plan has been implemented, almost 20 years later the trout may still merit listing. Problems addressed in the plan still exist, including stocking of non-native trout (including hatchery rainbow trout), grazing in riparian areas, and heavy recreational use of the basin, including angling. Future management actions should be based upon recommendations in this plan and updates to address developments in the past two decades should be performed (especially data and other gap analyses). Periodic genetic sampling and abundance and distribution data are needed in order to better assess the current status of the Kern River rainbow trout and establish a baseline from which to monitor trends over time. Erickson (2013) recommends establishing a hatchery broodstock program with strict genetic protocols to improve genetic diversity throughout the basin.

The Edison Trust Fund is supposed to provide at least $200,000 each year to implement the management plan and improve fisheries in the upper Kern Basin, including developing a conservation hatchery for Kern River rainbow trout, increasing patrols of wardens in areas where recreational angling occurs, and funding genetic studies. However, the Trust has not committed these funds to the project in recent years. CDFW will use the conservation hatchery to raise and stock only Kern River rainbow trout in the Kern Basin instead of coastal rainbow trout, which is a significant improvement in stocking practices (J. Weaver, CDFW, pers. comm. 2016).

LAHONTAN CUTTHROAT TROUT
Oncorhynchus clarkii henshawi

High Concern. Status Score = 2.0 out of 5.0. Despite significant efforts in recent years, wild, self-sustaining Lahontan cutthroat trout in California face a large and increasing risk of extinction over the next 50 years.

Description: Lahontan cutthroat trout (LCT) are the largest subspecies of cutthroat trout (USFWS 2013). They are variable in coloration; lake-dwelling Lahontan cutthroats are pale gold with pink or purple hues along their flanks and gill plates and a back that ranges from a greenish-bronze to dark olive color. In streams, LCT attain smaller sizes than those in lakes, and may exhibit purple or bluish hues along their flanks. LCT have large rounded black spots distributed over the back and caudal peduncle, but most spots are found above the lateral line. They possess characteristic yellow to red streaks of color along the underside of the mandible that give them their name. These marks are often absent or extremely faint in fish smaller than 8 cm TL. They have 21-28 gill rakers, and there are 40-70 pyloric cecae present. They have teeth on the upper and lower jaws, head and shaft of the vomer, palatines, tongue, and basibranchial bones. Their scales are generally smaller than those of rainbow trout, with 150-180 along the lateral line. Parr possess 8-10 narrow parr marks along the lateral line, which they often retain to adulthood in streams (Behnke 1992, 2002, Moyle 2002).

Taxonomic Relationships: Lahontan cutthroat trout are one of the most genetically distinct cutthroat trout subspecies, reflecting their long history of isolation from other salmonids. Lahontan cutthroat trout (LCT) are close relatives of the Paiute cutthroat trout, and are divided into three distinct population segments based on geographic distribution, ecology, behavior and genetics (USFWS 2010). However, recent genetics work (Peacock et al. 2010) suggests that a fourth segment should be designated for fish in the Steen Mountains, known as the Willow-Whitehorse basin in southeastern Oregon. The Western Lahontan basin segment is comprised of fish in the Truckee, Carson and Walker river basins (California and Nevada, Trotter 2008). Pritchard and colleagues (2012) found evidence to support the theory that Paiute cutthroat trout evolved from Lahontan cutthroat trout in the Carson River basin nearly 12,000 years ago. They also found 34 distinct segments of DNA that distinguish LCT from Paiute cutthroat trout, adding further weight to the subspecies status of Paiute cutthroats. LCT, like most other cutthroat trout, readily hybridize with rainbow trout, and can also spawn with Paiute cutthroat trout to create fertile offspring, resulting in loss of phenotypic expression and genetic diversity over time (California Trout 2015). This vulnerability has been a driver of their decline across their range.

Life History: Through their range of diverse habitats, LCT exhibit riverine resident, migratory, and lacustrine life histories, making them very adaptable to a range of conditions (Behnke 2002). Despite this habitat adaptability, LCT are very intolerant of other salmonids and rarely coexist with them (USFWS 2013). While LCT can thrive in lake environments, they are obligate stream spawners. Lacustrine trout are capable of making extensive migrations to find suitable spawning areas. Trotter (2008) indicates that some trout from Pyramid Lake in Nevada historically ascended the Truckee River to spawn in tributaries to Lake Tahoe.

They feed primarily on terrestrial and aquatic invertebrates in both larval and adult phases, as well as oligochaetes and other non-insect macroinvertebrates such as zooplankton

(USFWS 1995). Large LCT will feed on juvenile fish of other species when such food is abundant, and can attain great sizes of over 9kg; they historically attained even larger sizes in Pyramid Lake, Nevada, and supported a major commercial fishery there for decades (USFWS 2013). LCT generally can live from 4 to 9 years, with stream-dwelling fish having shorter life spans than lake-dwellers. However, there is anecdotal evidence that LCT in Independence Lake, CA may live as long as 13 years (W. Somer, CDFW, pers. comm. 2016).

Spawning takes place in streams from April to July depending on stream flow, water temperature and elevation (Peacock and Dochtermann 2012). However, autumn spawning runs have been reported from some populations (USFWS 2013), and spawning migrations occur at observed water temperatures between 5 and 16° C (USFWS 1995). LCT reach reproductive maturity at age 3 to 4 in stream habitats (Peacock and Dochtermann 2012). Consecutive-year spawning is unusual, and there is approximately 60-70% post-spawning mortality for females and 85-90% for males (USFWS 1995). Only 50% of surviving females spawn again as compared to 25% of males (USFWS 2013). Lacustrine females have higher fecundity rates than do riverine females based on their greater size and age. Lacustrine LCT females can produce 600 to 8,000 eggs each, while smaller stream-dwelling female LCT produce only 100 to 300 eggs. Gravel from 6 to 50 mm is optimum for redd construction and embryo incubation (Coffin 1981). Preferred water depths for redds average 13 cm, and velocities average 56 cm/s (Schmetterling 2000). Water must be saturated with oxygen and have minimal siltation to prevent eggs from suffocating. Spawning LCT develop bright red coloration on the underside of the mandible, operculum and along their flanks; coloration is more intense in males, which also display a hooked lower jaw during spawning.

Eggs hatch after incubating from 4 to 6 weeks, depending on water temperature, and fry emerge from gravel after 13 to 23 days (USFWS 1995). Fry can spend up to two years in their natal stream before migrating to available lake habitat, but most migrating fish move after their first summer (Trotter 2008). Growth rates vary with water temperature and food availability. Faster growth occurs in larger, warmer waters, particularly where forage fish are available. Summit Lake-strain fish at the Lahontan National Fish Hatchery Complex have recorded sustained growth rates up to 150 mm each year (USFWS 2016). In smaller, colder water bodies, growth is slower and longevity is reduced. Gerstung (1986) found mean fork lengths of LCT from 6 streams in the Sierra Nevada to average 89, 114, 203, and 267 mm at ages 1-4, respectively.

Habitat Requirements: LCT are noteworthy for being very adaptable and tolerate a variety of habitats and temperatures - from deserts to high alpine tributary streams - in California. The species can persist in streams where temperatures may exceed 27°C for short periods and can fluctuate 14-20°C in a day. They can also survive prolonged exposure to temperatures of 23-25°C, but cease to grow when temperatures exceed 22-23°C (Dickerson and Vinyard 1999, Moyle 2002) and show sublethal impacts such as reduced feeding, growth rates, and movement at temperatures above 24°C (Robison et al. 2008). Ideal summer temperatures for growth and development average 13°C ± 4° C (Hickman and Raleigh 1982), and recent studies (Peacock and Dochtermann 2012) suggest that upper thermal tolerances are heritable traits to some degree. Lacustrine LCT also have a considerably higher tolerance for alkalinity, total dissolved solids, and lower dissolved oxygen than most freshwater fish (Koch et al. 1979).

In streams, LCT prefer reaches with well-vegetated and stable stream banks, greater than 50 percent stream cover, and pools with close proximity to cover as well as riffle-run complexes for spawning and feeding (USFWS 2013). Stream substrate composition, cover, geomorphology, and water quality constrain LCT distribution; recent mark-recapture studies (Alexaides et al. 2012) indicate that LCT, like most other cutthroat trout, utilize pool habitat more than any other habitat type. Lack of such habitat caused increased migration distances and range sizes in individual tagged fish, and was associated with higher mortality in tagged fish in the Truckee River. Habitat diversity is important for survival of young-of-year fish, especially during winter months when LCT survival is lowest. Nick points, or transition areas between more and less confined valley stream reaches tend to concentrate LCT, indicating that hyporheic subsurface flows provide refuge from warm temperatures in summer and anchor ice in winter and create relatively deeper pools in shallow stream reaches (Boxall et al. 2008).

Distribution: Lahontan cutthroat trout are native to the greater Lahontan basin in eastern California, southern Oregon, and northern Nevada (Trotter 2008, Figure 1). In California, LCT were historically found in large, low-gradient rivers such as the Truckee River, moderate gradient streams such as the Carson and Walker rivers, and small, headwater streams at high elevations such as the many tributaries to Lake Tahoe. They were also present in the Susan River drainage in the Eastern Sierra, but have been extirpated from that drainage for some time (Trotter 2008). LCT remaining in California represent the western and southernmost extent of the species in their native range, making these periphery populations susceptible to habitat degradation and disturbance (Haak et al. 2010).

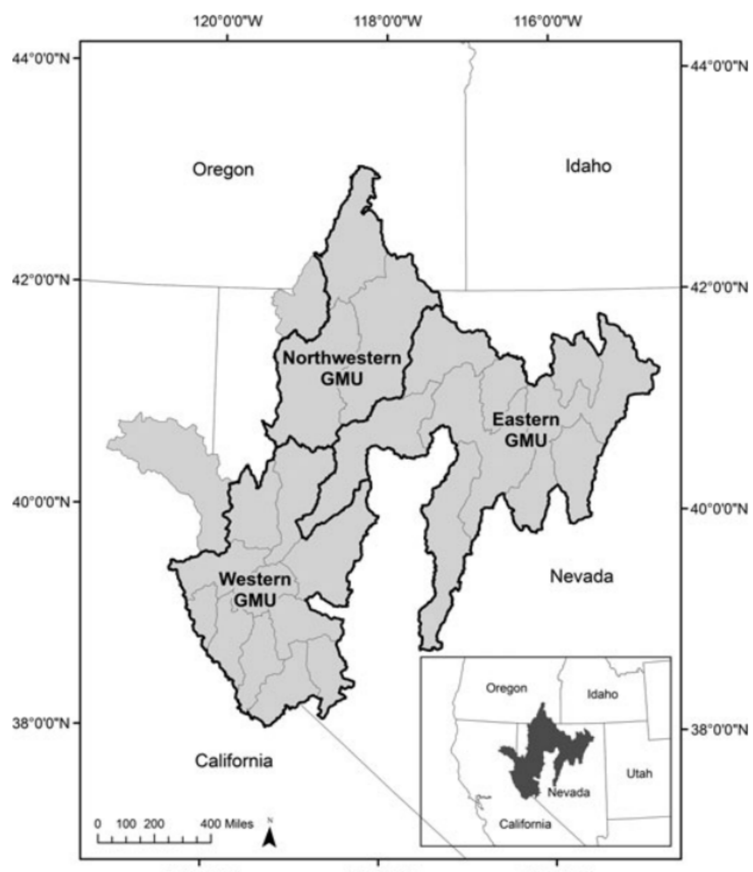


Figure 1. LCT Geographic Management Units. From: Pritchard et al. 2013. Fig 2, pg. 278.

The species first occupied Lake Lahontan during the Pleistocene about 25,000 years ago, which receded over time, leaving remaining populations at the former lake edges. Gerstung (1986) reported that LCT distribution in 1844 included some 11 lacustrine populations occupying approximately 334,000 acres of lakes, and between 400 and 600 fluvial populations over 3,600 miles of streams in the Lahontan Basin. They now occur in a wide variety of cold-water river and lake habitats, ranging from terminal alkaline lakes such as Pyramid and Walker Lakes to the alpine oligotrophic waters of Lake Tahoe and Independence Lake. In the Truckee Basin, LCT from Pyramid Lake migrated upstream to spawn in tributaries to Lake Tahoe, as well as in the main river. In the Carson, Walker, and Truckee basins, only a few scattered streams continue to maintain LCT populations (Trotter 2008, Table 1). In total, there are only 16 lakes and streams that are known to still contain LCT within their historical range in California. LCT have also been planted and become established in several creeks outside their historical range, including west slope drainages near the Truckee basin such as Austin Meadow Creek (Nevada County), Pole Creek (Placer County), and others.

Table 1. List of known populations of Lahontan cutthroat trout in their native California range.

Carson River Drainage	**Truckee River Drainage**	**Walker River Drainage**
East Fork Carson River* Murray Canyon Creek* Raymond Meadows Creek** Poison Flat Creek* Golden Canyon Creek* Heenan Lake*	Independence Lake* Upper Independence Creek* Pole Creek** Upper Truckee River** Fallen Leaf Lake**	Murphy Creek** Slinkard Creek** Mill Creek** Wolf Creek** Silver Creek** By-Day Creek* Bodie Creek***
*Reintroduced populations of Independence Lake strain, actively managed by CDFW **Population may have been lost in drought due to desiccation	*Population in historical range **Historical populations once extirpated but reintroduced	*Historical population maintained above a barrier **Historical populations once extirpated but reintroduced ***Population may have been lost in drought due to desiccation

Table 2. Known populations of LCT outside their historical native range (W. Somer, CDFW, and C. Mellison, USFWS, pers. comms. 2016).

Yuba River Drainage	**Stanislaus River Drainage**	**Mokelumne River Drainage**	**San Joaquin River Drainage**	**Owens River Drainage**
Macklin Creek Austin Meadows Creek East Fork Creek Unnamed tributary to East Fork Creek	Disaster Creek	Marshall Canyon Creek Pacific Valley Creek Milk Ranch Creek	West Fork Portuguese Creek Cow Creek	O'Harrel Canyon Creek

Trends in Abundance: As a species, LCT have lost over 30 remote, isolated populations between 1980 and 1995, and more local extirpations are likely to continue due to impacts stemming from interactions with nonnative species, habitat fragmentation and degradation, and climate change (Peacock et al. 2010). CDFW's (2016) estimate that LCT persist in less than 5% of their original stream habitat and in a mere 0.4% of their original lake habitat in California indicates that remaining LCT populations are a fraction of what they once were. Wild, self-sustaining populations in headwater streams in California likely total only a few hundred fish age 1+ and older (CDFW 2009). The small Bodie Creek and Raymond Meadows Creek populations that were opportunistically established by biologists in California may have been lost due to impacts associated with the ongoing drought (C. Mellison, USFWS, W. Somer, CDFW, pers. comms. 2016). To help combat this ongoing loss of populations, LCT have been established in nine creeks outside of their native range in California through stocking, nonnative species removal, barrier creation, and active management (Table 2). While abundance estimates are lacking, all stream populations most likely contain less than a few hundred adult fish, perhaps with the exception of the Upper Truckee River. Independence, Fallen Leaf, and Heenan Lakes contain more mature adults as a result of active management, hatchery stocking, or both, but reliable population estimates for lake habitats are not available for these habitats.

To bolster LCT populations in California, CDFW and USFWS are working with US Forest Service and other partners to continue to monitor reintroduction efforts on the Upper Truckee River. From 2007 to 2009, the number of LCT captured rose each year (1,700, 1,900, and 2,400, respectively) across repeat surveys in the same reaches, indicating the population is increasing and multiple year classes continue to survive and reproduce (CDFW 2009). Recent recovery efforts to remove invasive brook and brown trout in Upper Independence Creek, spawning tributary to Independence Lake and stronghold of natural spawning of LCT in their historical California range, has also shown some promise. Following several years of brook trout removals through strategic stream de-watering and electrofishing, management partners have begun de-watering the main lake in the fall to desiccate kokanee eggs spawned in the lake (C. Mellison, USFWS, pers. comm. 2016). Over several years, LCT juveniles emigrating from the creek jumped from about 14,000 to 40,000 individuals (Scoppettone et al. 2012). In 2010, 237 adult LCT passed the weir en route to spawning grounds in Upper Independence Creek, representing the largest number of potential spawners recorded in recent memory, perhaps due to increased survival of fry and juveniles after removal of brook trout (C. Mellison, USFWS, pers. comm. 2016). In follow-up surveys in lower Independence Creek, multiple size and age classes of LCT were observed, indicating that juvenile LCT are exhibiting a shift to a more resident life history by rearing in the creek for longer timeframes. This change in habitat utilization is perhaps increasing their fitness and survival to spawning age (CDFW 2012), but no LCT population studies have been conducted at Independence Lake recently.

Factors Affecting Status: Major factors affecting LCT habitat and abundance, especially alien species, are discussed below.

Alien species. Non-native trout introductions and invasions pose the single greatest threat to the continued persistence of LCT in California. Lahontan cutthroat trout were the only salmonid historically found in the Eastern Sierras, with the exception of the closely-related Paiute cutthroat trout and Eagle Lake rainbow trout and. They do not persist in habitats with non-native trout, except in Independence Lake, though recent studies suggest that active removal and management of nonnative salmonids are essential for LCT persistence in this remaining stronghold for the species in California (Rissler et al. 2006, W. Somer, CDFW, pers. comm. 2016). Brown trout are voracious predators on juvenile LCT and they are fall spawners, which gives juvenile brown trout a competitive advantage over LCT, which emerge from gravel in spring. Brook trout occur in much higher densities than other trout and effectively outcompete LCT for habitat and resources. Rainbow trout life history strategies and spawn timing overlap with LCT, causing frequent hybridization between the two species, resulting in dominance of rainbow trout-type phenotypes (Al-Chokhachy et al. 2009). Lake trout have also contributed to the demise of LCT from Lake Tahoe and Fallen Leaf Lake through predation, competition, and perhaps disease. Vander Zanden et al. (2003) indicate that the food webs of Lake Tahoe are now so altered from interactions with introduced species such as lake trout that re-establishing cutthroat trout in the lake may not be possible, though the US Fish and Wildlife Service has been actively stocking Fallen Leaf Lake for LCT reintroduction since 2002 (USFWS 2013). More recent introductions of centrarchids and other non-native fish and invertebrates will hamper restoration efforts even further in the future (W. Somer, CDFW, pers. comm. 2016).

The Lahontan Basin has changed dramatically as a result of human uses, such as dams and diversions for power generation and agriculture, that such interchanges are no longer possible. This makes it very challenging to maintain genetic diversity in isolated populations without genetic drift, founder's effects and possible inbreeding depression. The lack of interconnected habitat and large populations of non-native trout together effectively eliminate the possibility for recovering historical self-sustaining natural meta-populations throughout most of their range. Only limited LCT populations survive in isolated streams, leading to a dearth of genetic diversity that likely does not represent the entire historical genome of the species. Because LCT historically inhabited many isolated subbasins, there were presumably many genetically distinct populations with local adaptations that have been lost. USFWS (2010) recommended that remaining populations should be maintained in their basins of origin to the extent practicable and translocated where appropriate to reduce introgression and competition with nonnative trout and to safeguard genetic stocks. Lacustrine LCT are most at risk of losing their genetic integrity because there are only two small, naturally-reproducing populations left within their native range in California (Independence and Fallen Leaf lakes) (USFWS 2013).

Major dams. Dams are present on most major Lahontan cutthroat streams, fragmenting habitats, creating barriers to migration, and removing large areas in reservoirs and rivers that were once suitable LCT habitat. Agricultural water diversions (mainly from Derby Dam effectively disconnected Pyramid Lake from the Truckee River for most of the 20th century, lowering lake levels nearly 24 m and greatly increasing the lake's alkalinity. As the result of federal listing of LCT and Lahontan sucker, or "cui-ui," (*Chasmistes cujus*) as threatened species, flows have been somewhat restored to the river and lake levels have risen, but habitat is still reduced compared to historical levels. The Truckee Meadows Water Authority manages water withdrawals for downstream users, and stands committed with partners from The Nature Conservancy, California Department of Fish and Wildlife, US Forest Service, US Fish & Wildlife Service, Tahoe National Forest, and local conservation groups to helping restore and protect LCT populations in the basin. In the Walker River basin, the LCT population persisted until Bridgeport Dam was built in 1924, followed by Weber Dam in 1933, effectively cutting the population off from its spawning habitat.

Natural barriers also helped shape LCT distribution and life history over time. While native beavers create barriers, they may provide important benefits to LCT across the Eastern Sierra through dam creation that maintains water in streams throughout summer months and drought, as seen in the Upper Truckee meadow habitat over the last several years (C. Mellison, USFWS, pers. comm. 2016). Beaver dams can create habitat complexity, slow and store water, create deeper pool habitat, and disrupt solid-freezing of small headwater streams in winter. Conversely, their dams may negatively impact LCT by creating barriers to migration, increasing water temperatures in pools directly upstream, downstream siltation in the event of a washout during high flow events, and potentially reducing water quality downstream through blooms of iron-fixing bacteria (C. Mellison, USFWS, pers. comm. 2016).

Harvest. Heavy commercial fishing in lacustrine populations in the 19th and early 20th contributed to the collapse of commercial fisheries for Lahontan cutthroat trout in the 1940s and their eventual extirpation from Pyramid Lake and Lake Tahoe (Trotter 2008, Al-Chokhachy et al. 2009). Townley (1980) estimates that between 1873 and 1922, approximately 100,000-200,000 pounds of LCT were annually harvested from Pyramid Lake and the Truckee River system alone. By the 1940s, LCT were extinct in Pyramid Lake, and hatcheries were required to support the popular sport fishery there. Those populations continue to this day thanks to complete hatchery support from the Lahontan National Fish Hatchery Complex in Gardnerville, NV. In California, mandates for barbless hooks and catch-and-release angling for LCT have helped reduce impacts of harvest on this species, though the impacts of poaching on refuge populations are unknown (Alexaides et al. 2012).

Logging. Watersheds containing LCT were heavily logged in the 19th century to provide timber for mines in Nevada and railroad ties, removing vegetation and increasing silt loads in the rivers (Trotter 2008). In many LCT streams, water was either diverted down flumes to carry logs, or was impounded behind splash dams and then abruptly released to wash logs to downstream sawmills; the alternating drying and flooding destroyed habitat and depleted fish populations in the mainstem Truckee River (USFWS 1995). Historical logging in the species' native range discharged large amounts of industrial and sewage wastes directly into streams until the 1930s, further degrading habitat and water quality (USFWS 1995).

Grazing. Heavy livestock grazing throughout the LCT range, especially of cattle in riparian zones, has degraded habitat through trampling of banks and riparian vegetation leading to erosion, channel incision and siltation of streams. Resulting loss of riparian vegetation and cover has resulted in higher water temperatures and reduced cover, leaving fish more vulnerable to reduced water quality and predators. As much as 70% of LCT habitat occurs on Bureau of Land Management and National Forest lands, where grazing has historically been permitted. Public lands are far less heavily grazed today, which allows beaver to thrive, but active grazing still occurs throughout LCT range (C. Mellison, USFWS, pers. comm. 2016). In areas where grazing still occurs, canyon areas should be prioritized for cattle exclusion to protect areas of hyporheic flow that provide refuge (Boxall et al. 2008). For a more complete account of grazing on high elevation trout populations, see the California Golden Trout species account.

Mining. The historical effects of mining on fish populations is not well understood in California, in part because the most impacts occurred during the Gold Rush of the mid-19th century. Placer mining removed instream habitat and diverted water, while hardrock mining released debris, sediment, and toxic effluent into streams from mines. These activities took place across Lahontan cutthroat streams, but their impacts are largely unrecorded (Trotter 2008).

Disease. A number of parasites and pathogens have potentially adverse effects on LCT and their recovery. Heenan Lake and Lahontan National Hatchery Complex have been sites of significant disease outbreaks in the past, increasing the potential for negative population-scale impacts on wild populations. Hatcheries often present the biggest risk of exposure because they often recycle their water and have fish in close proximity. The release of infected hatchery fish could result in transmission of pathogens to wild fish populations. There have been widespread reports of *Renibacterium salmoninarium*, the causative agent of bacterial kidney disease, in both hatchery LCT and wild trout within the historical range of LCT as well as whirling disease and bacterial gill disease (J. Stead, URS, pers. comm. 2007). In the past, such diseases have infected LCT in hatcheries and have reduced the ability of CDFW to plant these fish (W. Somer, CDFW, pers. comm. 2007). Any hatchery operations carry the risk of disease spread to wild populations, but evolving hatchery management practices ameliorate these risks to the extent practicable.

Table 3. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Lahontan cutthroat trout in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Threat	Rating	Explanation
Major dams	Medium	Dams are present on most major LCT lacustrine and fluvial habitats, fragmenting populations and impeding essential expression of various life histories.
Agriculture	Medium	Water diversions and reduced instream flows from irrigation reduce suitable habitat and increase water temperatures; these impacts are compounded in drought.
Fire	High	Fire suppression, coupled with increasing aridity predicted with climate change, may contribute to increased fire frequency and intensity, posing serious threats to small, isolated populations.
Grazing	Low	Historically pervasive in the native range of LCT, especially in meadows and along riparian corridors, but impacts have been reduced in recent decades through grazing allotment closures.
Rural /residential development	n/a	Majority of LCT habitat exists within National Forest boundaries.
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Historical mining in LCT watersheds, but impacts unknown.
Transportation	Low	Roads are sources of erosion and sediment into streams; culverts have blocked access in the past, but impacts are likely small.
Estuary alteration	n/a	
Logging	Low	Logging and associated roads have likely contributed to stream degradation, increased water temperatures through riparian degradation, and reductions in water quality due to runoff, though much high-elevation habitat of LCT is sparsely wooded.
Recreation	Low	Little recreational use of most LCT habitats occurs.

Harvest	Low	Legal fishing pressure is light and limited to catch-and-release angling with barbless hooks only.
Hatcheries	High	Competition and hybridization with historically stocked rainbow trout represent a major threat to LCT persistence; Hatcheries will play a role in recovery of LCT.
Alien species	Critical	Historical and recent non-native trout introductions limit LCT recovery via competition, predation, and hybridization.

Effects of Climate Change: Despite their high thermal tolerances and adaptability, LCT in California are critically susceptible to the effects of climate change, such as shrinking available habitat due to increasing temperatures and interactions with nonnative species as their ranges shift (Wenger et al. 2011). Most populations of LCT in California persist in small, high-elevation headwater streams above barriers and devoid of other salmonids less than 8km in length; these areas represent marginal habitat at best for cutthroat trout. Fragmentation has reduced habitat availability, connectivity, and suitability, which over time diminishes gene flow and can lead to inbreeding depression. Climate change is likely to exacerbate these stressors through higher summer temperatures, decreased streamflow, desiccation, and increased frequency and magnitude of catastrophic fire (USFWS 2013).

In the past, LCT likely persisted in these harsh environments by expressing varied migratory life histories in interconnected habitats to find thermal refugia. In water over 22°C, LCT exhibit sublethal effects as well as increased mortality; the portion of their habitats above this threshold in California will continue to expand spatially and temporally in the face of a changing climate, putting the continued persistence of the species in serious jeopardy. Remaining LCT populations at the extreme edges of the species' range is crucial because these individuals have the greatest potential to adapt to increasing temperatures as climate change continues to intensify. Recent work (Warren et al. 2014) on cutthroat susceptibility to climate change impacts indicates that the Northern portions of LCT, not the southern populations in California, are the most at risk of losing potential habitat, as latitude plays an important role in shrinking habitat due to thermal tolerance thresholds butting up against topographic constraints. It is likely that the downstream elevation limits of LCT habitat will remain only in higher elevations in the California portion of their range in the future due to short growing seasons, harsh winters, and high variance in temperature extremes that will constrict their ranges. As their available downstream habitat shrinks, LCT will overlap even more with those of nonnative salmonids, changing species interactions, increasing competition and reducing survival (Wenger et al. 2011). LCT are unlikely to find refuge further upstream or in higher elevations in most areas due to natural barriers such as velocity, gradient, and flow limitations. As their distribution changes over time, LCT are likely to become even more restricted to smaller, lower-flow headwater streams, resulting in overall decreases in abundance and biomass, exacerbating their vulnerability to loss of genetic variability and local extirpations (Wenger et al. 2011).

Status Score = 2.0 out of 5.0. High Concern. Wild, self-sustaining populations of Lahontan cutthroat trout in California have a high likelihood of extirpation in their native range in the next 50 years, except as populations sustained by hatchery production. Wild populations of Lahontan cutthroat trout in California are small and isolated and will require continuous and enhanced management efforts over current levels to prevent extinction. Most remaining populations occupy marginal habitat in small, isolated areas outside of their native range, and are supported by intensive management of nonnative salmonids, stocking, and the coordinated recovery efforts of many partners. Local extirpations of isolated populations likely occurred during the ongoing drought, further reducing the species' remnants throughout its range. They are not formally listed under California's Endangered Species Act, but are instead specially managed as a heritage trout species by CDFW. A summary of the threats they face and their relative vulnerability is described below (Table 4).

Table 4. Metrics for determining the status of Lahontan cutthroat trout in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Occupies multiple watersheds in California, but no connectivity among them.
Effective adult abundance	2	Most wild populations are significantly less than 1,000 fish each, with lacustrine habitats and Upper Truckee River as exceptions.
Intervention dependence	1	Hatchery program using wild brood stock required for persistence.
Tolerance	5	LCT are fairly long-lived and demonstrate broad physiological tolerance and are iteroparous.
Genetic risk	1	Hybridization risk and loss of genetic variation is well documented and the major threat to the species.
Climate change	1	LCT are extremely vulnerable to climate change in all watersheds inhabited.
Anthropogenic threats	2	1 Critical and 2 High Factors.
Average	2.0	14/7.
Certainty (1-4)	4	Peer reviewed literature, agency reports, grey literature, and professional judgment.

Management Recommendations: Since their original listing under the Endangered Species Act, Lahontan cutthroat trout were subsequently relisted as threatened in 1975 to facilitate management activities (USFWS 1995). While LCT have often persisted in isolation throughout their history, their isolation has never been as great as it is now, and extant populations are mostly maintained by stocking or managed for non-native species control. Genetic drift and stochastic events are significant challenges in current LCT management and hinder growth of effective population size growth. LCT face tremendous odds of recovery despite the combined work of agencies, landowners, non-profits, and other partners; it is unlikely that they will be delisted in the near future. According to the USFWS recovery plan, priorities include:

1. Eradicating non-native salmonids and reintroducing LCT in their native range
2. Connecting fragmented habitats so LCT can express all their life histories
3. Monitoring population genetics and maintain genetic diversity
4. Supporting water transactions and acquisitions to restore instream flows
5. Enhancing riparian and in-stream habitat and complexity through restoration
6. Restoring connectivity between and among fragmented, isolated populations

With the goal of recovering and delisting LCT in mind, there have been considerable efforts at restoring populations of LCT both inside and outside of their native range in California over the last few decades, including recent efforts to control nonnative trout and monitor and restore existing populations in Upper Truckee River near Meiss Meadows, Upper Independence Creek, Heenan Creek and Lake, nearby Slinkard, Wolf, and Silver creeks, Fallen Leaf Lake, Glen Alpine Creek, Austin Meadows Creek, By-Day Creek, and others. California's periphery populations of LCT at the western and southern extent of their range represent important sources of genetic information and ecological adaptation to harsh conditions (Haak et al. 2010). Hatchery propagation of LCT has also been ongoing since around 1939 and continues today to support LCT recovery through maintenance of genetic information and releases from the Lahontan National Fish Hatchery totaling 300,000 to 400,000 LCT a year in Nevada and Fallen Leaf Lake, and the Heenan Lake fish hatchery spawning tagged LCT for use in statewide stocking programs.

Habitat fragmentation and alien trout invasion throughout LCT range and limited resources, political will, and time required to reverse those impacts make recovery a daunting task. While nearly all current LCT habitat in California is on public (National Forest) land, almost all of the reintroduced populations exist in previously fishless waters above barriers. CDFW restoration goals are to protect and expand existing wild populations of LCT using existing populations as sources for reintroduction into historical or out-of-basin refuge habitats. Toward that end, CDFG and USFWS have spent considerable resources in maintaining genetic diversity in LCT populations, and have begun reintroducing LCT in numerous locations. Their persistence will require innovative management, habitat restoration, and elimination of competing species of trout from streams and lakes in a variety of habitats so LCT can express varied life history forms to adapt to changing conditions in the face of climate change.

Independence Lake, in the Truckee River drainage, is the only lacustrine population left in the state where LCT have continuously survived and reproduced independently. Strong partnerships and commitment allow crucial adaptive management to continue to benefit LCT in this stronghold of LCT in California. The US Geological Survey, US Forest Service, The Nature Conservancy, CDFW, USFWS, and other regional partners collaborate to provide research, expertise, and fieldwork to help conserve and recover the LCT population here. Such collaborative efforts should be supported and replicated where possible to coordinate activities that leverage limited resources and maximize potential benefits.

Efforts to protect the endangered cui-ui in Pyramid Lake have resulted in increased flows in the Truckee River, thus raising the lake level, reducing its alkalinity, and providing access to the Truckee River for spawning, which benefits LCT as well. LCT have been observed migrating through the Truckee River delta to the fish elevator at the base of Marble Bluff Dam as well as swimming up the fish ladder around the dam on spawning runs, (G. Scoppettone, pers. comm. 2007). In 2014 and 2015, increased flows allowed large LCT to enter the Truckee River and successfully spawn for the first time since 1938 (USFWS 2016), though considerable efforts will be required to connect this population to its historical access to Lake Tahoe and tributaries.

In the Upper Truckee River basin near Meiss Meadows, CDFG has worked to restore LCT since the 1980sthrough brook trout eradication, barrier maintenance, and habitat restoration. During 1988 through 1995, a series of labor-intensive CDFG rotenone treatments were undertaken in the headwaters to eradicate invasive brook trout and reintroduce LCT to the basin. Electrofishing has been conducted each year to keep brook trout under control and preserve the small headwater reach for LCT; CDFG has successfully eradicated brook trout from the stream in 2008, though speckled dace have been found upstream of the rock wall barrier and so annual efforts to monitor the effectiveness of the barrier will continue (CDFW 2009).

Current management is focused primarily on maintaining genetic diversity and reintroducing LCT in streams and lakes with high potential of success. The lack of interconnected watersheds to support metapopulations, however, will only persist with significant support from managers. Restoration goals should weigh actions that increase short-term viability of populations against maintaining and increasing genetic diversity for the long-term (Peacock and Dochtermann 2012). It is likely that with the continued efforts of CDFW, USFWS, the Forest Service, and USGS, there will be some improvement in LCT populations in a few watersheds, but it is unlikely that LCT will be able to persist indefinitely in self-sustaining wild populations in California.

Despite the impacts of drought and potential loss of some populations, non-introgressed, wild LCT may have actually increased in abundance since 2008. The considerable management efforts on Upper Truckee River, Independence Lake and Creek, Heenan Lake and Creek, Fallen Leaf Lake and tributary Glen Alpine Creek, Austin Meadows, By-Day, Slinkard, Wolf, and Silver creeks have likely helped maintain and even bolster populations in some cases. Small populations such as Raymond Meadows and Bodie creeks may have been lost during the drought due to increased water temperatures and desiccation, while others were saved through rescue efforts. In July and August 2015, 49 and 37 LCT, respectively, were rescued from By-Day Creek in Mono County and released in suitable habitat in small groups in restored habitat in Wolf and Slinkard Creeks (CDFW 2015). In July 2014, CDFW monitored a small out-of-basin population in 2.1km of Austin Meadow Creek in Nevada County. During visual surveys, CDFW staff located illegal pumps, and worked in cooperation with other agencies and authorities to remove the diversion and reestablish connectivity between isolated pools to provide habitat for about 50 LCT individuals to persist through the drought (CDFW 2016).

In addition, considerable hatchery production, stocking, and active nonnative trout management in Fallen Leaf and Independence Lakes has helped bolster LCT numbers. In Fallen Leaf Lake, annual stockings of fingerling and yearling LCT since 2002 from the Lahontan National Fish Hatchery Complex have allowed a small population of adults to take hold, and natural reproduction is occurring as a result in the Glen Alpine Creek tributary (Smith 2013). Active management of introduced rainbow and brook trout in the creek and installation of two weirs allow passage of native species, while excluding mature trout has helped reduce interbreeding and restore a small foothold for natural LCT spawning in California (USFWS 2013). At CDFW's broodstock reservoir on Heenan Lake, manual rainbow trout removal is increasing prospects for natural Independence-strain LCT spawning in California in the near future. Only two rainbow trout were captured during electrofishing surveys in Heenan Creek in 2012, with no young-of-year fish seen, indicating that the spawning cycle of introduced fish has been broken. Now, after three consecutive years of capturing zero rainbow trout in the creek, CDFW can consider treatment of Heenan Lake to ensure Independence strain LCT remain so they can eventually spawn naturally in this watershed (CDFW 2012).

In the long-term, persistence of wild LCT in California will continue to require intense management, and remaining wild populations are likely to remain small and scattered even with significant allocation of resources to connect populations and allow LCT to re-colonize historical habitat void of nonnative trout. The species will continue to suffer from lack of genetic diversity, alien trout, and habitat fragmentation and alteration. Additionally, climate change will adversely impact LCT by increasing stream temperatures and reducing flows in some small streams during the summer and fall. Available habitat for LCT will likely continue to shrink based on upper thermal tolerances, shifting species distributions and resulting interactions with nonnative species, and alterations to streamflow. Hatcheries will continue to maintain sport fisheries for LCT based on changes to existing stocking practices from state legislation changes. The crucial California Fish and Game Code 1729 mandates that CDFW prioritize stocking native hatchery-produced species in place of nonnative species in state waters, and has used strategical stocking in areas known to contain LCT or where LCT could become established again. To date, the idea of utilizing a conservation hatchery for LCT in California has been tabled because the risks of genetic contamination of remaining strains of fish are too high for the perceived benefits of potential increases in genetic diversity. In the future, this may change. An aspirational goal of the Lahontan National Fish Hatchery Complex is to introduce large Pyramid Lake LCT back into the mainstem Truckee River above barriers to spawn and re-gain a foothold in their historical habitat and allow them to one day return beyond barriers in the Truckee-Tahoe basin volitionally (USFWS 2016).

In the next several years, state and federal resources that have been focused on restoring Paiute cutthroat trout will likely be freed up in the coming years and shift to LCT recovery priorities, which should further help the plight of the species (W. Somer, CDFW, pers. comm. 2016). Whether the political will, funding, and concerted effort of partners is available to continue and expand recovery efforts on a sufficient scale and timeframe remains to be seen.

LITTLE KERN GOLDEN TROUT
Oncorhynchus mykiss whitei

High Concern. Status Score = 2.0 out of 5.0. The Little Kern golden trout is vulnerable to extinction in its native range in the next 100 years. The restoration of Little Kern golden trout is ongoing, as recovery shifts from one focused on reducing hybridized populations to improving genetic diversity, population connectivity and size, as well as expanded distribution.

Description: This subspecies is similar in appearance to California golden trout but is not as bright in color (Behnke 2002). It also tends to have more small spots on the body and have more (ca.10) distinct parr marks. It has fewer scales along the lateral line (usually 155-160) than California golden trout, but more pyloric cecae (35-40) and more vertebrae (60-61).

Taxonomic Relationships: The complex history of nomenclature and taxonomy of the golden trouts is described in Behnke (2002) and in the California golden trout account in this report. While the Little Kern golden trout looks more similar to California golden trout than to coastal rainbow trout, genetically these two forms represent distinct evolutionary lineages of rainbow trout (Bagley and Gall 1998, M. Stephens 2007).

Life History: Only limited life history studies are available on this subspecies, but its life history is presumably similar to that of well-studied California golden trout, as described in this report. Spawning behavior, as described by Smith (1977) is similar to that of other rainbow trout, although Little Kern golden trout are known to produce fewer eggs during spawning and are generally less fecund. Konno (1986) showed the fish have relatively small home ranges compared to other inland salmonids.

Habitat Requirements: Little Kern golden trout have similar habitat requirements as California golden trout in the neighboring South Fork Kern River and Golden Trout Creek. They are adapted for living in small, meandering meadow streams and higher gradient tributaries. Myrick and Cech (2003) found that these trout are physiologically adapted to temperatures of 10-19°C. They co-occur with Sacramento suckers in some areas (Moyle 2002).

Distribution: This subspecies is endemic to roughly 160 km of the Little Kern River and tributaries, where it was isolated from the rest of the Kern River basin by natural barriers (Christenson 1994; Behnke 2002). By 1973 their range had shrunk to five headwater streams in the basin (Lower Wet Meadows Creek, Deadman Creek, upper Soda Spring Creek, Willow Creek, and Fish Creek) plus an introduced population (originating from Rifle Creek) in Coyote Creek, a tributary to the Kern River (Ellis and Bryant 1920; Christenson 1984). It was determined later that the upper Coyote Creek population was genetically influenced by California golden trout (M. Stephens 2007). Excluding Coyote Creek, the 1973 distribution of Little Kern golden trout included about 16 km of habitat. Beginning in 1974, systematic efforts were made by DFG and other agencies to restore Little Kern golden trout to its historical range, by applying rotenone to streams and lakes in the drainage, constructing barriers to immigration of non-native trout, and rearing Little Kern Golden trout at the Kern River Planting Base near Kernville. Between 1974 and 1996, approximately 100 applications of piscicides were used to reduce introduced or hybridized populations, 27 fish barriers were used to isolate populations, and nearly 80,000 fish were transplanted or moved to assist recovery efforts (Lusardi et al. 2015). The effort resulted in an apparent restoration of populations in about 51 km of stream plus introduction into three headwater lakes by 1998. Subsequent genetic studies suggest that coastal rainbow trout influence has been greatly reduced within the basin. Stephens (2007) found that 85% of the population showed less than 5% introgression with rainbow trout, suggesting beneficial results. However, data also indicate that Little Kern golden trout exhibit signs of significant genetic structuring and reduced genetic diversity associated with drift and inbreeding (Stephens 2007, Lusardi et al. 2015). Recent genetic studies have identified low (<2%) introgression levels in five subpopulations within the Little Kern basin including: upper North Fork Clicks Creek, Upper Clicks Creek, Trout Meadow Creek, Little Kern River above Broder's Cabin, and Little Kern River above Wet Meadows Creek (Stephens and May 2010).

Trends in Abundance: When Little Kern golden trout were at their minimum range (16 km of stream), their population was estimated at 4,500 fish (Christenson 1978). If it is assumed they currently persist in 50 km of small streams, with 300 fish age 1+ and older per km (500/mi; Christensen 1994), the total numbers are probably around 15,000 fish. However, densities at the subpopulation-level likely vary across the watershed and, consequently, their overall abundance may be less than these rough estimates. Stephens and May (2010) found five subpopulations showing less than 2% introgression and six others showed either low or high rainbow trout introgression estimates, depending on the type of genetic marker used (USFWS 2011). If only unhybridized fish are counted, then the number may be confined to 20 km or so of refuge streams, supporting perhaps 5,000-6,000 fish. The estimated number of spawning Little Kern Golden trout within each refuge subpopulation is unknown; spawner numbers may be small and limit long term persistence of some populations and/or negatively affect their genetic integrity.

Factors Affecting Status: Little Kern golden trout are largely confined to the headwaters of the Little Kern River in small, isolated, tributary streams. The streams in which they occur are on public lands administered by the Sequoia National Forest or Sequoia National Park. These disconnected subpopulations face genetic risks associated with lack of gene flow and small population sizes. Their isolation from one another likely promotes inbreeding, genetic drift, and further reductions in genetic variability. These factors may be contributing to limited effective population sizes and reductions in individual fitness.

Alien species. A principal threat is loss of genetic diversity, due to founder effects associated with reintroduction programs between 1974 and 1996. Little Kern golden trout remain threatened by hybridization with hatchery rainbow trout. Stephens and May (2010) found that several populations of Little Kern golden trout exhibited high levels of rainbow trout introgression ranging from 0.25-0.83 (% rainbow trout admixture), depending on the type of marker used. Thus, loss of genetic diversity continues to threaten their long-term persistence. Habitat loss from the region's long history of grazing, logging, and roads, as well as stochastic events such as floods, drought, and fire may increase local population extinction risks, especially considering current genetic status (Moyle 2002).

Fortunately, brown trout appear to have been removed from the basin and volitional movement upstream from the Kern River is limited by a series of large cascades in the lower portion of the river that serve as an effective fish barrier (C. McGuire, CDFW, pers. comm. 2016). Hatchery trout are no longer stocked in the basin (S. Stephens, CDFW, pers. comm.

2008). For a full discussion of broader threats and challenges facing fishes in the Kern basin, see the California golden trout account.

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Little Kern golden trout. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	n/a	All major dams outside native range of Little Kern golden trout.
Agriculture	n/a	
Grazing	Low	Ongoing threat to habitat but greatly reduced from the past.
Rural /residential development	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Low	Trails and off-road vehicle routes can cause sediment and pollution input into streams; however, most areas occupied are within designated wilderness.
Logging	Low	This is an important land use in the broader region but probably has no direct effect on Little Kern golden trout streams.
Fire	Medium	Can change watershed processes, cause siltation, and loss of habitat. Extensive fire could cause population vulnerability through reduction in habitat or mortality. Recent fires (e.g., Lion Fire) have directly impacted core conservation streams.
Estuary alteration	n/a	
Recreation	Low	Entire range is within Sequoia-Kings Canyon National Park and Sequoia National Forest.
Harvest	Low	Light fishing pressure; most fishing is catch and release.
Hatcheries	Low	Residual effects of hybridization with hatchery fish. Hatchery fish are no longer planted in the basin.
Alien species	High	Hybridized populations or remnant rainbow trout populations continue to threaten genetic integrity.

Effects of Climate Change: Climate change has altered Sierra precipitation and runoff patterns, with more precipitation falling as rain and runoff occurring earlier than historical averages. The southern Sierra Nevada, however, is the highest part of the mountain range which may at least partially offset substantial reductions in snowpack, as is predicted in the northern Sierra Nevada and other regions of the state. The autumn base flow period may be a particularly stressful period for Little Kern golden trout because the Kern basin experiences reductions in groundwater recharge and extent of wetted habitat, along with warmer water temperatures. During drier years, low flow conditions may periodically lead to further disconnection of subpopulations. This may be particularly true in smaller headwater streams that support the most genetically intact populations and provide key over-summering habitat. Elimination of grazing and other activities that compact meadows (reducing their ability to store water) and reduce riparian cover and shade can mitigate, in part, for the effects of climate change. An increase in fire frequency or intensity could remove riparian vegetation and promote sedimentation. Moyle et al. (2013) rated Little Kern golden trout as critically vulnerable to climate change, with extinction likely in California in the next 100 years if present climate change trends continue.

Status Score = 2.0 out of 5.0. High Concern. The Little Kern golden trout has high a probability of disappearing in the next 50-100 years, despite major efforts to protect this unique subspecies. This vulnerability has long been recognized and serious management efforts to protect it began in 1975. The Little Kern River was included within the Golden Trout Wilderness Area in 1977. The subspecies was listed as threatened by USFWS in 1978 and a management plan was completed by CDFW in the same year (Christenson 1978) and revised in 1984. The Little Kern golden trout's listing status was reaffirmed in 2011 (USFWS 2011), though the nature of threats has changed. Recent genetic work suggests that, while hybridization with rainbow trout remains a concern, the threat has greatly been reduced through management actions and most populations now exhibit less than 5% introgression with rainbow trout.

CDFW, beginning in 2012, has performed the most comprehensive basin-wide population and habitat assessment in the Little Kern River drainage to date, including documentation, evaluation, and geo-referencing of all artificial and natural barriers. Tissues have been collected throughout the drainage and will be used in future genetic studies. However, Little Kern golden trout populations remain small and highly structured, with reduced heterozygosity due to inbreeding and genetic drift. Given recent advances in our understanding of the genetic status of the subspecies (Stephens 2007, Stephens and May 2010, Lusardi et al. 2015), the Little Kern golden trout management plan should be revised and updated.

Table 2. Metrics for determining the status of Little Kern golden trout, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	1	Unhybridized populations occur in just 5 or 6 stream segments. With the exception of Coyote Creek (adjacent drainage), there are no other populations outside the Little Kern drainage.
Estimated adult abundance	2	Probably less than 5,000 adults in 5 isolated populations.
Intervention dependence	3	Requires intervention to maintain unhybridized populations, prevent invasions of alien trout, and promote population connectivity.
Tolerance	3	Presumably fairly tolerant, as are most Rainbow trout, but not tested.

Genetic risk	1	Significant as populations are fragmented. Ongoing threats from hybridization and loss of genetic diversity through inbreeding and genetic drift.
Climate change	1	Critically vulnerable to range reductions and habitat alteration from climate change.
Anthropogenic Threats	3	1 High and 1 Medium threat.
Average	2.0	14/7.
Certainty (1-4)	4	Recent publications and USFWS 5-year review.

Management Recommendations: A detailed review of recommended management actions can be found in Lusardi et al. (2015). Some of the key management actions to improve the status of Little Kern golden trout include:

Conduct further targeted genetic monitoring. A genetic baseline of all presumed Little Kern golden trout within the Little Kern drainage must be established. While genetic monitoring has been conducted over the last several years, additional monitoring is necessary and should focus on identifying all unhybridized and remaining introgressed populations. Based on the outcomes of future analyses, draft a new genetic management plan specific to the Little Kern golden trout.

Establish out-of-basin refuge populations. Little Kern golden trout and their habitats are vulnerable to stochastic events such as fire, disease, climate change and flooding. While a single out-of-basin population exists adjacent to the Little Kern basin (Coyote Creek), additional refuge populations should be established. Such populations could also be used as sources for future introductions and provide essential population redundancy, serving as an 'insurance policy' against potential population losses within the native range.

Use barriers strategically. Based on genetic monitoring, continue to use structural barriers where necessary to prevent colonization and reproduction between Little Kern golden trout and introgressed individuals. It should be noted, however, that barriers can impair connectivity and gene flow between populations, potentially reducing genetic diversity.

Evaluate existing locations of known barriers and, where possible, remove or alter redundant barriers to improve gene flow between populations.

Eliminate grazing or use riparian fencing. In key areas such as the Jordan grazing allotment and low gradient meadows such as Lion, Grey, and Loggy meadows, grazing should be eliminated or restricted significantly to improve meadow habitat condition and habitat resilience to climate change.

McCLOUD RIVER REDBAND TROUT
Oncorhynchus mykiss stonei (Jordan)

Critical Concern. Status Score = 1.4 out of 5.0. McCloud River redband trout populations are very small, fragmented, and exist in limited habitats so status could change rapidly, particularly related to predicted climate change impacts.

Description: The following description is based on the Sheepheaven Creek population (Hoopaugh 1974, Gold 1977), which was believed to have a somewhat narrower range of meristic characters than the other known populations found in Edson and upper Moosehead creeks. The population found in Swamp Creek was started from Sheepheaven fish in 1973-74, so comparisons have been removed. More recent work (Behnke 1992) considered this population to best represent the subspecies because it is unlikely to have had any history of hybridization with introduced rainbow trout. In 2002, Behnke noted that Sheepheaven Creek were distinct from other Upper Sacramento River redbands, perhaps due to their isolation in this secluded, volcanic spring creek less than 2 km long. Overall body shape is similar to the "typical" trout as exemplified by rainbow trout. It has a yellowish to orange body color with a brick-red lateral stripe. The dorsal, anal, and pelvic fins are white tipped. Adults retain parr marks. Gill rakers number from 14-18 (average 16), which is the lowest number known from any rainbow trout population (Behnke 1992). Pyloric caeca number is 29-42, which is also low. However, the numbers of scales along the lateral line (153-174) and above the lateral line (33-40) are greater than in most rainbow trout. Pelvic fin rays are 9-10 and branchiostegal rays range from 8-11. Many, but not all, McCloud River redband Trout have basibranchial teeth and an orange slash along the throat, characteristics more typically associated with cutthroat trout, but are most similar morphologically to golden trout (Behnke 1992, 2002, Simmons et al. 2009).

Taxonomic Relationships: Distinct "redband trout" from the lower McCloud River were first recognized in 1885 by Deputy U.S. Fish Commissioner, Livingston Stone, who was responsible for a fish hatchery located on the river. However, the lower portion of the McCloud River (below Middle Falls, historical barrier to anadromy) was historically inhabited by coastal rainbow trout, including steelhead, and other fishes. It is uncertain whether redbands were distributed in these lower reaches and, if so, whether Stone identified them as distinct. The redband trout we recognize today are varieties of inland resident rainbow trout that became isolated in headwater systems thousands of years ago. The taxonomic status of California populations of redband trout has been under much debate, reflecting the diversity of forms that are called 'redband' trout and the long isolation of many populations (Legendre et al. 1972, Miller 1972, Behnke 1992, 2002). A complicating factor is that many populations have hybridized with the closely related coastal rainbow trout, which have been widely planted in historical redband trout streams. Behnke (1992, 2002) considers redband trout in the western U.S. to consist of a number of distinct lineages, each independently derived from early invasions of ancestral forms of trout into headwater systems, with populations then becoming isolated through geologic events. Behnke (2002) indicated that McCloud River redband trout are part of a Northern Sacramento River basin trout complex in which all populations are, or were, tied to the headwaters of the Sacramento, McCloud, Pit, and Feather rivers. In theory, the subspecies name *O. m. stonei* could be applied to any population in these headwaters, but only a few streams in the upper McCloud River watershed are home to populations of relatively unhybridized and non-introgressed redbands; these fish should be the exclusive possessors of the subspecies epithet (Simmons et al. 2009).

The population in Sheepheaven Creek, described above, was believed to be distinctive from other McCloud River redband trout, so Behnke suggested it should be classified as a separate subspecies. Genetic studies by Berg (1987), using electrophoretic techniques, by Nielsen et al. (1999) using microsatellites, and more recently by Stephens (2007) using nuclear DNA methods, support the conclusion that the Sheepheaven Creek form is distinct. However, the most recent studies (Simmons et al. 2009, Simmons et al. 2011), which used both nuclear and mitochondrial single nucleotide polymorphisms, indicate that Sheepheaven Creek fish and fish from three other streams (Edson, Swamp, and Moosehead) should be considered together as the McCloud River redband trout group. Of the tributaries to the Upper McCloud River, these creeks were found to contain relatively "pure" populations, with few introgressed alleles (<5%) from coastal rainbow trout (Simmons et al. 2011). According to the authors, these populations, despite some introgression, should be managed as a group to maintain genetic integrity and gene flow in the remaining extant variations of the McCloud redband (Simmons et al. 2010, Stephens et al. 2011). Besides Moosehead Creek, most of the southern tributaries to the McCloud River contain redband populations with higher levels of introgression with coastal rainbow trout. Trout Creek (northern tributary) in the Upper McCloud River was chemically treated in 1977 and restocked with Sheepheaven Creek fish. Although Trout Creek was started with Sheepheaven fish, recent genetic testing (Simmons et al. 2009, Stephens et al. 2013) have shown mixed results on the genetic purity of this stream. At this time, more genetic work is needed to determine Trout Creek's population status (M. Dege, CDFW, pers. comm. 2016).

Life History: Available information suggests that the life history of McCloud River redband trout is similar to that of other *O. mykiss* populations, including golden trout, in small streams. redband trout caught from Sheepheaven Creek were in reproductive condition in June, indicating that they spawn in late spring (May-June), as do other rainbow trout at high elevations. The largest fish recorded during a 1973 survey (Hoopaugh 1974) was 208 mm FL, and the population was then estimated at 250 fish over 80 mm FL. Four size classes were found in the stream. Observations in August 2008, suggest the same age classes were still present (J. Katz, R. Quinones, and P. Moyle, unpublished observations). However, CDFW surveys of Sheepheaven Creek in 2011 indicated a lack of younger age classes, extremely low abundance, and limited distribution within suitable habitat that declined over time through drought (2012-2016, J. Weaver, CDFW, pers. comm. 2012, M. Dege, CDFW, pers. comm. 2016). Over the course of the drought, 1,597 trout were rescued from Sheepheaven, Swamp, Edson, and Moosehead creeks, which represents a substantial portion of the current extant population. McCloud River redbands can show territoriality, and were observed cannibalizing their own young during rescue operations in 2013-2015 (M. Dege, CDFW, pers. comm. 2016). Perhaps this territoriality contributes to their relatively low abundance in the available habitats that remain accessible to them in the wild.

Habitat Requirements: Habitat requirements for the McCloud River redband are derived from conditions in Sheepheaven Creek (Hoopaugh 1974, Moyle 2002) and the McCloud River based on descriptions in the 2016 Redband Trout Conservation Agreement (RTCA), which summarizes information from unpublished habitat surveys. This document is currently undergoing CDFW and U.S. Fish & Wildlife Service review and will be finalized soon (M. Dege, CDFW, pers. comm. 2016). Sheepheaven Creek is a small, spring-fed stream at an elevation of 1,433 m. Water temperature in summer typically reaches 10-13°C and the flow drops below 0.03 $m^3 sec^{-1}$ (1 cfs). The stream flows for about 2 km from the source and then disappears into the porous bedrock. During periods of drought, flows are greatly reduced and streams in the upper McCloud basin become intermittent; as a consequence, summer water temperatures can exceed 22°C, but most streams emerge from springs at around 5-7°C and have maintained temperatures in downstream reaches of about 9-15°C even through historically low precipitation (M. Dege, CDFW, pers. comm. 2016). The portion of the upper McCloud River historically inhabited by redband trout usually flows at 1.2 $m^3 sec^{-1}$ (40 cfs) through a steep canyon. It is extremely clear and cold (<15°C) but becomes very low or intermittent in times of drought.

The present day streams inhabited by putative redband trout are generally small and dominated by riffles and runs with small pools. Pools appear to be preferred habitat for larger fish, especially if they contain dense cover from fallen trees. Spawning substrates are gravel riffles, as described for other small trout (Moyle 2002). Spawning temperatures are usually 5-10°C. Fry rear in shallow water on stream edges for the first weeks after emergence.

Distribution: McCloud River redband trout are confined to small tributaries of the upper McCloud River (Table 1). All watersheds are wholly or partially located on the Shasta-Trinity National Forest and privately held lands in Shasta and Siskiyou Counties (M. Dege, CDFW, pers. comm. 2016, CDFW 2015). Historically, they were apparently present in the mainstem McCloud River above Middle Falls and perhaps in the lower river and its tributaries as well, especially in reaches not accessible to anadromous steelhead. Redband trout from an unnamed stream located between Sheepheaven Springs and McKay Creek (now known as Sheepheaven Creek, USGS 2012) were transplanted into Swamp Creek in 1973 and 1974 and into Trout Creek in 1977 (RTCA 2016). They are now established in both streams. According to a 2011 CDFW survey, putative redband trout exist in streams with a total length of about 8.9 km, with a total estimated population of 3,560 fish (Weaver and Mehalick 2011). Potential McCloud redband habitat, including the upper McCloud River, is about 98 km (about 50 km in dry years) (RTCA 2016). More recent data suggest that two additional streams that hold presumed non-introgressed McCloud redbands may expand this distribution if confirmed by genetic analysis (M. Dege, CDFW, pers. comm. 2016). Most of these tributary streams remain isolated from the upper McCloud River due to subsurface flows and the highly porous volcanic rock in the area, and may only experience limited connectivity with the McCloud River during extreme high flow events. The exception is Moosehead Creek, the only southern tributary of the group, which can have subsurface flows during drier periods, but also has an artificial fish barrier 2.2 km from the confluence with the McCloud River to prevent upstream migration of non-native or hybridized trout.

Table 1. Redband trout streams in the upper McCloud River. Summer flow class: 1 = < 1 cfs, 2 = 1-5 cfs, and 3 = > 5 cfs in late summer in most years. Redband status: 0 = all trout hybridized, 1 = 'pure' population, 2 = relatively 'pure' population with little introgression, 3 = good redband population but slightly higher levels of hybridization. Isolation: 1 = no barriers to non-native trout, 2 = connections present in wet years in lower reaches, 3 = no passable connections with other streams.

Stream	Summer flow class	Redband status	Isolation	Comments
Sheepheaven (McKay)	1	1	3	Key "pure" population.
Trout	2	2	3	Introduced from Sheepheaven Creek in 1977.
Swamp	1	1	3	Introduced from Sheepheaven Creek in 1973-74.
Edson	1	1	3	
Tate	2	3	1	
Moosehead (upper)	1	1	2	
Raccoon	1	3	2	
Blue Heron	1	3	2	Possibly extirpated.
Bull	1	2	2	
Dry	1	2	2	
Upper McCloud	3	0	1	Dominated by hybridized and non-native trout.

Trends in Abundance: McCloud River redbands presumably once had interconnected populations in the Upper McCloud River and tributaries, so the present isolated populations represent greatly reduced remnants of historical populations. Recent genetic analyses indicate that all populations sampled from across the upper McCloud watershed shared alleles in common with the distinctive Sheepheaven Creek population, indicating that redband trout with common ancestry were once widely distributed throughout the basin (Simmons et al. 2009, 2011). Existing redband trout streams were surveyed a number of times from 1975-1992 and in 2011 (Figure 5 in RTCA 2016; Weaver and Mehalick 2011). Numbers of fish estimated were highly variable and depended on the stream and habitat sampled; numbers ranged from 53 to 1,100 per km. Repeated drought cycles (e.g., 1976-1977, 1987-1992, 2012-2016), combined with the predominance of loamy volcanic soils in the watershed, have intermittently reduced surface flows in most McCloud basin streams and limited populations of McCloud redband trout. The same is expected under future drought conditions and will likely be exacerbated by the effects of climate change. If population estimates are confined to the relatively non-introgressed populations in Sheepheaven, Edson, upper Moosehead and Swamp creeks, then abundance is estimated at 3,500 putative McCloud redband trout in late summer, with large error bars around the estimate (Weaver and Mehalick 2011).

Habitat conditions and consequently abundance of McCloud River redband trout waned in 2012-2015 due to extremely dry conditions (M. Dege, CDFW, pers. comm. 2016). During drought, available wetted habitat was reduced significantly across all streams in the region, including spring-fed streams such as Sheepheaven Creek. The population has likely dwindled from the 2011 CDFW estimate, when less than 100 fish were captured in electrofishing surveys in Sheepheaven Creek (CDFW unpubl. data). It is likely that each of the four habitats for non-introgressed McCloud redbands may only support stream populations of just fewer than 100 to 500 individuals during extreme dry years. The mechanics of how drought effects each stream are not well understood at this time. Dual drought within a single year, as occurred in 2013-14, saw a hard freeze on top of low flows and frozen stream sections throughout the McCloud redband range and then drying of streams the following summer and fall (M. Dege, CDFW, pers. comm. 2016).

An increase would be expected due to the many ongoing habitat restoration and protection efforts that have taken place, post-drought. Presumably, habitat protection and restoration, including protection of springs, has moderated population fluctuations and reduced vulnerability to drought, but these impacts have not been tested. In future years, the population would likely rebound to exploit higher flows and access to greater wetted habitat availability, but cumulative impacts associated with climate change and redband-rainbow hybrids could limit the ability of the species to bounce back. It will take considerable effort to maintain McCloud redband trout populations, especially through extended droughts.

Factors Affecting Status: Long-term survival of populations of McCloud River redband trout confined to small, isolated, streams such as Sheepheaven Creek is tenuous because stream habitats are largely diminished during drought years, a process which can be accelerated by drought, climate change, and poor watershed management practices impacting upland and riparian areas (Table 2). Fortunately, interest in conservation and management of McCloud River redbands has resulted in recent efforts to relocate and rescue individuals in disconnected habitats. Factors which threaten McCloud River redband trout populations are: (1) alien species, especially coastal rainbow-redband trout hybrids, (2) long-term drought, (3) fire, (4) grazing, (5) roads, (6) logging, and (7) climate change. Upper McCloud streams can be regarded as exceptionally vulnerable to these factors due to their geologic and hydrologic nature (isolation, low flow in summer months, etc.). CDFW (2008) found that many road crossings and culverts showed signs of scour and could potentially be barriers to upstream fish migration; the report called for a comprehensive study of these potential barriers and habitat degradation due to logging and grazing practices.

Alien species. Coastal rainbow trout (*O. mykiss*), brown trout (*Salmo trutta*), and brook trout (*Salvelinus fontinalis*) have been repeatedly introduced into the upper McCloud watershed and have established self-sustaining populations. In particular, the McCloud River has received substantial numbers of stocked hatchery rainbow trout in the past to support a "put-and-take" fishery, although stocking of coastal rainbow trout in the upper McCloud River was discontinued in 1994 (RTCA 2016). Generally, where alien trout are present, redband trout are absent or have become hybridized. The exact causes of redband trout disappearance from the McCloud River itself have not been documented, but presumably it was a combination of predation on young (brown trout), competition for space (all species), disease introductions (all species), and hybridization and introgression (coastal rainbow trout). A number of redband trout streams are thought to be too small or isolated to be subject to introductions, although some (e.g. Trout Creek) were nevertheless invaded at one time or another by unknown means.

Hybridization between coastal rainbow trout and redband trout is a natural event: both are native to California and hybridization would have occurred where their populations overlapped (e.g. lower McCloud River and tributaries). However, due to planting of rainbows above natural barriers, hybridization has become a primary threat to headwater redband population persistence in the basin. Once hybridization occurs, the rainbow trout phenotype tends to dominate, resulting in a loss of the distinctive, brightly-colored redband trout form. This is likely coupled with a loss of adaptability to the distinctive streams where redband trout evolved. Rainbow trout and rainbow-redband hybrids have replaced McCloud River redbands in the majority of their historical range.

Fire. The 2016 RTCA considered fire a potential threat to this subspecies because fire suppression has greatly increased the amount of fuels in surrounding forests and increased the potential for high intensity fires. Such fires can cause direct mortality to fishes (high water temperatures), as well as indirect impacts from increased sedimentation and reduction in riparian vegetation and associated instream shading.

Grazing. Grazing by cattle and sheep has taken place in the McCloud River watershed for over 125 years and was especially intense in the first half of the 20th century. Heavy grazing, especially by cattle, reduced trout habitat by eliminating streamside vegetation, collapsing banks, making streams wider and shallower, increasing temperatures, reducing bank undercutting, polluting the water with feces and urine, silting up spawning beds, and generally making the habitat less complex and suitable for trout. The reduction of grazing pressure in the late 20th century and the increasing willingness of land managers to implement improved grazing practices and to use exclusion fencing along streambanks has led to better condition of small streams in the McCloud River watershed and improved habitat for redband trout. Today, much of Sheepheaven and lower Trout creeks have been fenced to exclude cattle. The grazing allotment associated with Sheepheaven Creek has not been active for several years, but this could change in the future.

Logging. The region in which McCloud River redband trout live contains a checkerboard of private and public ownership, with most public lands as part of the Shasta-Trinity National Forest. According to the RTCA (2016) "Potential impacts to McCloud redband and their habitat from past logging practices include loss of shade canopy, increased water temperatures, increased sedimentation, reduced recruitment of large woody debris, loss of fish habitat diversity, and increased peak storm flows".

These impacts continue into the present day through continued logging, including culverts potentially blocking or limiting instream movement, removal of water for dust control on dirt roads, erosion of sediment from roads, and similar factors. Fortunately, greatly improved logging practices have reduced the effects of logging and logging roads on streams, in good part because both private and public land managers recognize the uniqueness of the McCloud River redband trout and their habitats (RTCA 2016).

Harvest. It is likely that harvest was never a major problem in the small streams of the McCloud basin but redband trout populations are small enough that even occasional harvest by anglers or scientific collectors could reduce populations (RTCA 2016). Special angling regulations are in place for the following streams: Sheepheaven, Edson and Moosehead creeks (closed to all fishing all year); Swamp Creek (last Saturday in April through November 15 – zero limit, artificial lures with barbless hooks only) (CDFW 2015-2016).

Transportation. Roads, mainly from logging, are numerous and widespread throughout the upper McCloud River basin, providing a source of sediment input into streams, potentially covering spawning gravels. They also provide easy access to most redband streams in the watershed. Recently, all major timber companies in the region that own private land adjacent to streams and the US Forest Service have decommissioned roads to reduce these impacts (M. Dege, CDFW, pers. comm. 2016).

Table 2. Major anthropogenic factors limiting, or potentially limiting, viability of populations of McCloud River redband trout. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	Low	Major dams are downstream of remaining McCloud River redband habitat but their construction may have contributed to fragmentation of habitat in the past.
Agriculture	n/a	
Grazing	Medium	Historically pervasive in the area but currently limited on private and U.S. Forest Service lands through attrition and better grazing management.
Rural /residential development	n/a	
Urbanization	n/a	
Instream mining	n/a	
Mining	n/a	
Transportation	Medium	Roads are widespread in the upper McCloud basin and are sources of sediment input into streams; culverts etc. can prevent movement.
Logging	Medium	Logging is the major land use in the region; associated water drafting may reduce stream flows and cause direct or indirect mortality.
Fire	Medium	Headwater areas could be altered by more severe fires than occurred historically.
Estuary alteration	n/a	
Recreation	Low	Off-road vehicles a potential threat.
Harvest	Low	Light angling pressure in open streams; special fishing regulations to protect key redband populations.
Hatcheries	Low	Hatcheries are used to hold fish rescued from drought conditions; potential genetic effects if hatchery breeding program instituted.
Alien species	High	Major potential threat of competition, hybridization and introgression and cause of limited distribution.

Effects of Climate Change: The fact that existing redband trout streams are so small and flow through highly permeable volcanic soils means that they are exceptionally vulnerable to stressors such as floods, drought, and fire, which are likely exacerbated and compounded in complex ways under climate change scenarios. However, the persistence of distinctive trout in Sheepheaven Creek is due to the springs that maintain some level of surface flow at cool temperatures (albeit for a short distance), even during severe drought. Most of the other streams occupied by McCloud River redbands have similar 'safe' water sources. It is also worth noting that spring flows can be eliminated by even minor volcanic or seismic activity and these streams are located in a relatively geologically active region. Additionally, most streams currently inhabited by redbands are already subject to seasonal reductions in flow (during non-drought periods), so increases in air temperature or reductions in snow pack during prolonged drought may dramatically reduce available wetted habitat, as was seen in during the 2012-2016 drought.

Moyle et al. (2013) consider McCloud River redband trout to be "critically vulnerable" to climate change because of the small size of their streams, warmer temperatures, and the potential effects of lengthy droughts. While there is no evidence of long-term changes in stream temperatures in Sheepheaven Creek or the other three critical streams in the region, the ongoing drought has severely reduced available wetted habitat and reduced connectivity between relatively non-introgressed populations (Simmons et al. 2009, M. Dege, CDFW, pers. comm. 2016). The McCloud River redbands are a unique and robust species that have persisted through historic drought before, but may be facing additional challenges with the likely impacts of climate change that will create more and longer lasting unfavorable conditions for the species (M. Dege, CDFW, pers. comm. 2016).

Status Score = 1.4 out of 5.0. Critical Concern. Long-term drought, fire, or other factors that affect stream flows or habitat suitability, coupled with genetic risks associated with isolation of small populations, threaten McCloud River redbands with possible extinction. Populations are especially vulnerable to rapid changes in status due to their small, isolated populations. While high levels of interest and management scrutiny seem to preclude *immediate* risk of extinction, recent events such as rescue efforts and movement of vulnerable populations into suitable habitat and artificial holding tanks in Mount Shasta Hatchery is of serious concern. Rescue operations by CDFW in 2013-15 greatly reduced the drought mortality of the species.

While rescue operations were carefully deemed warranted, the quality of habitat for McCloud River redbands has not seemed to change very much during historic drought; rather, it appears as if only the total amount of habitat has waned. Should average or above average water years return in the future, it seems likely that the species will enjoy a resurgence (M. Dege, CDFW, pers. comm. 2016). In longer time frames, extinction probability will increase as the climate becomes warmer and droughts more frequent and prolonged. Genetic risks increase with habitat reductions, potentially leading to bottlenecks in small, isolated populations.

The McCloud River redband trout is considered to be vulnerable by American Fisheries Society (Jelks et al. 2008) because of its limited distribution and exposure to multiple threats. It was considered to be a Candidate Species for listing by the USFWS in 1994 but, following the signing of the 1998 RTCA by the USFS and other cooperators, it was removed from consideration. However, the conservation agreement does not actually preclude listing under the Endangered Species Act if needed (M. Dege, CDFW, pers. comm. 2016). The USDA Forest Service lists it as a Sensitive Species, while NatureServe considers it to be an imperiled subspecies. CDFW considers McCloud River redband trout a fish species of special concern (Moyle et al. 2015).

Table 3. Metrics for determining the status of the McCloud River redband trout, where 1 is a major negative factor contributing to status, 5 is a factor with no or positive effects on status, and 2-4 are intermediate values. See methods section for further explanation.

Metric	Score	Justification
Area occupied	1	Four 'core' populations clustered fairly close to each other and all are in Upper McCloud watershed, so are treated as one 'watershed.'
Estimated adult abundance	1	Population prior to the 2012-2016 drought was likely somewhat less than 3,000 fish over 80 mm FL, with each stream having 100-1,000 fish. In drought years, total numbers of fish over 80mm FL was likely less than 1,250 fish.
Intervention dependence	2	Drought necessitated rescue of several populations and relocation to holding facilities until natural conditions improved; ongoing implementation and recent revision and expansion of a Conservation Strategy is critical for survival.
Tolerance	3	It is likely they are fairly tolerant of high temperatures, as are other redband trout, but water quality in their small streams has to be monitored during drought years.
Genetic risk	1	Hybridization risk with rainbow trout is high; small isolated populations result in genetic bottlenecks and inbreeding depression.
Climate change	1	Vulnerability is high in all streams because of small size and cumulative effects of a changing climate and drought.
Anthropogenic threats	2	1 High and 4 Medium threats.
Average	1.4	10/7.
Certainty (1-4)	4	Most published information is on Sheepheaven Creek population, though recently more studies have come from Edson, Moosehead, and Swamp creeks habitats.

Management Recommendations: Conservation of McCloud River redband trout is active and ongoing, thanks to the leadership of the McCloud Redband Core Group (RCG), a multi-partner organization (California Department of Fish and Wildlife, Shasta-Trinity National Forest, U.S. Fish and Wildlife Service, private landowners, and others), which is dedicated to the conservation of the McCloud River redband trout. In addition, the western states, several tribes, and Trout Unlimited have been coordinating all recovery efforts under a formal Conservation Agreement, with regular meetings and information updates (Inland Redband Conservation Agreement 2014).

The California Department of Fish and Wildlife has undertaken significant genetic studies, fish rescue operations, and creation and implementation of conservation hatchery plans at the Mount Shasta Hatchery to protect McCloud River redband trout from severe impacts to extinction in the wild. In the past, most management attention focused on the Sheepheaven Creek population because of its historical reputation on being so distinctive. Recent attention has focused on the broader populations within the upper basin and four 'core conservation populations' (Sheepheaven, Edson, Swamp, and Moosehead) have been identified and will be managed collectively (J. Weaver, CDFW, pers. comm. 2012). Private and public landowners actively cooperate on conservation, particularly those who comprise the RCG. On private lands, considerable effort has been made to improve roads in ways that minimize impacts to streams, to fence streams from livestock, and to assist in restoration and management activities. The conservation agreement is an effort to provide a systematic framework for all restoration and management activities in the watershed. It is crucial that this agreement be finalized as the working plan to improve conditions for McCloud River redband trout. The following recommended actions to increase protection for redband trout and their habitats are largely drawn from this agreement. Recommendations are not in order of importance.

Establish a McCloud River Redband Refuge. A portion of the upper McCloud River basin should be managed for the protection and enhancement of McCloud River redband populations and their habitats. The refuge should include the main stem McCloud River and its tributaries above the Middle Falls and, within this broader refuge, a 'core conservation area' should be established to provide further protections for populations with low (or no) levels of introgression with coastal rainbow trout (Sheepheaven, Swamp, Edson, and Moosehead creeks, Interior Redband Trout Conservation Agreement 2016). While the refuge area contains all the streams known to contain presumed redband trout at the present time, suitable reaches of other perennial streams should, nevertheless, be evaluated for their potential as future translocation/restoration sites. Streams that have potential for expanding the range of redband trout (particularly within-basin, but also outside of the McCloud basin as warranted) would be of great value in terms of offsetting climate change impacts or stochastic events that may lead to the extirpation of one or more existing populations. Management plans that include eradication of non-native trout should be developed and construction of barriers to prevent alien trout invasions considered. In particular, the upper McCloud River itself should be evaluated as a refuge during periods of reduced stream flow caused by prolonged drought or climate change.

Maintain and enhance existing habitats. McCloud River redband trout survive in remarkably small and fragile habitats, so continued work is needed to improve the ability of these habitats to support redband trout and to reduce the impacts of human activities. Of particular concern are grazing and logging practices, but other factors such as fire protection, angling, and off-road vehicles have also been taken into consideration. While management plans and agreements are in place to protect streams, continued vigilance is required to avoid long-term loss of habitat. Creation or enhancement of pool refuge and barrier removal in these habitats should be prioritized to reduce genetic bottlenecks and mitigate the effects of drought on remaining populations.

Protect genetic integrity of existing populations. The present populations of McCloud River redband trout are highly vulnerable to loss of genetic integrity (and phenotypic distinctiveness) due to hybridization with introduced rainbow trout, introgression, and potential for genetic bottlenecking due to isolation of existing redband populations from one another. Efforts are needed, therefore, to protect populations from further inappropriate introductions (e.g., by making vehicle access difficult) or from 'natural' invasions from downstream areas (e.g., through construction of barriers). This program should include genetic and phenotypic monitoring as part of the assessment of population health. Consideration should also be given to active movement of putative redbands in order to promote and restore gene flow and increase genetic heterozygosity, in order to offset potential impacts from past and ongoing isolation of existing populations (e.g., mixing genetically distinct McCloud River redband streams, Upper McCloud River Redband Trout Reintroduction Plan 2013).

Continue to develop and enforce angling regulations appropriate for protection of redband trout. Sheepheaven, Edson, and Moosehead creeks are closed to all fishing all year. Catch-and-release angling is allowed in Swamp Creek from the last Saturday in April to

November 15th, using artificial lures with barbless hooks. These regulations need to be strictly enforced with frequent monitoring of streams to minimize impacts on the fragile remaining populations at this time.

Develop and implement a genetic management plan for all populations. Initial steps in genetically managing McCloud River redbands have been undertaken by CDFW. Genetic management elements have been implemented for rescued McCloud River redbands during hatchery operations and reintroduction, and a formal plan from CDFW is forthcoming (M. Dege, CDFW, pers. comm. 2016). Genetic management and conservation elements include enhancing local genetic diversity by crossing rescued fish between populations and minimizing the influence of captivity on rescued fish. To date, over 1,700 offspring have been produced from parents rescued from Edson, Moosehead, and Swamp creeks based on a genetic matrix designed to increase diversity among the population as a whole and to maintain the genetic integrity of captive populations (M. Dege, CDFW, pers. comm.). Reintroduction efforts occurred during the fall of 2016 (see below). (M. Dege, CDFW, pers. comm. 2016). Moving forward, out-of-basin translocations of these offspring should be considered to create a refuge population that can mitigate against potential devastating drought, fire, seismic activity, or other catastrophic events.

Establish a regular habitat and population monitoring program. This should be established for all putative redband trout populations and monitoring should occur at least once every 4-5 years (one redband generation).

Develop a formal re-introduction plan and perhaps a captive broodstock plan. Nearly 600 rescued McCloud River redbands of all size and age classes are being held separately according to their resident stream (Edson, Moosehead, and Swamp creeks,) in nine holding tanks in the Mount Shasta Hatchery (CDFW 2015, M. Dege, CDFW, pers. comm. 2016). After a wet season of above average precipitation in the Upper McCloud River basin and meeting reintroduction and flow criteria, the fish will be re-introduced and/or outplanted into genetically distinct McCloud River redband streams (M. Dege, CDFW, pers. comm.).

After a full year of average flows, the surviving F1a offspring and their rescued parents were released in September 2016 to their native streams, with a mixing of 10% of fish to different streams to mimic natural straying and recolonization rates (M. Dege, CDFW, pers. comm. 2016). In total, 349 rescued fish and 930 F1a offspring were reintroduced back into Edson, Moosehead, Swamp, and Sheepheaven Creeks based on the management plan. Should future drought and below-average precipitation return to the McCloud basin, this approach could be followed to allow the small population of McCloud River redbands to persist. A captive broodstock program may be developed based on the work of Simmons et al. 2011 and colleagues with captive fish that were spawned in the hatchery in 2015 and 2016.

PAIUTE CUTTHROAT TROUT
Oncorhynchus clarkii seleniris

High Concern. Status Score = 2.1 out of 5.0. PCT have a high likelihood of extirpation in their native range within the next 50 years without continued commitment to intense monitoring and management. All populations are small and isolated, and therefore highly susceptible to illegal introductions of alien trout and local stochastic events.

Description: The Paiute cutthroat trout (PCT) and Lahontan cutthroat trout (LCT, *O. c. henshawi*) are morphometrically and meristically identical. However, LCT are heavily spotted (particularly below the lateral line) and are bronze to olive in coloration, while PCT are virtually spotless, have iridescent copper, green, or purplish-pink body coloration, and retain their parr marks into adulthood (Moyle 2002, USFWS 2004, Finger et al. 2013). Originally described by Snyder (1933) as being spotless, most Paiute cutthroat trout have 1-5 small spots, with some having up to 9 spots above the lateral line (USFWS 2004). They possess the characteristic cutthroat reddish-orange slash at the base of the mandible and all meristic characteristics such as gill raker counts, pyloric caeca, lateral line scales, and number of vertebrae are within the range of those for LCT (Moyle 2002, USFWS 2004).

Taxonomic Relationships: The Paiute cutthroat trout is closely related to the Lahontan cutthroat trout and is the least genetically diverse species of trout in the state (Finger et al. 2013). Snyder (1933, 1934) first described this trout as *Salmo seleniris*, a species distinct from LCT based on coloration, the complete or near absence of spotting, and slender body shape. The name *seleniris* is a reference to the moon goddess, Selene (Moyle 2002). Vestal (1947) reclassified PCT as a subspecies of LCT. Subsequently, all North American *Salmo* have been reclassified as *Oncorhynchus*, and PCT is known today as *Oncorhynchus clarkii seleniris* to reflect the original, double-i spelling (USFWS 2004). Recent investigations of population structure for the Lahontan group of cutthroats (Lahontan, Paiute, and Humboldt) show that there are approximately 90 genetic markers that are distinct in Paiute cutthroat trout, and that these fish did not go through an evolutionary bottleneck in the past as previously hypothesized (Finger et al 2011, 2013). Interestingly, PCT share the most genes with Lahontan cutthroat trout from the out-of-basin Independence Lake, not the downstream Carson River, from which it has had the most recent connection, about 8,000-10,000 years ago (Behnke 2002, Finger et al. 2013). The latest research supports the hypothesis that PCT represent a distinct evolutionary lineage that diverged from LCT prior to LCT differentiating into their current populations (Saglam et al. 2017).

Life History: We can assume that the life history of LCT in small, cold headwater streams is similar to that of PCT in the drainages in which they reside. No PCT populations exist in areas that have temperature extremes observed in some of the LCT habitat and it is unknown if they have the capacity to survive the high levels of alkalinity, turbidity and temperature that LCT can withstand. Descriptions of LCT life history are presented in Moyle (2002) and Behnke (2002) and updated in the LCT chapter here.

Few life history studies have been conducted on PCT and most of what is known about them comes from studies of introduced populations in Cottonwood Creek in the White Mountains by Wong (1975) and Caldwell and Titus (2009) of the California Department of Fish and Wildlife (CDFW). PCT life expectancy is about 3-4 years of age in the wild, although some individuals live up to 6 years (Titus and Calder 2009). They mature at 2 years of age, and thus most have the potential to successfully spawn for only about two years on average (Wong 1975, USFWS 2004). Peak spawning activity takes place during the months of June and July (USFWS 2004). Mature fish range from 15-25cm TL. Females use their tails to dig redds in clean gravel substrate. The fertilized eggs hatch in approximately 6-8 weeks, and embryos spend an additional 2-3 weeks in the gravel as alevin before emerging as fry. The juveniles rear in backwaters, shoals and small tributaries until they reach approximately 50mm TL (Wong 1975, USFWS 2004). Adult fish establish dominance hierarchies and defend their established territories from intruders. The larger fish possess the more desirable pool habitats and smaller fish are relegated to riffle and run territories. Most fish do not undertake large-scale movements, and instead reside in close proximity to areas in which they were reared or introduced (USFWS 2004, W. Somer, CDFW, pers. comm. 2016). They require pools for overwintering habitat and are consequently vulnerable to ice scour during low winter flows, which are common in the small headwater tributaries they inhabit (USFWS 2004).

PCT, like most trout, are opportunistic feeders, consuming a variety of aquatic and terrestrial invertebrates in drift (Wong 1975, USFWS 2004). Growth rates depend on water temperature, stream size, and food availability, which can be quite low in the high-elevation, low-productivity Silver King drainage and out-of-basin habitats where PCT currently reside (W. Somer, CDFW, pers. comm. 2016). Titus and Caldwell (2009) back-calculated growth curves for PCT in different drainages based on length-at-age measurements and found that 1 year-old PCT were typically around 30mm TL, and that 6 year-old PCT could grow to about 260mm TL. Few PCT reach lengths over 25 cm, and the largest recorded PCT in Silver King Creek was 34cm (USFWS 2004). The largest PCT found in a lake measured 46cm and weighed 1.1 kg (USFWS 2004, Behnke 2002). There are no known naturally occurring populations of lacustrine Paiute cutthroat trout, although several attempts have been made in the last century to establish them in lakes outside their historical range through stocking.

Habitat Requirements: The only studies of PCT habitat requirements and preferences come from introduced populations in the North Fork of Cottonwood Creek, which were planted there in 1946 (W. Somer, CDFW, pers. comm. 2016). PCT seem to have similar requirements to other alpine stream trout, especially LCT: cold (<18-20°C), well oxygenated water, abundant cover and vegetation, clean gravel to spawn in and an adequate food source. Spawning begins when water temperatures reach 6-9°C (Behnke 2002). Despite decades of heavy grazing in the area, riparian vegetation is rebounding to provide shading and woody debris in Silver King Creek (W. Somer, CDFW, pers. comm. 2016). Deeper pool habitat and overhanging vegetation provide important refuge and overwintering areas (USFWS 2013).

Distribution: PCT are native to just a single drainage, Silver King Creek, in eastern California. Silver King Creek is a tributary of the East Fork Carson River located at an elevation of about 2400m in the Carson-Iceberg Wilderness in Humboldt-Toiyabe National Forest (Alpine Co.) (Caldwell and Titus 2009). As such, the native habitat of this species is the smallest of any known salmonid in North America (W. Somer, CDFW, pers. comm. 2016). They are thought to have existed in only 14.7 km of habitat from the base of Llewellyn Falls downstream to Silver King Canyon, including three small tributary creeks in the drainage, Tamarack Creek, Tamarack Lake Creek, and the lower reaches of Coyote Valley Creek downstream of barrier falls (USFWS 2004). By the time they were described in 1933, their native range had already been invaded by rainbow trout (*O. mykiss*), Lahontan cutthroat trout, and California golden trout (*O. m. aguabonita*). Conflicting recollections of the Silver King Basin's early settlers has complicated the true early distribution of PCT. CDFW records indicate that the first transfer of PCT out of their historical range took place in 1912 by Joe Jaunsaras, a Basque herdsman and early grazing permittee in the basin (USFWS 2004). Virgil Connell, another early settler in the area, reported that during this period the fish below the falls became "...mixed with other kinds, probably due to the stocking on the lower stream of different varieties." By the 1930s, PCT below the Llewellyn Falls were already highly introgressed with rainbow, golden, and Lahontan cutthroat trout (USFWS 2004). Two more creeks in the Silver King Drainage, Corral Valley Creek and Coyote Valley Creek, held PCT by the time Virgil Connell lived there. Falls near the mouth of Corral Valley Creek were presumably a historical upstream barrier to fish, but Vestal (1947) attributed the fish's presence there to herdsmen who "... reportedly planted Piute (*sic*) trout a few at a time in buckets from Upper Fish Valley" (USFWS 2004).

Throughout the 1900s, many transfers were made of Paiute outside the Silver King Basin. The first transfer to Leland Lakes in 1937 failed probably due to the presence of other salmonids. Next, about 400 fish were taken to the North Fork of Cottonwood Creek, a high elevation spring-fed creek in the White Mountains, in Mono County. That population persists to this day and has been a source population for most of the studies (Wong 1975, Titus and Calder 2009) and potential source for reintroduction of genetically pure PCT back into their native range. Introductions continued but were ultimately unsuccessful at McGee Creek (1956), Bull Lake (1957), Delaney Creek (1966), and nearby Heenan Lake (already a broodstock hatchery for LCT) in 1983. The only self-sustaining lacustrine population of PCT became established in Bircham Lake in Inyo County (1957), but by the early 1980s, this population was highly introgressed with rainbow trout (USFWS 2004). Of the 10 known introductions of PCT, there are reproducing populations only in Cottonwood Creek (Mono Co.), Cabin Creek (Mono Co.), Stairway Creek (Madera Co.), and at the outflow of Sharktooth Lake (Fresno Co., See Table 1). In addition to those four out-of-basin populations, another five pure populations of PCT exist in Coyote Valley, Corral Valley, Four Mile Canyon, and Upper Silver King creeks above the historical barrier at Llewellyn Falls (Finger et al. 2013).

Table 1. Known introductions of Paiute cutthroat trout in California by County.

Alpine County	Mono/Inyo/Tuolumne Counties	Fresno/Madera Counties
Silver King Creek (above Llewellyn Falls)* Corral Creek* Coyote Creek* Fly Valley* Four Mile* Bull Lake** Heenan Lake**	North Fork Cottonwood Creek Delaney Creek** McGee Creek** Cabin Creek Bircham Lake**	Sharktooth Lake Stairway Creek
*Introduced in-basin population (Tributaries of Silver King Creek) **Failed Introduction	**Failed Introduction	

Trends in Abundance: Paiute cutthroat trout were originally listed under the ESA as Endangered in 1967 due to their very small distribution and abundance, but were later downlisted to threatened in 1975 to allow flexible management of the species (USFWS 2013).

The USFWS (2013) determined that Paiute cutthroat trout currently occupy 37.8 km of stream habitat in five widely distributed drainages, and while they face several important threats, they have a high potential for recovery based on recent actions of CDFW, USFWS, USFS, and other partners to eliminate introgressed trout in their native range. Different creeks in- and -out-of-basin support different densities, size, and age classes of PCT based on a variety of factors such as growing conditions, carrying capacity of streams, available feeding, spawning, and rearing habitat, etc. CDFW population assessments in 2007 found approximately 1,025 PCT per mile in Coyote Valley Creek and about 485 PCT per mile in Silver King Creek (CDFW 2007). Further study by Titus and Caldwell (2009) indicated that from 2004 to 2007, the mean number of age classes present in population segments increased from 2 to 5, indicating demographic expansion associated with population stability. Upper Silver King Creek in Upper Fish Valley and Coyote Valley Creek also supported a complete age structure during those years (Titus and Caldwell 2009).

According to the USFWS (2013) and UC Davis researchers (Finger et al. 2011, 2013), there are nine streams that currently hold pure populations of Paiute cutthroat trout. Four Mile, Fly Valley, and Corral Creeks have all had numerous surveys and those populations appear to have long-term stability (despite some fluctuations) with about 400 to 700 fish in each (USFWS 2004). The out-of-basin populations (North Fork Cottonwood, Cabin, Stairway, and Sharktooth creeks) have been surveyed by visual assessment or fly fishing to prevent injury or mortality from electrofishing (W. Somer, CDFW, pers. comm. 2016). They appear to be stable, though populations likely declined somewhat in tributary streams due to reductions in available habitat resulting from ongoing drought, such as drying and anchor ice during winter (W. Somer, CDFW, pers. comm. 2016). Over the past three summers (2013-15), field crews have monitored populations at each source, and will continue to do so in the coming field seasons. (W. Somer, CDFW, pers. comm. 2016).

PCT populations have changed since the turn of the century due to impacts stemming from the ongoing drought in California. PCT numbers likely declined in the last three years due to the effects of lower flows and associated anchor ice due to drought (W. Somer, CDFW, pers. comm. 2016). Current populations are small, isolated by barriers and cannot interbreed. Non-native trout have been eradicated below Llewellyn Falls down to barriers in Silver King Canyon; though unlikely, conservation efforts could be unraveled by illegal introduction of non-native fish upstream of the barriers into the current PCT refuge habitat (C. Mellison, USFWS, pers. comm. 2016). In the short term, Paiute cutthroat trout are susceptible to decline due to small population size that can lead to genetic bottlenecks and drift, loss of habitat due to drought, habitat, and solid freezing in winter (as seen with McCloud redband trout) (USFWS 2013, R. Bloom, CDFW, pers. comm. 2016). The USFWS determined that individual population extirpation due to a catastrophic event (such as an extreme fire event) is likely in PCT. Any reduction in the population size and stability of PCT and resulting loss of genetic diversity can combine to create an extinction feedback cycle in which each threat exacerbates and makes the impacts of the other threats more likely (Lusardi et al. 2015).

After a multi-year rotenone treatment of Silver King Creek and tributaries to remove introgressed trout concluded in summer 2015, PCT now have access to 17.7 km of unimpeded stream habitat on Silver King Creek without barriers (W. Somer, CDFW, pers. comm. 2016).

Some individuals from refuge populations in tributaries to Silver King Creek have already volitionally recolonized the treatment area (W. Somer, CDFW, pers. comm. 2016). CDFW will continue to direct considerable resources to monitoring the treatment site and population of PCT in the coming field seasons to document the expansion of PCT back into their native range. The recovery plan lays out a goal of 2,500 individuals greater than 75 cm in length in order to make sure three or more age classes of PCT inhabit their native range downstream of Llewellyn Falls over five consecutive years.

Though Paiute cutthroat have a far more limited distribution than LCT, their habitat is in reasonably good condition and is all on public land (U.S. Forest Service). However, continued reintroduction efforts and transfers will be most likely be required to both maintain and maximize diversity in the nine remaining populations. Monitoring and removal of any alien trout must continue indefinitely to protect genetic integrity of the remaining fish. The high-elevation alpine habitat of PCT could buffer them against some impacts of climate change, but the USFWS still consider it a main threat to the species due to the effects of warming stream temperatures, lower streamflow in summer and autumn months, loss of habitat connectivity, and loss of snowpack (USFWS 2013). In addition, historical drought conditions in California have been much worse than the current drought, likely negatively impacted Paiute and helping to shape their populations (USFWS 2013). Small streams where Paiute live are more susceptible than larger ones to desiccation, high stream temperatures, and loss of adequate deep pool habitat (Haak et al. 2010), though the native range of PCT lies higher than 2,400m in altitude and less likely to rise to unsuitable temperatures. Out-of-basin populations at Cottonwood and Cabin Creeks may be more at risk because of the arid nature of the White Mountains (Haak et al. 2010). The status determination (Table 2) gauges the impacts of such threats on the effective wild populations, including introduced populations, of PCT.

Factors Affecting Status: PCT are relatively stable in number in their remaining refuge populations. Recent, volitional re-colonization is occurring in the Silver King basin following completion of rotenone treatment of the upper Silver King Creek as a priority action of the USFWS Recovery Plan for PCT (W. Somer, CDFW, pers. comm. 2016). The entire native range of PCT is in the Humboldt-Toiyabe National Forest, a factor that eases some of the management difficulties that plague other species' recovery plans, such as LCT, that inhabit a landscape with a patchwork of private and public land. The biggest threats to the persistence of PCT include 1) competition and hybridization with alien trout, especially California Golden trout, coastal rainbow trout, and LCT, 2) loss of genetic diversity due to isolated, small populations developing genetic bottlenecks, and 3) catastrophic loss by stochastic events, such as fire.

Alien species. Alien trout are the principal threat to the continued persistence of PCT. The introduction of non-native rainbow, golden and Lahontan cutthroat trouts in the historical range of the PCT below Llewellyn Falls resulted in the extirpation of PCT there. PCT readily hybridize with rainbow, golden, and cutthroat trout, resulting in the loss of genetic integrity and phenotypic distinctiveness. To conserve PCT, CDFW carried out a large non-native trout removal project spanning summer 2013-2015. After costly, contentious legal battles and delays over use of the piscicide rotenone in Silver King basin, the treatment was completed (Finlayson et al. 2010, W. Somer, CDFW, pers. comm. 2016). For a full treatment of the legal battles of the 1990s and 2000s over California Department of Fish and Wildlife's use of rotenone to eradicate fish species due to concerns about harm to listed invertebrates, water quality, and other issues, see CDFW 2014. The commitment of CDFW, USFWS, US Forest Service, and other partners to remove introgressed and non-native trout and erect barriers in Silver King Canyon has allowed access for downstream recolonization of PCT from refuge populations in headwater streams to their native habitat and the real prospect of PCT recovery.

The nature of the small, low-density, disconnected populations of PCT complicate recovery efforts focused on maintaining and increasing genetic diversity. There are three distinct lineages within current PCT populations as a result of their stocking history (Cordes 2004). Finger et al. (2013) showed that small PCT populations may suffer founder effects and genetic drift. This is due to the fact that most past transfers of PCT consisted of a few dozen fish into small streams with limited habitat. As a result, no population of PCT currently possesses all of the species' alleles, so future reintroductions from source populations in Silver King Creek should be carefully designed to avoid founder effects and consequent reduction of unique alleles in the remaining PCT population (Finger et al. 2013, UWFWS 2013). For instance, the Cabin and Sharktooth creeks populations have private alleles, or valuable genetic information not found in the other refuge populations, that could be vital to protecting a full range of phenotypic expression in PCT offspring (Finger et al. 2013). Uniqueness among populations must be maintained to confer potentially advantageous ecological adaptations and fitness that will be necessary for recovery of the species facing multiple evolving threats such as climate change (Finger et al. 2013, USFWS 2013).

Implementation of the PCT Genetic Management Plan and careful, adaptive management of reintroduction and potential future translocation efforts can help avoid potentially devastating losses of fitness and ecological adaptation. Using too many or too few individuals without proper monitoring could lead to loss of diversity through in-breeding or out-breeding depression. Since existing populations were originally established with so few individuals, introducing their genes into another population could potentially reduce fitness of offspring, and the costs and benefits of such reintroductions must be carefully weighed before active translocations are considered (Finger et al. 2013).

Grazing. Historically, the Silver King Basin was subject to heavy grazing pressure from livestock and caused heavy degradation of riparian habitats, akin to the habitat of the California golden trout. For a full discussion of grazing impacts, see that species section. As part of recovery efforts, grazing allotments in Silver King basin were permanently closed in 1994, but riparian habitat is still recovering throughout much of the PCT range and has shown marked progress with narrowing and deepening channels in meadow portions of Silver King Creek (W. Somer, CDFW, pers. comm. 2016).

Table 2. Major anthropogenic factors limiting, or potentially limiting, viability of populations of Paiute cutthroat trout. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. Certainty of these judgments is high. See methods for explanation.

Factor	Rating	Explanation
Major dams	n/a	No major dams or diversions in PCT habitat.
Agriculture	n/a	Most occupied and historical habitat lies within National Forest boundaries.
Grazing	Medium	Long-term degradation of watersheds has occurred as a result of sheep and cattle grazing, but has been closed since 1994 and is slowly recovering.
Rural/ residential development	n/a	Most Paiute cutthroat trout is within National Forest boundaries.
Urbanization	n/a	
Instream mining	n/a	
Mining	Low	Gold and silver mining may have altered habitat in the past, but the impacts are largely unknown.
Transportation	n/a	Roads are not near most PCT populations.
Logging	Low	The remote locations and high elevations were logged in the past, with signs of historical dam building to convey logs downstream, but impacts on PCT are unknown.
Fire	High	Fires may increase siltation of habitat, and potentially wipe out entire populations of PCT in some areas. Climate change may increase the risk of catastrophic fire.
Estuarine alteration	n/a	
Recreation	Low	Fishing has been closed since 2006 in their native habitat.
Harvest	n/a	
Hatcheries	n/a	There are no hatcheries currently raising PCT to avoid genetic impacts from domestication of wild fish.
Alien species	High	Alien trout represent the highest threat to continued PCT recovery and persistence in the wild through competition and hybridization. Small, fragmented populations remain in close proximity to alien trout.

Effects of Climate Change*:* The most recent US Fish and Wildlife Service five year status review of PCT found that climate change is likely an important threat to PCT for a variety of reasons. First, climate change in the Eastern Sierra is likely to cause increases in water temperature, associated decreases in streamflow, hydrography changes, and more frequent and pronounced cycles of drought and fire (USFWS 2013). Climate change impacts in California are likely to lengthen the fire season, while increasing frequency and intensity of fires. Extreme fire events are likely a greater threat to PCT than to many other salmonid species in California due to the fragmented, isolated habitats that currently sustain refuge populations of the species at high altitude in remote areas (Haak et al. 2010). However, the native range of PCT is characterized by patchy vegetation and granite rather than an abundance of fuels, so once established in their native range in Upper Silver King Creek, these risks may diminish.

Status Score = 2.1 out of 5.0. High Concern. PCT have a high likelihood of extirpation in their native range within the next 50 years without continued commitment to intense monitoring and management. All populations are small and isolated, and therefore highly susceptible to illegal introductions of alien trout and local stochastic events. The 2013 USFWS Recovery Plan's short-term goal is to restore PCT to their native range in Silver King Creek and continue to monitor and protect all existing populations. The newly-available native habitat may have a carrying capacity of between 4,000 to 7,000 individuals, which is more than enough to meet the recovery population goal of 2,500, giving hope for recovery of this species in the future (W. Somer, CDFW, pers. comm. 2016). Table 3 presents a snapshot in time of the various factors affecting PCT status. If the treatment is deemed successful, these scores will be adjusted upward to reflect new accessible habitat range much greater than historical habitat.

Table 3. Metrics for determining the status of Paiute cutthroat trout, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is high. See methods for explanation.

Metric	Score	Justification
Area occupied	2	Occupies several watersheds, but connectivity between headwater populations has recently been established on Silver King Creek.
Estimated adult abundance	2	The largest effective population may be less than 1,000 individuals, but most are much smaller.
Intervention dependence	2	Human assistance required to maintain and increase genetic diversity through reintroduction efforts and protect limited habitats.
Environmental tolerance	2	Actual physiological tolerances not known but adapted for small cold-water headwater streams, which suggests limited tolerance.
Genetic risk	1	Genetic diversity is very low.
Climate change	3	Vulnerable because streams very small and some may become dry during droughts.
Anthropogenic threats	3	2 High, 1 Medium.
Average	2.1	15/7.
Certainty (1-4)	4	Well documented in peer reviewed literature and agency reports.

Management Recommendations: The Paiute cutthroat trout is still listed as threatened under the federal Endangered Species Act but is recovering. The species is volitionally re-populating its native habitat in over 17 km of uninterrupted habitat in Silver King Creek from refuge populations in headwater tributaries. The USFWS laid out clear recovery objectives in order to de-list PCT:

1. Remove all nonnative salmonids in Silver King Creek below Llewellyn Falls to barriers in Silver King Canyon. This was completed in the summer of 2015.
2. Introduce a viable population inhabiting all historical habitat in Silver King Creek and downstream tributaries to barriers in Silver King Canyon. Volitional recolonization from headwater refuge populations is ongoing.
3. Existing PCT should be maintained in all occupied streams, including out-of-basin streams should be maintained as refugia and secured from invasion from alien trout species. This is ongoing.
4. A long-term conservation plan agreement must be developed to manage PCT once the species is delisted. This has not yet occurred.

One of the primary management actions noted in the revised 2013 USFWS Recovery Plan was removal of non-native trout in the waters between Llewellyn Falls to Silver King Canyon. This was finally completed by piscicide treatment in 2015. Concern for the ESA-listed mountain yellow-legged frog (*Rana mucosa*), Yosemite toad (*Bufo canorus*), and opposition over the use of rotenone compounds for fear the treatments could harm invertebrates, downstream fishing in the East Fork Carson River, and the ecology of the treatment area significantly delayed removal of introgressed trout in the Silver King drainage and led to studies meant to minimize their impact (Finlayson, Somer, and Vinson 2010, USFWS 2013).

Unauthorized transfers both of PCT and the non-native trout that threaten them have been both a scourge and a savior. In 1949, an unauthorized transfer introduced rainbow trout above the natural barrier at Llewellyn Falls caused PCT to disappear as 'pure' fish; had it not been for the 1946 stocking in Cottonwood Creek and introduced populations within Silver King basin in Fly Valley and Four Mile Creeks, PCT could well have been lost. A cycle of authorized and unauthorized introductions, hybridization, and eventual culling has occurred repeatedly throughout the 20th century for PCT, and reintroduction efforts remain necessary for their conservation and recovery. Current PCT populations are fairly stable, even during drought, but have low genetic diversity (W. Somer, CDFW, pers. comm. 2016). All non-native salmonids have been eradicated from headwater streams to downstream barriers in Silver King Canyon, which was one of the most challenging aspects of recovery deemed necessary to delist the species under ESA. CDFW, U.S. Forest Service and USFWS partners continue to evaluate whether the treatment was successful, and monitor the populations in Silver King Creek to determine abundance and collect data on post-treatment stream conditions. No restocking PCT in Silver King Creek has begun as of this writing, but some individuals have been found in the treatment area that volitionally moved downstream and are starting to re-populate their native range. Paiute monitoring and re-introduction efforts will likely demand a significant portion of CDFW's wild trout efforts in the next few years.

In summer 2017, reintroduction efforts may begin using source populations from headwater streams to Silver King Creek (Finger et al. 2013, W. Somer, CDFW, pers. comm. 2016, R. Bloom, CDFW, pers. comm. 2016). Further recovery efforts entail plans by CDFG and USFWS to include monitoring and securing all existing populations of PCT and their habitat from invasions of non-native trout, as well as continued analysis on the appropriateness of utilizing out-of-basin individuals for reintroduction to their native range (USFWS 2013, W.

Somer, CDFW, pers. comm. 2016). One possible option under an adaptive management approach is to translocate individuals from different reaches and size classes from in-basin refuge populations to their native range. This could be done during spawning to increase exchange of genetic material (Finger et al. 2013).

The idea to employ a conservation hatchery to help recover PCT was discussed, but ultimately rejected (W. Somer, CDFW, pers. comm. 2016). The very low numbers of individuals, coupled with very limited and difficult access into Silver King basin, make this inappropriate. In addition, fishery managers are leery of altering the genetics of these fish through domestication (W. Somer, CDFW, pers. comm. 2016). The very remote and publicly owned habitat of PCT has limited degradation, giving the species a good chance of recovery and delisting in the future. However, illegal introductions of alien trout continue to be a major threat to the continued existence of this species, especially because of the close proximity of PCT to introgressed and non-native trout in lower Silver King Creek and the East Fork Carson River downstream of the barriers in Silver King Canyon (USFWS 2013). Fishing was closed in Silver King Creek in 2006 to help aid species recovery following chemical treatment (USFWS 2013). In addition, small beaver dams were removed during treatment of the Silver King creek to facilitate chemical treatment. The dams have since been naturally rebuilt by beavers and have created valuable pool habitat. During the time of initial listing of PCT, beaver control and dam removal were thought to be required actions necessary to recover the species in its native habitat, but recent evidence suggests that beaver are actually native to the Eastern Sierra and could have positive impacts on the species through creation of dams and valuable pool refuge.

WHITEFISH

MOUNTAIN WHITEFISH
***Prosopium williamsoni* (Girard)**

Moderate Concern. Status Score = 3.4 out of 5.0. Mountain whitefish persist in some fragmented water bodies in California, but their overall abundance and distribution are reduced from historic levels and may be continuing to decline. Population estimates are generally lacking throughout their range, as are comprehensive distribution surveys, so their overall status remains uncertain.

Description: Mountain whitefish are silvery, large-scaled (74-90 on lateral line) salmonids with a prominent adipose fin, a small ventral mouth, a short dorsal fin (12–13 rays), a cylindrical body and a forked tail. Gill rakers are short (19–26 on the first gill arch), with small rakers. They have 11-13 anal fin rays, 10-12 pelvic fin rays (with a conspicuous axillary process at the base), and 14-18 pectoral fin rays. Their sides are silvery and their backs are olive green to dusky. Scales on the back are often outlined in dark pigment. Breeding males develop distinct tubercles on the head and sides. Juveniles are elongate and silvery with 7–11 dark, oval parr marks.

Taxonomic Relationships: Mountain whitefish are sometimes placed in a separate family, the *Coregonidae* (Moyle 2002), from other salmonids and are regarded as one species throughout their extraordinarily wide range. However, a thorough genetic analysis may reveal a number of distinct population segments within this range. The Lahontan population in California and Nevada is the one most isolated from other populations and may eventually be recognized as a distinct taxon.

Life History: Mountain whitefish are usually observed in loose shoals of 5–20 fish, close to the bottom. As their subterminal mouths and body shape suggest, they are bottom-oriented predators on aquatic insects (Moyle 2002). Juveniles feed on small chironomid midge, blackfly, and mayfly larvae but their diet becomes more diverse with increasing size. Adults feed on mayfly, caddisfly, and stonefly larvae during summer (Ellison 1980). In Lake Tahoe, they consume snails, a variety of insect larvae, crayfish, and amphipods (Miller 1951). Most feeding takes place at dusk or after dark, but they will feed during the day on drifting invertebrates, including terrestrial insects (Moyle 2002). The invertebrate diet and feeding patterns of mountain whitefish are slightly different than those of trout species with which they typically share habitat (Behnke 2002).

According to Moyle (2002), "Growth is highly variable, depending on habitat, food availability, and temperature. Growth of fish from a small alpine lake (Upper Twin, Mono County) was… 11 cm SL at the end of year 1, 13.5 cm at year 2, 15 cm at year 3, 17 cm at year 4, and 20 cm at year 5. Fish from rivers at lower elevations seem to be 25-30 percent larger at any given age after the first year. Young reared in tributaries to Lake Tahoe were largest in the Truckee River (8.6 cm FL at 10 months) and smallest (7.3-7.8 cm) in small tributaries (Miller 1951). Large individuals (25-50 cm SL) are probably 5-10 years old. The largest whitefish in California come from lakes; one measuring 51 cm FL and weighing 2.9 kg came from Lake Tahoe. In Fallen Leaf Lake, the population sampled by gill nets was on average 31 cm FL, with the largest fish being 44 cm long (Al-Chokhachy et al. 2009). Rogers et al. (1996) have developed a standard length-weight relationship for mountain whitefish, based on data from 36 populations throughout their range.

Spawning takes place in October through early December and is preceded in streams by upstream or downstream movements to suitable spawning areas, possibly by homing to historical spawning grounds. Movement is often associated with a fairly rapid drop in water temperature, with spawning occurring at 1-11°C (usually 2-6°C). Spawning takes place in riffles where depths are greater than 75 cm and substrates are coarse gravel, cobble and rocks less than 50 cm in diameter. From lakes, most whitefish migrate into tributaries to spawn, although some spawning may take place in gravel in shallow water. Whitefish do not dig redds, but instead scatter eggs over gravel and rocks, where they sink into interstices. The small eggs are not adhesive. Spawning behavior is not well documented, but they seem to spawn at dusk or at night, in groups of more than 20 fish. They are 2-4 years old when they mature, although age of maturity depends on sex and size. Fecundity varies with size, from 770 to over 24,000 eggs per female, with an average of around 5,000 eggs. The embryos hatch in 6-10 weeks in early spring. Newly hatched fish are carried downstream into shallow (5-20 cm) backwaters, where they spend their first few weeks. As fry grow larger, they move into deeper and faster water, where there are rocks or boulders for cover. Fry from lake populations move into the lake fairly soon after hatching and seek out deep cover, such as beds of aquatic plants (Moyle 2002). Egg hatching and fry survival is highly correlated with temperature, and maximum growth of mountain whitefish fry occurs between 13-14°C (Brinkman, Crockett, and Rogers 2013).

Habitat Requirements: Mountain whitefish in California inhabit clear, cold streams and rivers at elevations of 1,400–2,300 m. While they are known to occur in a few natural lakes (e.g. Tahoe), there are few records from reservoirs. In streams, they are generally associated with large pools (<1 m deep) or deep runs. They prefer deep water or large pools as opposed to other salmonid species that seek cover and shade where possible (Behnke 2002). Deinstadt et al. (2004a) notes, "...deeper pool habitat sustains moderate densities of large (> 300mm) mountain whitefish" (pg. 51). In lakes, they typically live close to the bottom in fairly deep water (Al-Chokhachy et al. 2009).

Environmental tolerances of mountain whitefish in California are poorly understood but they are largely found in waters with summer temperatures < 21°C. More northern populations have been reported to have temperature preferences of 10-18°C, depending on season (Ihnat and Bulkley 1984). Spawning has been recorded at temperatures of 1-11°C but 2-6°C is typical, which corresponds with optimal temperatures of less than 8°C for development of embryos (Northcote and Ennis 1994, Brinkman, Crockett, and Rogers 2013). Behnke (2002) and Mebane et al. (2003) noted that mountain whitefish were somewhat more tolerant of adverse water quality (low dissolved oxygen, higher turbidity) than other salmonids and, therefore, likely more resilient in response to environmental change. More recent research has shown that the upper lethal thermal tolerance of mountain whitefish is actually very similar to that of bull trout (23.6°C compared to 23.5°C), and much lower than the brown, Lahontan cutthroat, and rainbow trout (24.7, 28.5and 26.2°C, respectively) with which they are likely to overlap with in California waters (Dunham, Shroeter, and Rieman 2003, Brinkman, Crockett, and Rogers 2013).

Distribution: Mountain whitefish, as the taxon is broadly recognized, are found in western North America, from California to Alaska. They are distributed throughout the Columbia River watershed (including Wyoming, Montana, Oregon, Washington, Idaho, British Columbia, and Alberta), the upper reaches of the Missouri and Colorado rivers, the Bonneville drainage, and the Mackenzie and Hudson Bay drainages in the Arctic. In California and Nevada, they are present in the lower, Little, and Upper Truckee, East Fork Carson, and East and West Walker river drainages on the east side of the Sierra Nevada and in the Humboldt River drainage in Nevada. It is not known if any mountain whitefish reside in the West Fork Carson River drainage in either California or Nevada (Deinstadt et al. 2004 b). Their range includes both natural lakes (e.g., Tahoe, Independence, Cascade, and Fallen Leaf) and streams. Distribution of mountain whitefish in California is difficult to assess because they have not been the focus of studies in the region; sampling efforts mostly target trout species that support important recreational fisheries in the region (W. Somer, CDFW, pers. comm. 2016).

Mountain whitefish used to be present throughout the greater Lake Tahoe basin before construction of several dams in the 1960s and 1970s. South Fork Prosser Creek, just a short distance from the Independence Lake drainage, supported small numbers of mountain whitefish, but they were gone by 1983, perhaps as a long-term consequence of the construction of Prosser Reservoir in 1962 (CDFW 2015 database). They are also absent from Sagehen Creek, Nevada County after construction of Stampede Reservoir (Decker and Erman 1992, Moyle unpubl. data, W. Somer, CDFW, pers. comm. 2016). Nearly annual surveys conducted on the Upper Truckee River from the mouth of Lake Tahoe upstream have only occasionally turned up mountain whitefish, and the fish have only been encountered in lower reaches of the Upper Truckee River and Lake Tahoe itself, suggesting that this population may spend only a portion of their lives in the river (M. Maher, CDFW, pers. comm. 2011, US Forest Service 2013). Of 25 creeks sampled in the Tahoe basin, only Taylor Creek and the mainstem Upper Truckee River near its mouth into Lake Tahoe had mountain whitefish (US Forest Service 2013). They are absent from the Susan River and from Eagle Lake (Lassen Co.), which contain other Lahontan drainage fishes.

Stream surveys by CDFW have indicated that mountain whitefish do not use high-gradient headwaters of streams and tributaries in the Carson River basin (such as those near Silver King Canyon) and are restricted to low-gradient portions of tributaries and larger rivers in California (Deinstadt et al. 2004a). Over one-quarter of streams sampled by electrofishing in the East Fork Carson River drainage had mountain whitefish present in California, and nearly all of 26 streams sampled across the state line in Nevada showed mountain whitefish were present in the drainage near Highway 395 (Deinstadt et al. 2004a).

Trends in Abundance: Mountain whitefish are still present in much of their limited California range, but their populations are disconnected from one another. This represents a marked decline from the 19th century, when they were harvested in large numbers by Native Americans and commercially harvested in Lake Tahoe (Moyle 2002). There are still runs in tributaries to Lake Tahoe, but they are very small, poorly documented, and seemingly shrinking (US Forest Service 2013). Whitefish were apparently already reduced in numbers by the 1950s. They still appear to be present in low-gradient reaches of the Truckee, East Fork Carson, East and West Walker, and Little Walker rivers, albeit at low to moderate densities of approximately 120 – 300 fish per mile

(Deinstadt et al. 2004a). Higher densities tended to be found in pool and riffle-pocket water areas, and suggest that the East Fork Carson River may carry more and larger whitefish than the surrounding streams (Deinstadt et al. 2004a). Small populations are also present in the Little Truckee River, Independence Lake and some small streams, such as Wolf and Markleeville creeks, tributaries to the East Fork Carson River. Dams and reservoirs have fragmented their populations in Sierra Nevada rivers and tributaries, and whitefish are generally scarce in reservoirs (Moyle 2002). They disappeared from Sagehen and Prosser creeks 10-20 years after construction of reservoirs that covered their lower reaches (Erman 1973, Moyle, unpubl. data). However, a population in nearby Independence Lake (a natural lake) did not show an obvious decline in the period from 1997- 2005 (Rissler et al. 2006). In fact, Independence Lake still has an intact native Lahontan species assemblage and mountain whitefish seem to be thriving there, unlike surrounding impounded waters, which is probably due to the presence of non-native trout in many other impoundments (W. Somer, CDFW, pers. comm. 2016). Recent observations suggest that mountain whitefish are less abundant and less widely distributed in California than they once were, although they continue to be common enough in the Truckee, Carson, and Walker rivers to be caught in recreational fisheries (M. Wier, CalTrout, pers. comm. 2016). However, there is some indication from diving and raft electrofishing surveys of dramatic decline in the mountain whitefish population in the Truckee River over the past 20 years (R. Cutter, pers. comm. 2013, USFS 2013). At present, California allows 5 whitefish per day to be taken by anglers.

Overall, indications are that whitefish populations have declined significantly in last 20-30 years. The East Fork Carson and East Fork Walker seem to be the riverine strongholds of the species in California at present, with limited electrofishing data supporting this hypothesis (CDFW 2015 database). While mountain whitefish have been observed in 2015 during Paiute cutthroat trout surveys on lower Silver King Creek, tributary to the East Fork Carson River, there is evidence that mountain whitefish and other native species are in decline throughout the Lake Tahoe Basin, possibly displaced by non-native fishes (USFS 2013, W. Somer, CDFW, pers. comm. 2016). Deinstadt et al. (2004a, pg. 118) concludes, "The absence or comparatively low densities of mountain whitefish may indicate that the historical status of this species is changing."

Factors Affecting Status: Factors affecting the abundance and distribution of mountain whitefish in California are poorly documented (Table 1). The keys to understanding their apparent decline, however, are habitat-related: (1) they live primarily in the larger streams of the northeastern Sierra Nevada and associated lakes, (2) they do not do well in reservoirs or streams that have been impounded, and (3) they require high water quality and generally low water temperatures for persistence. In general, they live in the waters of the eastern Sierra Nevada most likely to be impacted by human activities, especially by expanding development (*e.g.,* in areas surrounding Truckee and Lake Tahoe), dams and diversions, and by highways and railroads.

Major dams. Whitefish inhabit the larger streams of the eastern Sierra Nevada, which have been dammed or impounded for agricultural or municipal water delivery. Dams may block movements of whitefish to favored spawning and feeding grounds and create unfavorable conditions both above reservoirs and below them, especially poor water quality. For example, when Farad Dam (Nevada) on the Truckee River was blown out by high flows in 1997-98, the river below it recovered rapidly, with higher flows creating more complex habitat and cooler summer temperatures that favored whitefish and trout. Erman (1986) noted that mountain whitefish abundance dropped in Sagehen Creek following the flooding of its lower reaches by Stampede Reservoir, and they may now be absent from this drainage altogether. In contrast, Carson and Walker rivers, with smaller impoundments, have populations that are still self-sustaining and perhaps stable (W. Somer, CDFW, pers. comm. 2016). It is possible that flow releases to support trout fisheries below dams also improve conditions for mountain whitefish in certain areas, such as the Little Truckee River.

Agriculture. Pasture and alfalfa fields line streams occupied by mountain whitefish, especially in the lower reaches of the West and East Walker rivers in California, as well as in Nevada. Attendant diversions and warm, often polluted, return water may impact whitefish populations, which generally require cold, high quality water. Diversions may also reduce stream flows and corresponding water quality required by whitefish, restricting their ability to seek out deep holding water in rivers and streams.

Grazing. Watersheds in which mountain whitefish occur in California were extensively grazed in the past. Continued open range and allotment grazing may contribute to increased sedimentation and water temperatures, as well as degradation of riparian and stream habitats.

Urbanization. The Truckee River and tributaries to Lake Tahoe have been altered in many ways by urban and suburban sprawl, along with associated road and highway networks; however, the effects and potential impacts of such developments on whitefish are not quantified.

Transportation. The Truckee and Carson rivers have roads and railroads along one or both banks. The effects of long-standing changes are not documented but they reduce natural meandering and contribute pollutants to the streams.

Logging. Mountain whitefish watersheds in California were heavily logged in the 19th century. While logging is less intense than it once was, continued timber harvest operations, including use of roads, may contribute to increased sedimentation and higher water temperatures.

Harvest. Over-exploitation in the past presumably depleted whitefish numbers although this threat is largely gone, in part because few anglers target them today, despite their edibility.

Alien species. Whitefish coexist in many areas with alien brown, brook, and rainbow trout and it is possible that these trout may limit whitefish populations by preying on their fry and juveniles or by competing with them for food and space. In recent years, smallmouth bass have spread into some parts of the Truckee River system, which may present a new predation threat. In lakes where invasive lake trout are present such as Lake Tahoe, Lake Cascade, and Fallen Leaf lakes, mountain whitefish are preyed upon frequently, and this predation pressure may have changed their distribution over time to be less benthic-focused (Vander Zanden et al. 2003).

Table 1. Major anthropogenic factors limiting, or potentially limiting, viability of populations of mountain whitefish in California. Factors were rated on a five-level ordinal scale where a factor rated "critical" could push a species to extinction in 3 generations or 10 years, whichever is less; a factor rated "high" could push the species to extinction in 10 generations or 50 years whichever is less; a factor rated "medium" is unlikely to drive a species to extinction by itself but contributes to increased extinction risk; a factor rated "low" may reduce populations but extinction is unlikely as a result. A factor rated "n/a" has no known negative impact. Certainty of these judgments is moderate due to limited studies of the species in California. See methods for explanation.

Factor	Rating	Explanation
Major dams	Medium	Construction of major dams and associated reductions in water quality and flow, and increases in temperature, have had documented negative impacts.
Agriculture	Medium	Each watershed inhabited by mountain whitefish currently has some kind of diversion or flow modification for agricultural purposes, reducing availability of cold, clean water.
Fire	Low	More frequent large fires are likely within species range in the future and may increase siltation of spawning habitat.
Grazing	Low	Degradation of watersheds from open range grazing increases siltation and temperatures, reducing habitat quality.
Rural/ residential development	Low	Much of the range of mountain whitefish is relatively rural; residential diversions and groundwater pumping may reduce coldwater habitat.
Urbanization	Low	Despite rapid urbanization of the Tahoe/Truckee basins, impacts of increasing sprawl on aquatic species are not well understood. Much of the range of mountain whitefish is relatively rural.
Instream mining	Low	Gravel mining and gold dredging could alter habitat.
Mining	Low	Remnant mines could negatively impact water quality through contaminant seepage; these impacts are not well documented.
Transportation	Low	Roads are present along many streams, and may negatively impact water quality via runoff of contaminants.
Logging	Medium	Most watersheds used by mountain whitefish have been subject to extensive logging in the past; most impacts are likely legacy.
Estuarine alteration	N/A	
Recreation	Low	Some riparian areas damaged by recreational use.
Harvest	Low	Some recreational harvest; poaching may occur.
Hatcheries	N/A	
Alien species	Medium	Mountain whitefish are prey for non-native trout species, especially lake trout, where they are present and predation pressure may shift their distribution and behavior. Competition with non-native trout species may limit abundance.

Effects of Climate Change: Moyle et al. (2013) rated mountain whitefish as "highly vulnerable" to extinction in California in the next 100 years as the result of climate change severely altering their already limited habitats. Climate change effects are likely to be substantial for this species because it relies on sufficient flows of cold water from snowmelt in the otherwise relatively arid Eastern Sierra region. Even small decreases in flows, increases in temperatures and changes in timing of spring runoff could eliminate of further fragment populations that are already constrained by dams, diversions or other factors. Increasing temperatures in the Sierra can decrease thermally suitable habitat, increase direct temperature-related mortality, lower dissolved oxygen content, and potentially increase predation from, and competitive advantages in more tolerant, invasive, warm water species, (Brinkman, Crockett, and Rogers 2013) especially in the Truckee River. In fact, mountain whitefish may be disproportionately affected by climate change compared to other salmonids because they are constrained to low gradient sections of rivers and do not effectively colonize high-gradient headwater streams that could provide thermal refuges. These factors may in fact jeopardize the persistence of the species at the southern portion of its range (Brinkman, Crockett, and Rogers 2013).

Status Score = 3.4 out of 5.0. Moderate Concern. Mountain whitefish are locally abundant in some areas, although their overall abundance and distribution are reduced from the past. Little is known about their abundance, distribution and population trends, but survey information indicates mountain whitefish are a declining species.

Table 2. Metrics for determining the status of mountain whitefish in California, where 1 is a poor value and 5 is excellent. Each metric was scored on a 1-5 scale, where 1 is a major negative factor contributing to status; 5 is a factor with no or positive effects on status; and 2-4 are intermediate values. Certainty of these judgments is moderate. See methods for explanation.

Metric	Score	Justification
Area occupied	4	Present in three watersheds and several lakes.
Estimated adult abundance	4	Numbers appear to be declining in many streams so this number may be high.
Intervention dependence	4	Populations persist but intervention will be needed if their decline continues.
Tolerance	3	Whitefish appear to be more physiologically tolerant than many salmonids, live at least 5 years and are iteroparous; however, they require high water quality and low temperatures.
Genetic risk	4	Genetics have not been studied but most populations are isolated from one another.
Climate change	2	Whitefish are likely to be negatively affected by decreased flows, warmer temperatures and increased diversions but persist in lakes.
Anthropogenic threats	3	4 Medium threats.
Average	3.4	24/7.
Certainty (1-4)	2	Grey literature, survey data, and professional judgment.

Management Recommendations: It is clear that mountain whitefish in California would benefit from a thorough study of their biology including systematics, genetics, distribution, abundance, environmental tolerances, and habitat requirements of all life stages. Existing fish surveys in eastern Sierra Nevada streams where mountain whitefish occur are generally focused on trout species, especially non-native species, and the popular recreational fisheries they support. While mountain whitefish are sometimes captured during these surveys (Deinstadt et al. 2004a, Deinstadt et al. 2000b), few efforts have been made, thus far, to assess their distribution or population trends, although there is some evidence of a potential downward shift in abundance. A shift in fisheries management toward native species restoration and recovery is occurring within their range but is currently focused on Lahontan cutthroat trout, which are a listed species (threatened) under the federal Endangered Species Act of 1973. Because of their low tolerance for high water temperatures and poor water quality, they also are a good indicator of 'health' of the Carson, Walker, and Truckee rivers, as well as of Lake Tahoe and other natural lakes. As such, perhaps the best recommendation to benefit mountain whitefish populations is to advocate that they become an integral part of ongoing management and restoration efforts currently focused on other salmonids. Specific recommendations include: (1) basic research on their biology and distribution, (2) monitoring of existing populations at least once every 5 years, (3) habitat restoration in degraded (simplified) stream reaches in which they are known to live, and (4) maintenance or enhancement of flows in regulated rivers so that temperatures remain below 21° C and high water quality is maintained throughout the year. Of particular importance are the ongoing Truckee River Operating Agreement negotiations. These negotiations focus on providing reliable water for communities in California and Nevada from the Carson, Truckee, and Walker Rivers but also provide an important opportunity to enhance habitat for native fishes, including mountain whitefish (W. Somer, CDFW, pers. comm. 2016).

It is also important to continue to educate the public as to the value of native fishes such as mountain whitefish. For example, in 2010, California Trout and partners from the U.S. Forest Service Lake Tahoe Basin Management Unit, Tahoe Resource Conservation District, CDFW, U.S. Fish and Wildlife Service, and local schools helped rear and release mountain whitefish and speckled dace (*Rhinichthys osculus*) into Lake Tahoe tributaries to promote awareness and conservation of native fishes. Such educational outreach programs could be expanded with partners where possible to increase public awareness of native fishes in the Tahoe Basin.

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Appendix A. Summary of fish species, federal and state listings, status scores in 2008 vs. 2017, Levels of Concern, and top threats.

SOS II Status Scoring Comparison

Species	**ESA/CESA Listing Determination**	**2017 status score**	**2008 status score**	**2008/2017 Level of Concern**	**Threats**	**Top 3 Threats**	**Climate Change**
Central CA coast coho	State and Federally endangered	1.3	1.5	critical/critical	1-critical 3 - high 7 - medium 4 - low	1. Agriculture 2. Urbanization 3. Estuary Alteration (4. Logging)	critical
Chum		1.6	1.5	critical/critical	2 - high 2 - medium 8 - low 3 - n/a	1. Estuary alterations 2. Agriculture 3.logging	critical
Pink		1.6	1.5	critical/critical	2 - high 3 - medium 6 - low 4 - n/a	1. Estuary Alteration 2. Agriculture 3. Logging	critical
CA coast Chinook	Federally threatened	2.9	2.5	high/high	8 - medium 7 - low	1. Dams 2. Logging 3. Agriculture (many medium threats)	High
Central Valley Late fall-run Chinook	State and Federal Species of Special Concern	2.1	2.3	high/high	2 - high 4 - medium 9 - low	1. dams 2. Agriculture 3. estuarine alteration	critical

Central Valley spring-run Chinook	State and Federally threatened	1.7	2.3	high/critical	4 - high 3 - medium 8 - low	1. Dams 2. Hatcheries 3. Agriculture (4. Estuary Alteration)	critical
Sacramento River Winter-run Chinook	State and Federally endangered	1.3	1.7	critical/critical	1 - critical 2 - high 5 - medium 7 - low	1. Dams 2. Hatcheries 3. Agriculture	critical
SONCC coho	State and Federally threatened	1.7	1.8	critical/critical	1 - critical 3 - high 5 - medium 6 - low	1. Hatcheries 2. Estuary alteration 3. Agriculture	critical
UKTR spring-run Chinook	State Species of Special Concern, Federal Sensitive Species	1.6	2.0	high/critical	3 - high 7 - medium 5 - low	1. Dams 2. Hatcheries 3. Logging	critical
UKTR fall-run Chinook	Federal Sensitive Species	3.1	3.3	moderate/ moderate	1 - high 8 - medium 6 - low	1. Dams 2. Mining 3. Hatcheries	high
Central Valley fall-run Chinook	Federal Species of Special Concern	2.7	3.7	moderate/ high	1 - high 8 - medium 6 - low	1. Hatcheries 2. Estuarine alteration 3. Agriculture	high
SONCC Chinook	Federal Sensitive Species	3.1	4.0	low/moderate	8 - medium 4 - low 3 - n/a	1. Logging 2. Estuary Alteration 3. Agriculture	high

Klamath Mountains Province summer steelhead	State Species of Special Concern	1.9	2.0	high/critical	2- high 6 - medium 7 - low	1. Dams 2. Agriculture 3. Logging	critical
Northern CA summer steelhead	Federally threatened	1.9	2.0	high/critical	3 - high 5 - medium 6 - low 1 - n/a	1. Dams 2. Agriculture 3. Estuary alteration	critical
South-Central CA coast steelhead	Federally threatened	1.9	2.3	high/critical	4 - high 2 - medium 8 - low 1 - n/a	1. Dams 2. Agriculture 3. Estuary alteration (4. Fire)	critical
Southern steelhead	Federally endangered	1.9	2.0	high/critical	5 - high 6 - medium 4 - low	1. Dams 2.Urbanization 3. Estuary alterations (4. Hatcheries 5. Fire)	critical
Central CA coast steelhead	Federally threatened	2.0	2.9	high/high	3 - High 3 - Medium 9 - Low	1. dams 2. Urbanization 3. Estuary Alteration	critical
Central Valley steelhead	Federally threatened	3.0	2.5	high/moderate	2 - high 5 - Medium 8 - low	1. Dams 2. Estuary Alteration 3. Hatcheries	Moyle et al. 2013 did not score
Northern CA winter steelhead	Federally threatened	3.3	3.3	moderate/ moderate	2 - high 7 - medium 5 - low 1 - n/a	1. Dams 2. Estuary Alteration 3. Agriculture	high

Klamath Mountains Province winter steelhead	Federal Sensitive Species, State Species of Special Concern	3.3	4.3	low/moderate	9-Medium 6 - Low	1. Dams 2. Hatcheries 3. Mining	high
Bull trout	Extinct	0.0	0.0	extinct/extinct	N/A		
California golden trout	State Species of Special Concern	1.9	2.0	high/critical	1 - high 1 - medium 6 - low 7 - n/a	1. Alien Species 2. Grazing 3. Recreation	critical
Eagle Lake rainbow trout	State Species of Special Concern, Federal Sensitive Species	2.3	2.5	high/high	1 - high 5 - medium 4 - low 5 - n/a	1. Alien Species 2. Hatcheries 3. Harvest	critical
Kern River rainbow trout	State and Federal Species of Special Concern	1.4	2.2	high/critical	1 - critical 1 - high 4 - medium 4 - low 5 - n/a	1. Alien Species 2. Hatcheries 3. Grazing	critical
Lahontan cutthroat trout	State and Federally threatened	2.0	2.5	high/high	1 - critical 2 - high 2 - medium 6 - low 4 - n/a	1. Alien species 2. Hatcheries 3. Fire	high
Little Kern golden trout	Federally threatened	2.0	2.3	high/high	1 - high1 - medium6 - low7 - n/a	1. Hatcheries 2. Alien species 3. Fire	critical

McCloud River redband trout	State Species of Special Concern	1.4	2.3	high/critical	1 - high 4 - medium 4 - low 6 - n/a	1. Alien species 2. Fire 3. Grazing	critical
Paiute cutthroat trout	Federally threatened	2.1	2.3	high/high	2 - high 1 - medium 3 - low 8 - n/a	1. Alien species 2. Fire 3. Grazing	critical
Coastal cutthroat trout	State Species of Special Concern	2.7	3.3	moderate/high	1 - high 7 - medium 6 - low 1 - n/a	1. Logging 2. Estuary Alteration 3. Agriculture	critical
Goose Lake redband trout	State Species of Special Concern	3.1	3.5	moderate/ moderate	1 - high 2 - medium 7 - low 5 - n/a	1. Agriculture 2. Grazing 3. Harvest	high
Coastal rainbow trout		4.7	4.6	low/low	14 - low 1 - n/a	1. Rural/ residential Development 2. Agriculture 3. Alien Species	high
Mountain whitefish	State Species of Special Concern	3.4	4.3	low/moderate	4 - medium 9 - low 2 - n/a	1. Dams 2. Alien Species 3. Agriculture	high

APPENDIX B. Summary of anthropogenic threats scoring by species.

Anthropogenic Threats	Anadromous (21 species including coastal cutthroat trout) and Inland species (10 species including whitefish)					
	N/A	**Low**	**Medium**	**High**	**Critical**	
MAJOR DAMS	Goose Lake redband Paiute cutthroat CA golden Eagle Lake rainbow Little Kern golden	McCloud redband Coastal rainbow trout Pink salmon Chum salmon SONCC Chinook	CV fall Chinook CC Chinook CCC coho Coastal cutthroat Lahontan cutthroat Kern River rainbow KMP winter steelhead Mountain whitefish	CV steelhead CV Late fall Chinook CV spring Chinook SONCC coho UKTR spring Chinook UKTR fall Chinook CCC steelhead NCC summer steelhead NCC winter steelhead KMP summer steelhead Southern steelhead SCCC steelhead	Sac winter Chinook	
total	**5**	**5**	**8**	**12**	**1**	**31**
anadromous	0	3	5	12	1	
inland trout	5	2	3	0		

AGRICULTURE	McCloud redband Paiute cutthroat trout Kern River rainbow CA golden trout Little Kern golden	Coastal rainbow trout Eagle Lake rainbow SONCC Chinook	CV steelhead CV fall Chinook CV Late fall Chinook CC Chinook UKTR spring Chinook UKTR fall Chinook Coastal cutthroat Lahontan cutthroat Chum salmon CCC steelhead NCC winter steelhead KMP winter steelhead Southern steelhead Mountain whitefish	Sac winter Chinook Goose Lake redband CV spring Chinook SONCC coho Pink salmon NCC summer steelhead KMP summer steelhead SCCC steelhead	CCC coho	
total	**5**	**3**	**14**	**8**	**1**	**31**
anadromous	0	1	12	7	1	
inland trout	5	2	2	1	0	

GRAZING		CV steelhead Sac winter Chinook CV fall Chinook CV Late fall Chinook CV spring Chinook Coastal rainbow trout Lahontan cutthroat Pink salmon Chum salmon Little Kern golden CCC steelhead Southern steelhead SCCC steelhead Mountain whitefish	CC Chinook McCloud redband Goose Lake redband CCC coho SONCC coho UKTR spring Chinook UKTR fall Chinook Coastal cutthroat Paiute cutthroat trout Kern River rainbow CA golden trout Eagle Lake rainbow SONCC Chinook NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead			
total	**0**	**14**	**17**	**0**	**0**	**31**
anadromous	0	10	11	0	0	
inland trout	0	4	6	0	0	

RURAL/RESID. DEVELOPMENT	McCloud redband Lahontan cutthroat Paiute cutthroat trout CA golden trout Little Kern golden	CV steelhead Sac winter Chinook CV fall Chinook CV Late fall Chinook CC Chinook CV spring Chinook SONCC coho UKTR spring Chinook UKTR fall Chinook Coastal rainbow trout Kern River rainbow Pink salmon Chum salmon Eagle Lake rainbow CCC steelhead KMP winter steelhead KMP summer steelhead Goose Lake redband Mountain whitefish	CCC coho Coastal cutthroat SONCC Chinook NCC summer steelhead NCC winter steelhead Southern steelhead SCCC steelhead			
total	**5**	**19**	**7**	**0**	**0**	**31**
anadromous	0	14	7			
inland trout	5	5	0			

URBANIZATION	McCloud redband Goose Lake redband Lahontan cutthroat Paiute cutthroat trout Kern River rainbow Pink salmon Chum salmon CA golden trout Eagle Lake rainbow Little Kern golden SONCC Chinook	Sac winter Chinook CV Late fall Chinook CV spring Chinook SONCC coho UKTR spring Chinook UKTR fall Chinook Coastal rainbow trout Coastal cutthroat Mountain whitefish NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead Mountain whitefish	CV steelhead CV fall Chinook CC Chinook SCCC steelhead	CCC coho CCC steelhead Southern steelhead		
total	**11**	**13**	**4**	**3**	**0**	**31**
anadromous	3	11	4	3		
inland trout	8	2	0	0		

INSTREAM MINING	Goose Lake redband McCloud redband Lahontan cutthroat Paiute cutthroat trout Kern River rainbow CA golden trout Eagle Lake rainbow Little Kern golden SONCC Chinook	CV steelhead Sac winter Chinook CV Late fall Chinook CC Chinook CV spring Chinook Coastal rainbow trout Coastal cutthroat Chum salmon CCC steelhead NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead SCCC steelhead Mountain whitefish	CV fall Chinook CCC coho SONCC coho UKTR spring Chinook UKTR fall Chinook Pink salmon Southern steelhead			
Total	9	15	7	0	0	31
anadromous	1	13	7	0		
inland trout	8	2	0	0		

MINING	McCloud redband Coastal cutthroat Kern River rainbow Pink salmon Chum salmon CA golden trout Eagle Lake rainbow Little Kern golden SONCC Chinook NCC summer steelhead NCC winter steelhead	CV steelhead CV Late fall Chinook CC Chinook Goose Lake redband CCC coho CV spring Chinook SONCC coho UKTR fall Chinook Coastal rainbow trout Lahontan cutthroat Paiute cutthroat trout CCC steelhead KMP summer steelhead SCCC steelhead Southern steelhead Mountain whitefish	Sac winter Chinook CV fall Chinook UKTR spring Chinook KMP winter steelhead			
total	11	16	4	0	0	31
anadromous	6	11	4	0		
inland trout	5	5	0	0		

TRANSPORTATION	Paiute cutthroat trout	CV fall Chinook CV Late fall Chinook Goose Lake redband CV spring Chinook Coastal rainbow trout Lahontan cutthroat Kern River rainbow CA golden trout Little Kern golden SCCC steelhead Mountain whitefish	CV steelhead Sac winter Chinook CC Chinook McCloud redband CCC coho SONCC coho UKTR spring Chinook UKTR fall Chinook Coastal cutthroat Pink salmon Chum salmon Eagle Lake rainbow CCC steelhead SONCC Chinook NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead Southern steelhead			
trout	**1**	**11**	**19**	**0**	**0**	**31**
anadromous	0	4	17	0		
inland trout	1	7	2	0		

LOGGING		CV steelhead Sac winter Chinook CV fall Chinook CV Late fall Chinook Goose Lake redband Coastal rainbow trout Lahontan cutthroat Paiute cutthroat trout Kern River rainbow CA golden trout Little Kern golden CCC steelhead Southern steelhead SCCC steelhead	CC Chinook McCloud redband CV spring Chinook SONCC coho UKTR fall Chinook Coastal cutthroat Pink salmon Eagle Lake rainbow SONCC Chinook NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead Mountain whitefish	CCC coho UKTR spring Chinook Chum salmon		
total	**0**	**14**	**14**	**3**	**0**	**31**
anadromous	0	7	11	3		
inland trout	0	7	3	0		

FIRE		CV steelhead Sac winter Chinook CV fall Chinook CV Late fall Chinook CC Chinook Goose Lake redband CCC coho SONCC coho UKTR spring Chinook Coastal rainbow trout Coastal cutthroat Kern River rainbow Pink salmon Chum salmon CA golden trout Eagle Lake rainbow CCC steelhead NCC summer steelhead NCC winter steelhead Mountain whitefish	McCloud redband CV spring Chinook UKTR fall Chinook Little Kern golden SONCC Chinook KMP winter steelhead KMP summer steelhead	Lahontan cutthroat Paiute cutthroat trout Southern steelhead SCCC steelhead		
total	**0**	**20**	**7**	**4**	**0**	**31**
anadromous	0	14	5	2		
inland trout	0	6	2	2		

ESTUARY ALTERTATION	McCloud redband Goose Lake redband Coastal rainbow trout Lahontan cutthroat Paiute cutthroat trout Kern River rainbow CA golden trout Eagle Lake rainbow Little Kern golden Mountain whitefish	UKTR spring Chinook UKTR fall Chinook	Sac winter Chinook CV fall Chinook CC Chinook SONCC Chinook KMP winter steelhead KMP summer steelhead	CV steelhead CV Late fall Chinook CCC coho CV spring Chinook SONCC coho Coastal cutthroat Pink salmon Chum salmon CCC steelhead NCC summer steelhead NCC winter steelhead Southern steelhead SCCC steelhead		
total	**10**	**2**	**6**	**13**	**0**	**31**
anadromous	0	2	6	13		
inland trout	10	0	0	0		

	CV steelhead Sac winter Chinook CV fall Chinook CV Late fall Chinook CC Chinook McCloud redband Goose Lake redband CCC coho CV spring Chinook SONCC coho UKTR fall Chinook Coastal rainbow trout Coastal cutthroat Lahontan cutthroat Paiute cutthroat trout Pink salmon Chum salmon CA golden trout Eagle Lake rainbow Little Kern golden CCC steelhead SONCC Chinook NCC summer steelhead	UKTR spring Chinook Kern River rainbow
RECREATION	NCC winter	Southern steelhead

		steelhead KMP winter steelhead KMP summer steelhead SCCC steelhead Mountain whitefish				
total	**0**	**28**	**3**	**0**	**0**	**31**
anadromous	0	19	2	0		
inland trout	0	9	1	0		

HARVEST	Paiute cutthroat trout	CV steelhead McCloud redband CV spring Chinook Coastal rainbow trout Coastal cutthroat Lahontan cutthroat Pink salmon Chum salmon CA golden trout Little Kern golden Mountain whitefish CCC steelhead NCC summer steelhead KMP winter steelhead KMP summer steelhead SCCC steelhead Mountain whitefish	Sac winter Chinook CV fall Chinook CV Late fall Chinook CC Chinook Goose Lake redband CCC coho SONCC coho UKTR spring Chinook UKTR fall Chinook Kern River rainbow Eagle Lake rainbow SONCC Chinook Southern steelhead NCC winter steelhead			
total	**1**	**16**	**14**	**0**	**0**	**31**
anadromous	0	10	11	0		
inland trout	1	6	3	0		

HATCHERIES	Goose Lake redband Paiute cutthroat trout Pink salmon Mountain whitefish	CC Chinook McCloud redband Coastal rainbow trout Chum salmon CA golden trout Little Kern golden SCCC steelhead Southern steelhead	CV steelhead CV Late fall Chinook CCC coho UKTR fall Chinook Coastal cutthroat Eagle Lake rainbow CCC steelhead SONCC Chinook NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead	Sac winter Chinook CV fall Chinook CV spring Chinook UKTR spring Chinook Lahontan cutthroat Kern River rainbow	SONCC coho	
total	**4**	**8**	**12**	**6**	**1**	**31**
anadromous	1	4	11	4	1	
inland trout	3	4	1	2	0	

ALIEN SPECIES	Pink salmon Chum salmon SCCC steelhead	CC Chinook Goose Lake redband CCC coho SONCC coho UKTR spring Chinook UKTR fall Chinook Coastal rainbow trout Coastal cutthroat CCC steelhead SONCC Chinook NCC summer steelhead NCC winter steelhead KMP winter steelhead KMP summer steelhead	CV steelhead Sac winter Chinook CV fall Chinook CV Late fall Chinook CV spring Chinook Mountain whitefish	McCloud redband Eagle Lake rainbow Little Kern golden CA golden trout Paiute cutthroat trout	Lahontan cutthroat Kern River rainbow Southern steelhead	
total	**3**	**14**	**6**	**5**	**3**	**31**
anadromous	3	12	5	0	1	
inland trout	0	2	1	5	2	

CLIMATE CHANGE	Coastal rainbow trout	CV fall Chinook Paiute cutthroat trout KMP winter steelhead	Goose Lake redband Coastal cutthroat CC Chinook UKTR fall Chinook SONCC Chinook NCC winter steelhead CV steelhead Mountain whitefish	Sac winter Chinook CV Late fall Chinook McCloud redband CCC coho CV spring Chinook SONCC coho UKTR spring Chinook Lahontan cutthroat Kern River rainbow Pink salmon Chum salmon CA golden Eagle Lake rainbow Little Kern golden Southern steelhead KMP summer steelhead NCC summer steelhead SCCC steelhead CCC steelhead

total	**0**	**1**	**3**	**8**	**19**	**31**
Anadromous	0	0	2	6	13	
Inland	0	1	1	2	6	

GRAND TOTALS	**N/A - 65**	**Low - 199**	**Medium - 145**	**High - 62**	**Critical - 25**	**496**
Anadromous	**14**	**135**	**120**	**50**	**17**	**336**
Inland	**51**	**64**	**25**	**12**	**8**	**160**

APPENDIX C. Summary of Medium, High, or Critical threats and High or Critical threats by species group and top threats (highlights).

Summary	**Salmon (Pink, Chum)**	**Chinook**	**Coho**	**Steelhead**	**Trout**
	Medium, High, Critical/High, Critical	MHC/HC	MHC/HC	MHC/HC	MHC/HC
Major dams	9/6	7/5	2/1	8/7	4/0
Agriculture	11/10	7/2	2/2	8/3	4/1
Grazing	6/0	4/0	2/0	4/0	7/0
Rural/residential development	2/0	1/0	1/0	4/0	1/0
Urbanization	3/1	2/0	1/1	4/2	0/0
Instream mining	6/0	3/0	2/0	1/0	0/0
Mining	3/0	3/0	0/0	1/0	0/0
Transportation	9/0	5/0	2/0	7/0	3/0
Logging	9/3	5/1	2/1	4/0	4/0
Fire	3/0	3/0	0/0	4/2	4/2
Estuary alteration	10/6	6/2	2/2	8/6	1/1

Recreation	1/0	1/0	0/0	1/0	1/0
Harvest	9/0	7/0	2/0	2/0	3/0
Hatcheries	9/5	7/4	1/1	6/0	3/2
Alien species	4/0	4/0	0/0	2/1	8/7