Salmon, Steelhead, and Trout in California

Status of an Emblematic Fauna

A report commissioned by California Trout, 2008

PETER B. MOYLE, JOSHUA A. ISRAEL, AND SABRA E. PURDY

CENTER FOR WATERSHED SCIENCES,

UNIVERSITY OF CALIFORNIA, DAVIS

DAVIS, CA 95616

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Beyond Conservation: New knowledge for a new era of river restoration and management.

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SALMON, STEELHEAD, AND TROUT IN CALIFORNIA: STATUS OF AN EMBLEMATIC FAUNA

INTRODUCTION

PETER B. MOYLE, JOSHUA A. ISRAEL, AND SABRA E. PURDY
CENTER FOR WATERSHED SCIENCES,
UNIVERSITY OF CALIFORNIA, DAVIS
DAVIS CA 95616

ABSTRACT

The southernmost populations of salmon, steelhead, and trout, uniquely adapted to California's climatic regime, are in deep trouble. 20 of 31 living taxa (65%) are in danger of extinction within the next century. Of the 22 anadromous taxa, 13 (59%) are in danger of extinction, while seven (78%) of the nine living inland taxa are in danger of extinction. All of these species currently support or historically supported fisheries, thus having economic as well as cultural value. They are also strong indicators of the condition of California's streams; large self-sustaining populations of native salmon and trout are found where streams are in reasonably good condition. The reasons for their widespread decline are complex and multiple, but basically boil down to a combination of human competition for use of the high quality water salmonids require, alteration of the landscapes through which salmonid waters flow, overfishing, and introductions of alien species as predators or competitors. Ensuring ecologically sustainable flows, reducing migratory barriers to juveniles and adults, restoring watersheds, and minimizing competition from non-native salmonids are some of the essential steps to the recovery of California's salmonids. Bringing these fish back from the brink of extinction will not be easy but it is possible, thanks to the inherent adaptability of California's salmonids to changing conditions. However, the growing threats of climate change and increasing human populations, with increases in water use and in intensity of land use, will need to be addressed. In the long run, restoring fisheries for most species, however, will require reducing or at least not increasing human impacts on the California landscape.

INTRODUCTION

Salmon, trout, and their relatives, which make up the fish family Salmonidae (salmonids), are the iconic fishes of the Northern Hemisphere. They are characteristic of the region's cold productive oceans, rushing streams and rivers, and deep cold lakes. They are adapted for life in dynamic landscapes created by glaciers, volcanoes, earthquakes, and climatic extremes. Salmonids thrive through their mobility, moving freely through the ocean and large river systems, as well as their 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, many with distinctive color patterns and other attributes, all with life histories superbly tuned to local environmental conditions (e.g., Behnke 2002, Moyle 2002).

Salmonids have a long history of interactions with humans in the northern parts of the world. Salmon appear as images in Cro-Magnon cave art of 10,000 or more years ago and have been important food for indigenous peoples throughout their range. The importance of salmonids stems from their accessibility and high nutritional content; salmon bring nutrients and calories from the rich northern oceans into streams while trout and other inland forms concentrate the scarce resources present in cold water streams and lakes. In both situations they become available for human harvest and have historically been important food resources. In the 17th century, at the beginning of the Industrial Revolution, angling for trout developed in Europe as a popular source of recreation (Walton 1653). This peculiar aesthetic led to salmonids, mainly brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*), being introduced into suitable waters all over the world, as an artifact of cultural imperialism (Crosby 1986). Their importance as food fish also led to the successful introduction of anadromous salmonids, mainly Chinook salmon (*O. tshawytscha*), Atlantic salmon (*Salmo salar*), and steelhead rainbow trout, into the southern hemisphere. Today, Atlantic salmon are cultured worldwide in cold coastal waters while rainbow trout are cultured in more inland areas; both are farmed in high-production operations to satisfy human demand for their flesh.

Despite their cultural, historic, aesthetic, and economic importance, salmonid fishes are in severe decline in many, if not most, of their native habitats and many populations have been extirpated, especially in heavily industrialized areas (Montgomery 2003). The reasons for this are complex and multiple, but basically boil down to a combination of human competition for use of the high quality water salmonids require, alteration of the landscapes through which salmonid waters flow, overfishing, and introductions of alien species as predators or competitors. Concern for the loss of salmonid fisheries led to some of the earliest fish conservation efforts in Europe but during the 20th century; the principal responses were to culture them in hatcheries while restricting fisheries.

The natural ability of salmon and trout to rapidly adapt to changing conditions has made them relatively easy to culture. Not surprisingly, their life histories and other characteristics have been modified in response to hatchery environments and to match the desires of hatchery managers. This has resulted in some varieties of trout and salmon that are true domestic animals, wonderful for meat production but poor at surviving the wild. For anadromous salmon and steelhead, hatchery operations were established to enhance wild populations, mainly for fisheries. As a result they have had to satisfy two rather contradictory goals: production of large numbers of fish, which requires producing fish adapted to an artificial environment, and production of fish that will survive and grow in the wild. Their mixed success at satisfying the second goal is best indicated by the gradual decline in most fisheries for anadromous species and rapid decline of many wild populations (Levin et al. 2000). It is also indicated by the listing of many salmonids as species threatened with extinction under the statutes of multiple countries.

Perhaps 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, but its dynamic geology and climate has resulted in the evolution of many distinctive inland forms, such as the three golden trout subspecies of the Sierra Nevada. The diversity of salmonids 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 region of the Pacific Ocean, one of the most productive ocean regions of the world (Moyle 2002). All this has resulted in hundreds of genetically distinct populations, although there are just eight recognized native species. For the purposes of this study, we recognize 32 salmonid taxa (genetically and ecologically distinct groups) in California, 21 of them anadromous, 11, non-anadromous (Table 1). These taxa are a combination of species, subspecies, and various units recognized by managers, characterized by genetics and/or life history patterns.

Table 1. California Salmonidae. Names in bold are listed as threatened or endangered by federal or state governments, usually both. Taxon type is how they are generally formally recognized today (ESU = Evolutionary Significant Unit, DPS = Distinct Population Segment, LHV = Life History Variant, an informally recognized DPS). Status is a 1-5 scale, where 1 = very rare, in danger of extinction soon and 5 = widespread (see Table 3 for details). Species with scores of 1 or 2 are regarded as in danger of extinction within the next 50-100 years. Certainty is the confidence we have in our status rating where 1 is low confidence and 4 is high confidence (See Table 3).

Taxon	**Taxon type**	**Endemism**	**Status**	**Certainty**
Klamath Mountains Province winter steelhead[1]	ESU/DPS	CA+OR	4	4
Klamath Mountains Province summer steelhead	LHV	CA	2	2
Northern California Coast winter steelhead	ESU/DPS	CA	4	4
Northern California Coast summer steelhead	LHV	CA	2	3
Central Valley steelhead	ESU/DPS	CA	3	2
Central Coast steelhead	ESU/DPS	CA	3	3
South/ Central coast steelhead	ESU/DPS	CA	2	3
Southern steelhead	ESU/DPS	CA	2	3
Coastal rainbow trout, *O. mykiss irideus*	Subspecies	Pacific coast	5	4
California golden trout, *O. m. aguabonita*	Subspecies	CA	2	4
Little Kern golden trout, *O. m. whitei*	Subspecies	CA	2	4
Kern River rainbow trout, *O. m. gilberti*	Subspecies	CA	2	4
McCloud redband trout, *O. m. stonei*	Subspecies	CA	2	3
Goose Lake redband trout, *O. m. subsp.*	Subspecies	CA+OR	3	2
Eagle Lake rainbow trout, *O. m. aquilarum*	Subspecies	CA	2	3
Lahontan cutthroat trout, *O. clarki henshawi*	Subspecies	CA+4 states	2	4
Paiute cutthroat trout, *O. c. seleneris*	Subspecies	CA	2	4
Coastal cutthroat trout, *O. c. clarki*	Subspecies	Pacific coast	3	2
Southern Oregon Northern California Coastal Chinook[2]	ESU	CA+OR	3	4
Klamath-Trinity fall Chinook	ESU	CA	3	4
Klamath-Trinity spring Chinook	LHV	CA	2	3
California coast Chinook	ESU	CA	2	3
Central Valley fall Chinook	ESU	CA	4	4

[1] All steelhead and coastal rainbow trout are treated as *O. m. irideus* after Behnke (2002)
[2] All Chinook salmon are treated as *O. tshawytscha*

Central Valley late fall Chinook	LHV	CA	2	3
Sacramento winter Chinook	ESU	CA	2	4
Central Valley spring Chinook	ESU	CA	2	4
Southern Oregon –Northern California coho[3]	ESU	CA+OR	2	4
Central California coast coho	ESU	CA	1	4
Pink salmon, *O. gorbuscha*	Species	Pacific coast	1	1
Chum salmon, *O. keta*	Species	Pacific coast	1	2
Bull trout, S*alvelinus confluentus*	Species	CA+OR?	Extinct	4
Mountain whitefish, *Prosopium williamsoni*	Species	Widespread	4	2

Many (15, 47%) of California's salmonids are already recognized as threatened, endangered, or extinct by state and federal governments (Table 1), but there is no overview of the status of this highly diverse and distinctive group of fishes in California. We undertook to produce an overview for the following reasons:

- California salmonids are characteristic of most of California's inland and coastal waters and they are exceptionally vulnerable to climate change, through rising temperatures and reduced summer flows. This study was partly designed to serve as a baseline for looking at the effects of climate change on aquatic systems in California by using one of its most valuable and charismatic groups of fishes as an indicator of ecosystem change.
- It is our perception that current lists of threatened and endangered species do not reflect the true condition of California salmonids.
- We wanted to evaluate the state of information on California salmonids by conducting a thorough search of the published and unpublished literature. Our perception from previous work was that most taxa were not being monitored as closely as they should be, even the listed forms.
- We wanted to alert both agencies and the public to the potential extent of the problem with declining salmonids and salmonid waters, in order to encourage strategic conservation, especially in the face of climate change.
- We wanted to called attention to the status of California salmonids as a problem of national significance. Because of its size and geographic complexity, California produces conditions similar to conditions throughout the range of salmonids, only its southern location and rapid urbanization means the problems presage those of other areas.

Our over-arching questions were: What is the population status of all California salmonids, both individually and collectively? What are major factors responsible for present status, especially of declining species?

METHODS

Our general approach to this overview was to:

[3] All coho salmon are treated as *O. kisutch*

1. Select the taxa for investigation.
2. Compile the existing literature on native California salmonids.
3. Produce detailed accounts of the biology and state of all 32 taxa. These are included as the main body of this report.
4. Evaluate the status of each taxon using a set of standard criteria.
5. Conduct an analysis of the overall status of California's salmonids and of the factors affecting status, using the information summarized in the species accounts.

Selection of taxa: For the most part, we used species, subspecies, Evolutionary Significant Units, or Distinct Population Segments already recognized by agencies. However, we also chose to recognize distinct life history variants of Chinook salmon and steelhead (i.e., spring Chinook salmon and summer steelhead). While these runs are not formally recognized by management agencies, they possess significant evolutionary and ecological differences from recognized forms. Although genetically similar to fall/winter runs in the same watersheds, the spring/summer forms are so distinctive in their life history, including the immature state of migrating adults and their behavior of holding through the summer in deep pools, that we thought they deserved separate consideration for conservation of life history diversity within the species.

Literature compilation: Much of the early literature had been compiled by Moyle et al. (1995) and Moyle (2002). However, we conducted extensive additional literature searches to (1) update information each taxon, (2) conduct detailed summaries for taxa not treated adequately in previous reviews, and (3) find 'gray' literature not reported in previous accounts, or unpublished in agency files. We also consulted with individuals familiar with each taxon to gain a better appreciation of local conditions and status, as well as to locate additional reports.

Production of taxon accounts: Each species has two accounts written for it. The main species accounts are literature reviews with extensive documentation and are posted on line (website). From these accounts, we produced the condensed versions for a non-technical audience. These condensed accounts necessarily leave out many important details, so the main accounts should be consulted as the basis for the information in the condensed accounts.

Each main account was drafted using a standard format (species description, taxonomic relationships, life history, abundance, factors affecting status, conservation, trends, and status). Each draft was reviewed and revised by all three co-authors, until we were reasonably satisfied with its accuracy. Most accounts were then sent out for review by one or more biologists familiar with the taxon and its status.

Evaluation of status: The status of each taxon was determined using six criteria (Table 2), all scored on a 1-5 scale where 1 was a low score and 5 was a high score. The six criteria were then averaged to produce an overall score for each species. A taxon scoring a 1or 2 was regarded as being in serious danger of extinction, while a taxon scoring a 4 or a 5 was regarded as reasonably secure for the immediate future. Supporting information for each score is found in the full species accounts. Because we recognized that the information on status was sketchy for some species, we also developed a reliability index for our scores, on a 1-4 scale, where 1 was unreliable because little peer-reviewed information was available and 4 was highly reliable, based on numerous accounts in the published and agency literature (Table 3).

Overall analyses: We summarized the status of all 32 taxa and of the each of the six criteria used to determine overall status of California salmonids and to compare the status of anadromous and non-anadromous taxa. These are presented graphically as histograms. This approach is similar to that Williams et al. (2007) used for their Conservation Success Index (CSI) for salmonids, in which twenty indicators were used to develop scores. Because we were trying to compare 31 taxa with very different life histories and variable amounts of information available on them, we only used six indicators (criteria), although they are similar to those used for the CSI.

Table 2. Metrics used to evaluate the status of California salmonids (score and criteria, based on a 1-5 scale)

1A. Inland fish area occupied

1. One watershed/stream system in California only
2. 2-3 watersheds/stream systems without fluvial connections to each other in California only
3. 1-3 watersheds/stream systems but populations present but depleted/rare outside California
4. 1-3 watersheds/stream systems in CA but widely distributed outside state.
5. More than three watersheds in CA and widely distributed and abundant outside state

1B. Anadromous fish area occupied

1. 0-1 apparent self-sustaining populations[4] in California today
2. 2-4 apparent self-sustaining populations in California today
3. 5-7 apparent self-sustaining populations in California today
4. 8-10 apparent self-sustaining populations in California today
5. More than 10 apparent self-sustaining populations in California today

2. Effective population size in CA

1. <50
2. 50-100
3. 100-1000
4. 1000-10,000
5. 10,000 +

3. Dependence on human intervention (hatcheries, water management, manual passage, barriers) for persistence in California

1. Captive broodstock program or similar extreme measures required to prevent extinction
2. Hatchery program using wild broodstock or similar measures required for persistence

[4] Equivalent of Functionally Independent Population (FIP) of NMFS. "Self-sustaining" means some evidence of natural reproduction through multiple generations in past 10-25 years.

3. Population persistence requires annual intervention (e.g., management of barriers, special flows, protection from poaching)
4. Persistence requires periodic habitat improvements (e.g., gravel augmentation, habitat restoration)
5. Self-sustaining population does not require intervention

4. Environmental tolerance (mainly physiological tolerances in relation to existing conditions, plus flexibility in reproduction [iterparity vs semelparity])[5]

1. Extremely narrow physiological tolerance during freshwater residence and/or, short lived, semelparous, determinant reproductive pattern (recruitment failure potential)
2. Narrow physiological tolerance during freshwater residence, and/or short lived, semelparous.
3. Moderate physiological tolerance during freshwater residence, and/or short lived, semelparous
4. Broad physiological tolerance in fresh water, and/or short lived, iteroparous
5. Physiological tolerance rarely an issue during freshwater residence, and/or long lived, iteroparous

5. Genetic risk/problems

1. Fragmentation, genetic drift, and isolation by distance, owing to very low levels of migration, and/or hybridization with hatchery fish are the major forces shaping genetic diversity within and among extant California populations
2. As above, but limited gene flow among populations reduces risk, although hybridization can continue to be a threat,
3. Moderately diverse genetically; hybridization risks low but present
4. Genetically diverse but limited gene flow to other populations.
5. Genetically diverse with gene flow to other populations (good metapopulation structure).

6. Vulnerability to climate change

1. Vulnerable in all watersheds inhabited
2. Vulnerable in most watersheds inhabited (possible refuges present)
3. Vulnerable in portions of watersheds inhabited (e.g., headwaters, lowermost reaches of coastal streams)
4. Low vulnerability due to location, cold water sources and/or active management
5. Not vulnerable to significant population loss due to climate change.

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[5] A species may have fairly broad physiological tolerances in the laboratory but if it lives in a region (e.g. southern California) where habitat conditions (e.g., temperature) naturally reach close to the limits of that tolerance, its environmental tolerance will be scored lower.

Table 3. Overall status categories and certainty of status categories used in evaluating the status of California salmonids.

Status categories

0 extinct
1. Highly vulnerable to extinction in native range in the next 50 years
2. Vulnerable to extinction in native range in next 100 years
3. No immediate extinction risk but populations declining or small and isolated
4. No extinction risk; populations are large and appear to be stable.
5. Populations expanding

Certainty of status categories

1. Status is based on educated guesses
2. Status is based on expert opinion using limited data
3. Status is based on reports found mainly in the in gray literature
4. Status is based on reports from multiple sources including peer reviewed literature

RESULTS

Of the 32 kinds of salmonids found in California, 20 (62%) are endemic to California (5 more are found only in Oregon in addition). One species (bull trout) is extirpated, three had their status scored as "1", 17 had their status scored as "2", six had their status scored as "3", four had their status scored as "4" and one (coastal rainbow trout) had its status scored at "5" (Figure 1, Table 1). By lumping the fish that scored "1" and "2" together and excluding the extirpated bull trout, 20 of 31 living taxa (65%) are in danger of extinction within the next century. Of the 21 anadromous taxa, 13 (62%) are in danger of extinction, while seven (78%) of the nine living inland taxa are in danger of extinction, Fifteen (75%) of the taxa in danger of extinction are endemic to California. All of the six metrics used to determine the status score contributed to the low scores of most species, although area occupied (Figure 2) and genetic risks (Figure 3) were perhaps the best predictor of endangerment, especially for inland taxa.

The histograms (Figures 2-7) indicate that species that already had a limited distribution are among the most vulnerable to extinction, a problem that is likely to exacerbated by competition and hybridization with non-native species or hatchery fish. Most species, however, still have large enough populations so that the impact of a random event (e.g., a landslide) on a small number of spawners is not a big concern, although it does affect a few species (Figure 4). While human intervention is essential to maintain species such as Eagle Lake rainbow trout, it is a secondary consideration for many taxa (Figure 5). Propagation and other actions probably increase population size but the taxa can often persist without the intervention, at least for a while. A key for persistence is habitat with water quality that is within the physiological limits of each species, which are surprisingly broad for some species (e.g., Lahontan cutthroat trout, Goose Lake redband trout) but many of the species are increasingly experiencing periods of poor water quality caused by human activities. In addition, climate change is already reducing the amount of suitable habitat (Figure 6) through increasing stream temperatures and reduced flows and is likely to be an increasing problem for California's salmonids in the future, as climate changes (Figure 7).

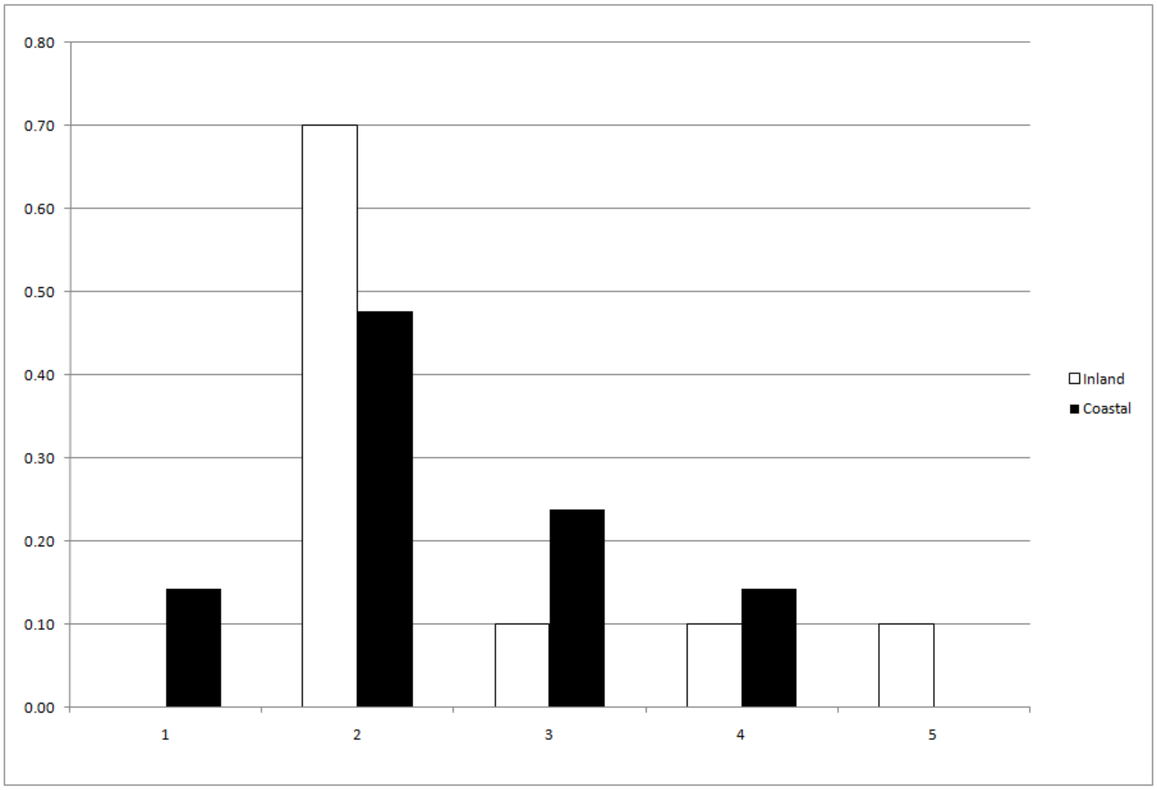


Figure 1. Status of existing California salmonids (N=31), where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). Status ranges from 1 (species in immediate danger to extinction) to 5 (species range stable or expanding). Status See Table 1 for individual species contributing to the total and Table 3 for explanation of categories. Categories 1 and 2 represent species in danger of extinction in the near future.

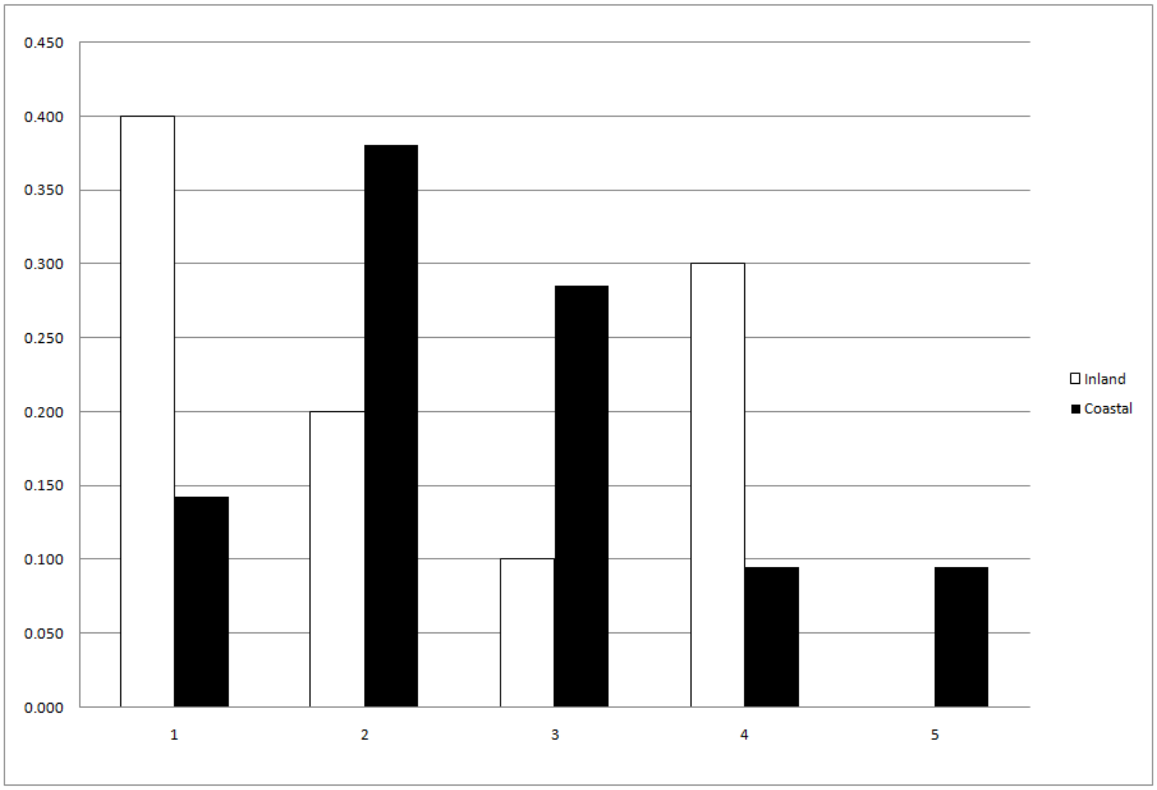


Figure 2. Area occupied by 31 kinds of California salmonids as a contributor to their status, where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). The factor is scored on a 1 to 5 scale, where 1 = small native range is a major contributor to decline; expansion would probably result in recovery of populations; 3 = range is a moderately important factor contributing to decline; expansion would result in some improvement in status; 5 = factor not a major cause of decline (species widely distributed). 2 and 4 are intermediate values.

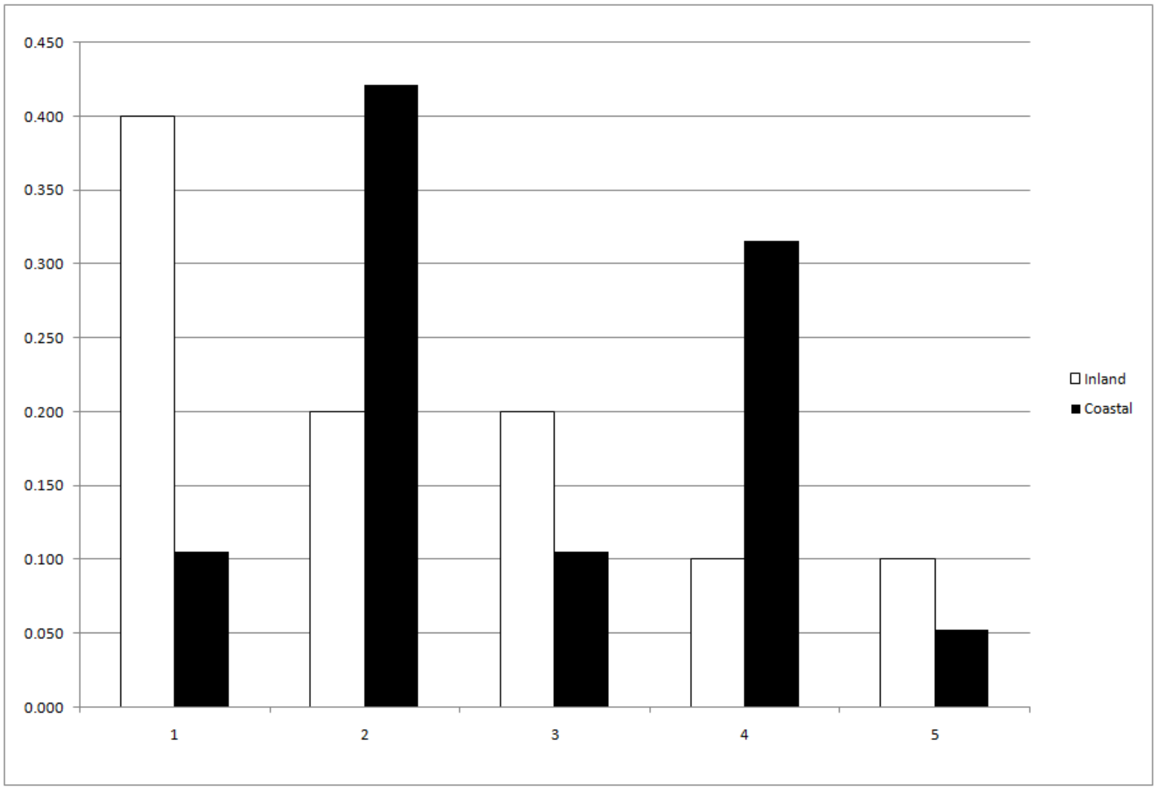


Figure 3. Genetic issues (hybridization, low population size etc.) for 31 kinds of California salmonids as a contributor to their status, where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). The factor is scored on a 1 to 5 scale, where 1 = major contributor to decline; removal/reversal would probably result in recovery of populations; 3 = moderately important factor contributing to decline; reversal would result in some improvement in population status; 5 = factor not a major cause of decline. 2 and 4 are intermediate values.

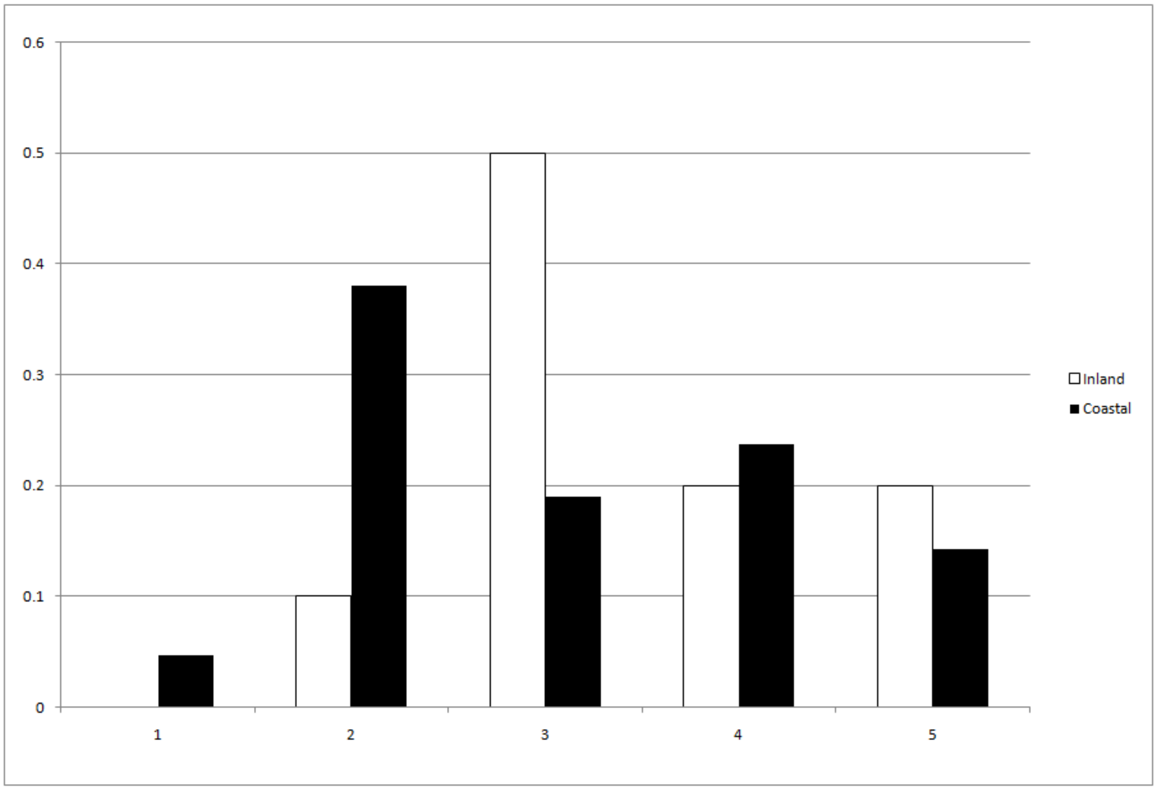


Figure 4. Small effective population size as a contributor to the status of the 31 kinds of California salmonids, where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). The factor is scored on a 1 to 5 scale, where 1 = major contributor to decline; removal/reversal would probably result in recovery of populations; 3 = moderately important factor contributing to decline; reversal would result in some improvement in population status; 5 = factor not a major cause of decline. 2 and 4 are intermediate values.

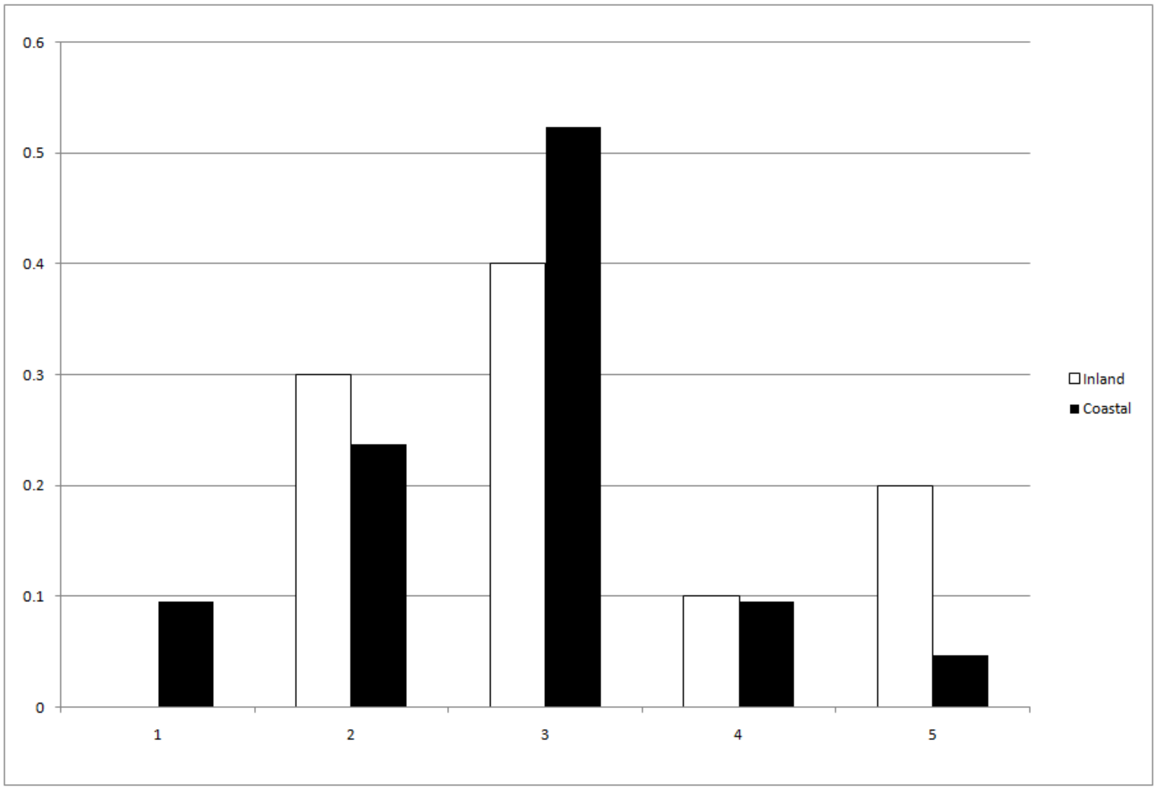


Figure 5. Dependence on human intervention as an contributor to the status of 31 kinds of California salmonids, where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). The factor is scored on a 1 to 5 scale, where 1 = human intervention essential for persistence; removal/reversal would probably result in extinction; 3 = human intervention moderately important factor contributing to persistence; reversal would result in some decline in population; 5 = intervention not needed or persistence. 2 and 4 are intermediate values.

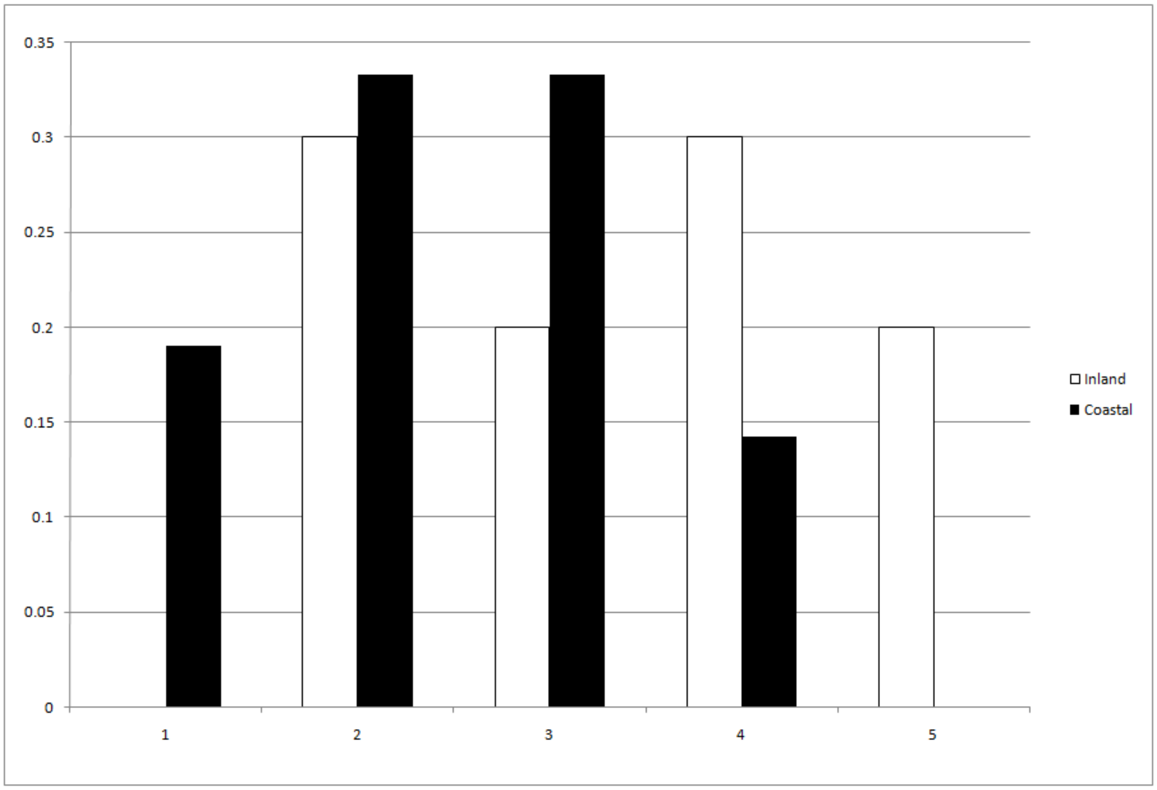


Figure 6. Physiological tolerance of environmental conditions likely to be encountered as a contributor to the status of 31 kinds of California salmonids, where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). The factor is scored on a 1 to 5 scale, where 1 = low physiological tolerance is a major contributor to decline, i.e. species has low tolerance of anthropogenic change or environment has been altered to point where environmental limits are being reached; improvement in conditions would probably result in recovery of populations; 3 = physiological tolerance is a moderately important factor contributing to decline; improvements would result in some increase in populations; 5 = physiological tolerance not a major cause of decline. 2 and 4 are intermediate values.

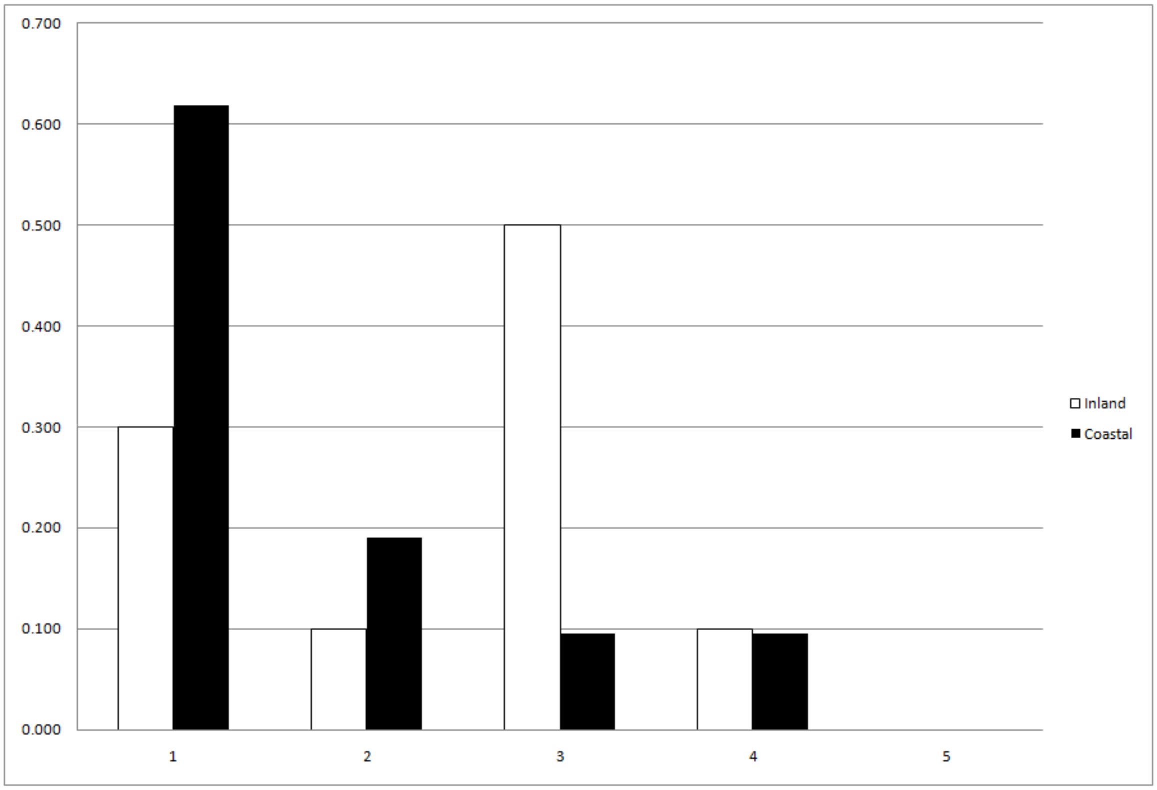


Figure 7. Likelihood that climate change will be a contributor to the status of 31 kinds of California salmonids, where white bars represent inland taxa and black bars represent anadromous taxa, expressed as proportion of total for each category (inland =9, anadromous =22). The factor is scored on a 1 to 5 scale, where 1 indicates that climate change is or will be a major contributor to decline and improvement in conditions would probably result in recovery of populations; 3 indicates that climate change is likely to be moderately important factor contributing to decline; improvements would result in some increase in populations; 5 = climate change is not likely to be a major factor in decline. 2 and 4 are intermediate values.

DISCUSSION

Our analysis of the status of California salmonids tells us that most taxa are declining rapidly and, if present trends continue, 65% (20 taxa) will be gone within a 100 years, probably within 50 years. Seventy-five percent of these endangered taxa are found only in California, so fit well the definition of Waples et al. (2007) for species likely to qualify for listing as threatened or endangered, if they are not already listed. Seventy-five percent of these endangered taxa are found only in California. While each salmonid has its unique problems, they all are basically in decline because of increased competition with humans for resources, mainly water. The cumulative impact of degraded habitats and biological threats (e.g., alien species) do not allow salmonid populations to rebound as readily in response to 'natural' long term physical stresses, such as extended drought. Climate change is exacerbating the problem because it ultimately will reduce the amount of cold water habitat that salmonids require. On the bright side, only one taxon, bull trout, has gone extinct so far and many have shown remarkable resilience in the face of human changes to their streams.

There are 13 different taxa of anadromous salmonids facing extinction. The two species most likely to go extinct in California are pink salmon and chum salmon, species that have never been particularly common in California although they were a recognized part of fish fauna in the 19th and 20th centuries and contributed to historic salmon harvests. However, close on the extinction heels of these two species are two ESUs of coho salmon, which numbered in the hundreds of thousands in California only 50-60 years ago and were significant players in the state's coastal stream and ocean ecosystems (Moyle 2002). Other taxa facing extinction are the two groups of summer steelhead and the two groups of spring Chinook salmon; both types of fish are unusually vulnerable because their populations are confined to a few small headwater streams into which they migrate to spend the summer before spawning. This makes their populations exceptionally vulnerable to a wide array of factors, from poaching to climate change.

Nevertheless, some salmonids will persist in California over the next century and nine anadromous salmonids were found not to be in danger of extinction. However, even these salmonids are in decline, so fisheries for them are probably not sustainable. Remarkably, all coastal salmon and steelhead pretty much still occupy their extensive native ranges, albeit in decreased numbers. However, over the next century, most of the populations will persist only with heroic efforts to protect streams all along the California coast.

Seven of the nine remaining resident salmonids are in trouble, mostly because they are endemic to a few streams in very small areas, such as the three golden trouts of the Upper Kern River basin. In these isolated areas, they are exceptionally vulnerable to hybridization with introduced salmonids (mainly rainbow trout) and well as grazing, logging, and other factors. They could easily follow bull trout into extinction in the state due to localized effects.

Still, it is astonishing to think that most of California's salmonids still occupy, if in a fragmented manner, most of their native ranges. This says a great deal about their resilience in the face of the ever increasing demand of humans on the resources they need to survive, especially water and diverse habitat. Saving California's native salmonids will not be easy, but by doing so we not only protect a unique biological heritage but the ecosystem services, such as clean water, that salmonid streams provide. Saving our salmonid heritage will not be easy and will be expensive, but here are few more general actions to take:

1. Develop and implement individualized conservation strategies for all 31 extant taxa that have as their basic goal the maintenance of self-sustaining populations through the indefinite future throughout their range. The strategies must take into account climate change as well as increasing water demand and changing land use. An initial step in the strategy would be to evaluate all species that scored 1 and 2 in this report for formal listing as threatened or endangered species.
2. Provide immediate additional protection to 'salmon strongholds' where salmonid diversity is high and habitat conditions are still reasonably good, such as the Smith River and Blue Creek. This means reducing the human footprint on the watershed as much as possible by managing the streams first and foremost for fish.
3. Develop a statewide hatchery policy that has as its first goal protection of wild populations of fish, rather than enhancing fisheries. At the very least, all hatchery fish should be marked and mark-selective fisheries instituted.
4. Develop a salmonid awareness program for the public and public schools that strives to educate Californians about the importance, both cultural and economic that salmon, steelhead, and trout have in California, and about the unique challenges and responsibilities that come from coexisting with species at their southern-most limit.
5. Develop a statewide research and monitoring program for salmonids and other cold-water fishes, funded by both state and federal agencies, with status reviews required at least once every 5 years.
6. Chose a few high-profile salmonid rivers in each part of the state for focused restoration, such as the Shasta River, Lagunitas Creek, Battle Creek, and the Santa Margarita River.
7. Continue and expand the work of citizen watershed groups to enhance and protect all California streams.
8. Enforce and strengthen existing laws and regulations, tied to the Clean Water Act, the Endangered Species Act, State Forestry Practice Rules, the Fish and Game Code, and similar measures to increase protection for salmonids and their rivers.
9. Fully fund ongoing efforts to restore the San Joaquin River for salmon to create a positive example of large scale recovery of a river system.
10. Develop creative ways to fund salmonid protection, such as a surcharge on all beverages (extra for bottled water), water bills, and water transactions.
11. Develop restoration projects for critical life stages that also benefit other conservation goals such as setback levees to open up floodplain habitat for juvenile rearing (a habitat in critically short supply) while simultaneously improving flood control and human safety.

Ultimately, as Lackey et al. (2006) bluntly point out, maintaining fisheries for each species will take a fairly radical restructuring of the way our society works and treats the resources of California and elsewhere. If present trends continue, California will have only 'museum' populations or runs of most salmonids, maintained with very high effort for display purposes (to remind people what has been lost). Truly wild salmon and trout will persist in the long run only if the human population levels out or decreases, the per capita demand for water declines dramatically, and we as a species learn to live lighter on the land. Until that time, the less dramatic measures envisioned above will have to do, as the bare minimum required to keep the populations going through the hard times ahead.

KLAMATH MOUNTAINS PROVINCE WINTER STEELHEAD
Oncorhynchus mykiss

Description: Klamath Mountains Province (KMP) winter steelhead are similar to other steelhead in their characteristics (see North Coast winter steelhead for a description). They are separated from other steelhead mainly through genetics and life history traits. They differ from summer steelhead in the Klamath Mountains Province steelhead ESU mainly in their entry during the winter into fresh water as mature, rather than immature, fish and in various behavioral traits.

Taxonomic Relationships: For general relationships, see North Coast winter steelhead account. The KMP winter steelhead are treated separately from summer steelhead that are part of the same ESU because there are low levels of genetic differentiation between the two runs and they are distinctive in their behavior and reproductive biology. Winter steelhead appear to contain two genetically distinct populations (Papa et al. 2007). A recent genetic study by Pearse et al. (2007) determined that genetic structuring was primarily at the individual site level, with each population being most similar to adjacent winter steelhead populations. Collections of steelhead in the lower Klamath River (Turwar, Blue, Pecwan, Cappell, and Tully Creeks) showed limited gene flow among these sites and sites above the Trinity River confluence. Populations in Blue and Hunter Creeks grouped more closely to other coastal KMP populations from the Smith River and Wilson Creek. Populations in the middle and upper-middle Klamath regions clustered closely together. Populations in the Shasta and Scott River clustered with Iron Gate Hatchery fish and were genetically different from other steelhead in the middle Klamath region. Trinity River Hatchery steelhead clustered within the relatively homogeneous group of collections from the middle and upper middle Klamath regions, presumably due to decades of egg transfer from this area into the Trinity River Hatchery (Busby et al. 1994). The only Trinity River fish used in the Pearse et al. (2007) study were from Horse Linto Creek and appeared to group with the lower Klamath collections.

Life History: KMP winter steelhead mature in the ocean and are the predominant steelhead in the Klamath River. Fall-run steelhead are generally included with winter-run steelhead because it is not clear that separate runs exist (but see Table 1). These fish enter the river as sexually mature adults in September-March and spawn shortly after reaching spawning grounds (Busby et al. 1996). A peak in spawning occurs by March. The overlap in migration and spawning periods make differentiating winter steelhead from the stream-maturing summer steelhead difficult (see KMP summer steelhead account). Winter steelhead as defined here are part of a complex of life history patterns for steelhead in the KMP region (Table 1).

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	

Table 1. Different classifications for Klamath Mountain Province steelhead based on run timing.

The early life history of winter steelhead in the Klamath and Trinity River basins is fairly well understood. Steelhead fry in the Trinity River emerge starting in April and begin downstream emigration in May, before reaching a peak in June and July (Moffett and Smith 1950). Newly emerged steelhead initially move into the shallow, protected margins of streams (Moyle 2002). Steelhead are territorial and exhibit aggressive behavior to establish territories (Shapovalov and Taft 1954) in or below riffles, where food production is greatest. Moffett and Smith (1950) found steelhead fry (individuals not yet surviving through a winter) 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, Moffett and Smith (1950) observed increased downstream movement. Steelhead parr showed the greatest freshwater movement towards the end of their first year and spent their second year inhabiting the mainstem. 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). Kesner and Barnhart (1972) determined that Klamath steelhead rearing in fresh water for longer periods made their seaward migration more quickly. Klamath River basin steelhead remain in the ocean for one to three years before returning to spawn. Their ocean migration patterns are unknown.

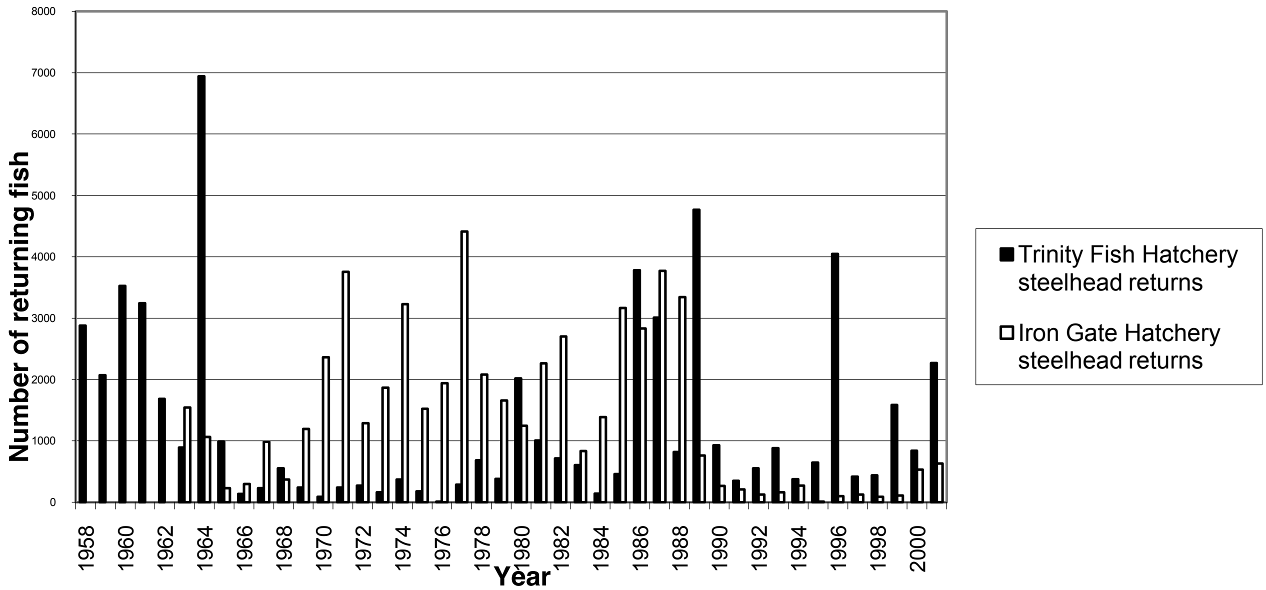
The presence of “half-pounder” steelhead is a distinguishing life history trait of steelhead found in the Klamath Mountains Province ESU. Half-pounder steelhead are subadult individuals that have spent 2-4 months in the Klamath estuary or inshore marine environments before returning to the river to overwinter. They overwinter in the lower and mid-Klamath regions before returning to the ocean the following spring. The presence of half-pound fish is uncommon above Seiad Valley (Kesner and Barnhardt 1972). There was a negative linear relationship between rates of half-pounder migration and first-time spawning size. The occurrence of half-pounders was greater in spawning winter steelhead from the mid-Klamath region tributaries (86-100%) when compared to the Trinity River (32-80%). The lowest occurrence of half-pounders was from Lower Klamath River winter steelhead (17%), which also demonstrated the greatest first-year growth rate (Hopelain 1998). The proportion of these fish that become ocean-maturing steelhead is not known.

Habitat Requirements: Habitat requirements of KMP winter steelhead are basically the same as Northern California Coastal winter steelhead. Due to their migration and spawning period coinciding with the period of greatest flows, winter steelhead often ascend into smaller tributaries not accessible during low-flow periods or by other salmonids. These include streams in medium sized watersheds impacted by sedimentation, where their confluences are often not passable earlier in the fall. They also attain headwater reaches of lower-order tributaries, in which flows are too low during early fall for access by large fish.

Distribution: KMP winter steelhead range includes coastal rivers and creeks throughout the Klamath and Trinity basins and streams north of the mouth of the Klamath River to the Elk River near Port Orford, Oregon. Their range encompasses the Smith River in California and the Rogue River in Oregon. In the Klamath River, they currently ascend as high as Iron Gate Dam although it is likely they historically ascended into tributaries to Upper Klamath Lake (Hamilton et al. 2005). In the Trinity River, their upstream access is blocked by Lewiston Dam (Moffett and Smith 1950).

Abundance: Only sketchy data are available to evaluate wild Klamath River steelhead population trends. The *California Fish and Wildlife Plan* (CDFG 1965) estimated a Klamath-Trinity basin-wide annual run size of 283,000 adult steelhead (spawning escapement + harvest). Busby et al. (1994) reported winter steelhead runs in the basin to be 222,000 during the 1960s. Numbers declined to 87,000-181,000 adult spawners between 1977-1978 and 1982-1983. Based on creel and gill net harvest data (Hopelain 2001), the winter steelhead population was estimated at 10,000-30,000 adults annually in the early 1980s in the Klamath River. The Trinity River steelhead run was estimated to be in the same range, though more variable, and ranged from 7,833 to 37,276 adults (average from 8 years was 15,185) during the 1980s. Returns to the Iron Gate hatchery are highly variable and have been distinctly depressed in recent years (Figure 1). Trinity River hatchery returns have been on the increase since 2000, with some of the highest hatchery returns recorded in the last several years. In the Smith River, spawning escapement was estimated to be approximately 30,000 adult steelhead during the 1960s, but there are no subsequent drainage-wide estimates.

Figure 1. Historical steelhead returns at two hatcheries in the Klamath River basin, 1958-2001 (from Hopelain 2001).



Factors affecting status: Populations of KMP winter steelhead are large enough to support sport fisheries but appear to be in a long-term decline in most rivers and Klamath-Trinity populations are increasingly supported by hatcheries. The general decline of winter steelhead likely has multiple causes (see Northern California coastal winter steelhead account for a more general discussion of the issues). The main factors impacting steelhead include 1) dams, 2) diversions, 3) logging, 4) agriculture, 4) hatcheries, and 5) harvest.

Dams: Like other large river systems in California, the Klamath-Trinity system has been heavily dammed to provide water for human use. Three dams that particularly affect KMP steelhead runs (all part of larger projects) are Iron Gate, Dwinnell, and Lewiston dams.

Iron Gate Dam is the lowermost dam on the Klamath River and it is part of a chain of hydropower dams that have altered flows in the Klamath River in combination with operations of the USBR's Klamath Project, which diverts water to irrigate farmland in the upper Klamath Basin (NRC 2004). The dams have served as barriers to upstream migration ever since Copco Dam was constructed in 1917. A primary impact to steelhead has been the elimination of access to historic spawning and rearing areas upstream of the dams. Another combined impact of the dams and the Klamath Project on Klamath River steelhead has been the alteration of natural flow regimes below Iron Gate Dam. Basically, mainstem flow peaks have been shifted a month or more earlier than historic peaks and summer flows have been reduced. The lower flows result in increases in summer temperatures of the river, although the water coming out of Upper Klamath Lake in summer is warm in any case, so that releases from Iron Gate Dam in August are often above 22°C (NRC 2004). The water warms up further as it moves downstream, due to absorption of heat from the warm summer air, so that mainstem water temperatures can reach 24-26°C during the day for extended reaches. Because food is abundant in the Klamath River, juvenile steelhead can persist under these conditions if water temperatures cool a few degrees at night or if there are cool water refuges available at the mouths of cold tributaries (see bioenergetics discussion in the SONCC coho salmon account). In general, the warm temperatures are stressful for steelhead and other juvenile salmonids in the mainstem of the Klamath River by reducing available habitat for juvenile steelhead.

Dwinell Dam, constructed in 1928, blocks access to 30+ km of high-quality habitat in the upper Shasta River, a tributary to the Klamath. In combination with seven small diversion dams and other diversions, Dwinell Dam significantly reduces flow in the lower Shasta River. The dam also changes the hydrograph, eliminating peak flows (NRC 2004). The effects of the dam exacerbate the effects of other diversions downstream. As a result, daily minimum temperatures in the river are usually above 20°C in summer and daily maxima are usually above 22-24°C, stressful for steelhead. Thus, while the lower Shasta River still supports steelhead spawning and rearing, the quality and quantity of habitat is greatly reduced.

Lewiston Dam on Trinity River, which closed in 1963, blocks access to over 170 km of streams in the upper watershed. In combination with Trinity Dam just upstream, it dramatically reduced flows in the river and changed the hydrograph, greatly reducing habitat available for steelhead and other fishes in the mainstem river. In 1984, the Trinity River Restoration Program was initiated to examine the benefits of restoring 25 to 48 percent of the average annual inflow to the Trinity River. In 2003, a flow regime with lower spring and much 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. Recently, 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: Stream flows in many Klamath tributaries, as well as other streams in the KMP winter steelhead range, have been reduced by domestic and agricultural diversions, either directly or indirectly by pumping from wells adjacent to the streams. In many streams, this may be the biggest factor steelhead affecting steelhead numbers. In the Scott and Shasta Rivers, diversions have major impacts on steelhead and other fishes by reducing flows, with consequent reduction in habitat and increases in temperatures, as well as by returning 'excess' water to the river (NRC 2004). This 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: Much of the Klamath Basin is covered with public and private forest lands, which has been heavily logged for the past century. The effects on streams of logging, and its accompanying road-building, are particularly severe in the basin because the steep slopes of the mountains are naturally unstable and subject to landslides and mass wasting (NRC 2004). The effects of logging are especially severe in tributaries where steelhead concentrate for spawning and rearing. The degradation of this habitat and potential impacts to juvenile salmonid production is well documented (Borok and Jong 1997, Jong 1997, Ricker 1997). 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. Where habitat is severely altered, juvenile production greatly decreases due to loss of cover, filling in of pools, and increased temperatures (Burns 1972). In addition, in many streams, improperly constructed culverts are barriers to upstream spawning and rearing areas.

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 are still reducing the ability of habitat to produce steelhead.

Agriculture: Agriculture, especially irrigated pasture and alfalfa for livestock grazing, impacts streams throughout the Klamath and Trinity basins through both runoff of agricultural constituents and sedimentation. Impacts are usually increased by diversions of water as well (see above). The Shasta and Scott valleys have been identified as two regions where improved agricultural practices could dramatically increase salmon and steelhead populations (NRC 2004)

Hatcheries: Two hatcheries are currently operated by the California Department of Fish and Game (CDFG) as mitigation for lost habitat above Iron Gate and Lewiston Dams. While hatchery production has primarily relied upon native brood stock, there have been numerous documented transfers of fish from outside the basin. Prior to 1973, transfers came from the Sacramento, Willamette, Mad and Eel Rivers (Busby et al. 1996). Because the length of freshwater occupancy of juvenile Klamath River steelhead is long, risk to wild fish is potentially increased by competition and predation from hatchery fish. About 1,000,000 smolts per year are produced by the two hatcheries (NRC 2004). In 2003, 191,000 steelhead yearlings were released from Iron Gate hatchery using a volitional release that started on March 28. About half the fish moved downstream on their own, while the other half were released by CDFG on May 9 (K. Rushton, pers. comm.). Historic returns of steelhead to both hatcheries are shown in Figure 2. The behavioral 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).

Harvest: A sport fishery for Klamath River steelhead and other salmonids provides benefits to the local Klamath River economy. The net annual economic benefit of steelhead in the Klamath River is over 12 million dollars per year (McEwan and Jackson 1996). Currently, sport fishing regulations prohibit take of wild winter steelhead and do not allow fishing of summer steelhead, although the fishing season for Chinook salmon in the Klamath and Trinity Rivers overlaps with summer steelhead distributions, thus subjecting the latter fish to possible fishing pressure. The effects of the fishery on steelhead populations are not known but are assumed to be small compared to other factors.

Conservation: Key elements of the *Steelhead Restoration and Management Plan for California* (McEwan and Jackson 1996) for the Klamath River include:

1. Increasing naturally produced stocks of steelhead. The plan recognizes the importance of protecting selected subbasins where natural processes take precedence over human use, in order to create refuges to protect steelhead distribution and diversity.
2. Improving flows below Iron Gate and Lewiston Dams. The latter has already taken place to a certain extent and flows below Iron Gate depend on the outcome

of the Federal Energy Regulatory Commission's relicensing of the four hydropower dams, including Iron Gate Dam.

3. Restoring favorable instream conditions to benefit multiple species and desired ecosystem function instead of single species. This concept recognizes that steelhead in the Klamath Basin do well when part of a complex fish and invertebrate community that includes other salmonids. A good first step would be the creation of a basin-wide restoration program involving stakeholders, managers, and policymakers from the upper and lower basins. Such a group could identify physical and hydrological processes and other habitat conditions that are necessary for conserving the aquatic communities of the Klamath basin.

Watersheds identified by McEwan and Jackson (1996) as high priority for stream restoration to benefit steelhead included the South Fork of the Trinity, Scott, and Shasta rivers. Many subbasins of the Klamath River are predominantly within public ownership and were designated key watersheds as part of the Northwest Forest Plan. Further steps will be necessary on private lands to restore functioning aquatic habitats and steelhead populations. Already, fish and watershed restoration projects bring money into rural parts of the Klamath River basin where the economy can no longer depend on timber and mining dollars. However, without increased flows and suitable water quality (i.e., cool and sediment-free), the effectiveness of restoration is marginalized (Wu et al. 2000). Great potential exists for steelhead to increase in value as a trophy fishery on the Klamath and this should bring additional local economic benefits to local communities. The importance of steelhead and a healthy Klamath River to the economy of Klamath basin communities has yet to be fully realized. This applies to other streams inhabited by winter steelhead as well.

In recent years, significant funding has been directed towards treating many of the detrimental impacts that road building and logging have had on KMP steelhead habitats. Additionally, protection efforts have increased by private landowners that graze livestock in riparian areas and divert water for agriculture. 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 watershed-level activities that need to happen.

Another need is for more research on the complex needs of KMP steelhead, especially in the Klamath Basin. Managers would benefit from a better understanding of the physical and biological cues that lead to the diverse migration patterns. Determination of survival and escapement rates for wild steelhead is essential understanding the viability and persistence of individual populations. For an accurate assessment of all populations, monitoring must increase within the basin. Additional information regarding the genetics, ecology, and behavior of KMP steelhead will contribute to a broader recognition of their rivers as an important and productive aquatic systems.

The river with the highest degree of protection for KMP steelhead is the Smith River, Del Norte County, the largest river in California without a major dam. In 1990, the Smith River National Recreation Area Act by signed by President George H. W. Bush as

Public Law 101-612, which provides some protection on paper for the river. The local conservation group, the Smith River Alliance, has employed a conservation strategy of acquiring large chunks of land to protect important watersheds, such as Goose Creek and Mill Creek. This is a valuable mechanism for conserving steelhead sanctuaries.

Trends:

Short term: KMP winter steelhead are abundant enough to continue to support a fishery although they appear to be in slow decline at the present time. This is especially true of the Klamath Basin where present numbers are far below population estimates from even two decades ago. If restoration efforts continue and flows improve in Klamath Basin rivers, it is possible to optimistic about the health of KMP steelhead populations in California in the next 15-20 years. Trinity River Restoration Program actions, such as improved flows, manipulation of shallow edge habitats, and removal of barriers, will benefit Trinity River steelhead populations. Also, the Smith River remains relatively undisturbed, with major conservation activities taking place within the watershed, so it is likely to remain a strong refuge for KMP steelhead regardless of what happens in other watersheds, especially in the upper Klamath Basin.

Long term: The long-term trends in KMP winter steelhead are downwards, which is evident despite the relatively poor records that are available, especially in the Klamath Basin. While KMP winter steelhead populations in the Trinity and Smith rivers nevertheless appear healthy, the downward trends may continue unless even more effort is made to protect water and public lands in the basin. These basins are both National Wild and Scenic Rivers and this designation should protect water quality and quantity necessary for strong runs of KMP steelhead. However, the impacts of the Trinity River Hatchery steelhead on wild steelhead need to be better understood. Steelhead in the Klamath River region face increased challenges due to climate change and the potential for flows to remain impaired in this area. Although tributaries (e.g., Salmon River, Dillon Creek, Clear Creek, Elk Creek) may provide healthy spawning and nursery areas, water quality and quantity in the mainstem may be seasonally too poor to for provide connectivity between these locations and for rearing habitat of larger juveniles. Numbers could increase, however, if connections were re-established with the upper Klamath Basin, through dam removal or provision of passage (fish ladders, etc.).

Status: 4. There is no immediate extinction risk for KMP winter steelhead, although some populations will likely decline further or even be extirpated under current management trends. The KMP summer steelhead, however, has a high risk of extinction (see separate account). The entire ESU was first identified as "not warranted" for listing by NMFS in March 1998. A court decision in 2000 overturned this decision, finding that the agency relied too heavily on the expected effects of future conservation efforts. A final decision was reached on April 4, 2001 and the listing of Klamath Mountain Province steelhead ESU under the ESA was again determined to be not warranted. Klamath Mountain Province steelhead are listed by the US Forest Service Pacific Southwest Region as a Sensitive Species and are managed by CDFG for sport fishing.

Metric	Score	Justification
Area occupied	5	Widely distributed
Effective population size	5	Wild populations in Klamath seem to be large
Intervention dependence	4	Wild populations may require protection from hatchery fish
Tolerance	4	Steelhead are physiologically tolerant and have flexible life history
Genetic risk	4	Some risk from hatchery fish in Klamath
Climate change	4	More opportunities to respond than most salmonids
Average	4.3	26/6
Certainty (1-4)	4	Well documented population

Table 1. Metrics for determining the status of KMP winter steelhead, where 1 is poor value and 5 is excellent.

KLAMATH MOUNTAINS PROVINCE SUMMER STEELHEAD
Oncorhynchus mykiss

Description: Klamath Mountains Province (KMP) summer steelhead are anadromous rainbow trout that return to freshwater streams in the Klamath Mountains Province in April through June. Summer steelhead in general are distinguishable from other steelhead by (1) time of migration (Roelofs 1983), (2) the immature state of gonads at migration (Shapovalov and Taft 1954), and (3) location of spawning (Everest 1973, Roelofs 1983). Attempts to distinguish juvenile summer and winter steelhead and resident juvenile rainbow trout using otolith nuclei widths, scale circuli densities, and visceral fat content have only been partially successful (Rybock et al. 1975, Winter 1987) primarily because of difficulties in setting up rigidly controlled experiments (Winter 1987). Summer steelhead are similar in appearance to the more common winter steelhead (see description under Northern California coastal winter steelhead). In addition, they have an apparent "half-pounder" run of non-reproductive steelhead, which return to fresh water after the first summer in estuarine and coastal waters but return to sea after a few months in fresh water.

Taxonomic Relationships: For general relationships of steelhead, see Northern California coastal winter steelhead account. Genetic studies of the KMP steelhead Distinct Population Segement (DPS) indicate that KMP summer steelhead are more closely related to KMP winter steelhead than to summer steelhead elsewhere (Reisenbichler et al. 1992). Recent genetic studies of summer and winter steelhead show a low level of differentiation between them over multiple years, but also demonstrated there are likely greater levels of differentiation between spatially isolated reproductive populations (Papa et al. 2007, Pearse et al. 2007). NMFS does not classify Klamath River basin steelhead "races" based on run-timing of adults, but instead recognizes two distinct reproductive *ecotypes* of steelhead in the Klamath Basin based upon their reproductive biology and freshwater spawning strategy (Busby et al. 1996, Table 1). These two reproductive ecotypes are largely summer and winter steelhead. In the future, KMP summer steelhead could be recognized as a distinct DPS and managed separately from winter steelhead. See Box 1 in the Northern California coastal steelhead account for a discussion of this distinction.

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	

Table 1. Classification of different run-timings and reproductive ecotypes of steelhead found in the Klamath River basin.

It is possible that the runs of steelhead that made it up into the upper Klamath Basin before the construction of Copco Dam were KMP summer steelhead. The other alternative is that the upper basin steelhead were anadromous or fluvial redband trout (*O. mykiss newberri*), which currently persist in the upper basin. The genetic relationship of

KMP steelhead to these redband trout, which show migratory and resident life history variations, has not been determined.

Life History: Stream-maturing (summer) steelhead are uncommon, but continue to persist in subbasins of the Klamath Mountains Province and are distinguishable from the more common winter steelhead on the basis of adult migration and the morphological and physiological differences that result from it. Summer steelhead in California typically enter their 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 apparently ascend into the summer holding areas during a similar period. The holding areas are typically deep pools in canyon reaches of stream with some subsurface flow to keep temperatures cool.

Summer steelhead enter their rivers when still sexually immature and 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. The peak of spawning in the Trinity River is February, earlier than winter steelhead, which peak in March. On the Rogue River, Oregon, spawning begins in late December and peaks in January (Roelofs 1983) and this early spawning is apparently found throughout the Klamath Mountains Province. In Rogue River tributaries, spawning begins in late December, peaks in late January, and tapers off by March. Fecundity has been estimated at 2,000 to 3,000 eggs per female. In the Eel River system, only 9% of returning summer steelhead are repeat spawners (Jones and Ekman 1980) while in the Klamath drainage are 40 to 64% of the total (Hopelain 1998). 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).

Half-pounders (see KMP winter steelhead account) are not traditionally considered to be part of summer steelhead life history because they do not mature or reproduce while in the river. However, annual surveys of summer steelhead in late summer in the Salmon, New, and South Fork Trinity rivers generally encounter apparent half-pounders (Israel and Moyle, pers. observation). Frequently, the half-pounders outnumber adult steelhead during these surveys. The presence of half-pounders over-summering with adult summer steelhead is not typically characterized in the literature (Kesner and Barnhardt 1972, Hopelain 1998). It is possible that these fish are jack males.

Traditionally, half-pounders are smaller fish (25-35 cm) that return to the river in late summer and early fall (between late August and early October); they are subadult individuals who have spent only 2-4 months in the Klamath estuary or near shore environments before returning to the river to over-winter and forage in the lower and mid-Klamath river reaches (Kesner and Barnhart 1972). They return to the ocean the following spring. The presence of half-pound fish is uncommon above Seiad Valley

(Hopelain 1998) and summer steelhead are also not found in tributaries above this location. Thus it is possible that the 'standard' half-pounders are partially summer steelhead.

Habitat Requirements: Juvenile habitat requirements of summer 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. Adult summer steelhead in the New River occupy confluence pools 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) did not find a significant positive relationship between adult fish density and mean hourly pool temperature. 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 (Nakamoto 1994, Baigun 2003). Cover was used by 99% of the summer steelhead observed during the day on the New River; bedrock ledges and boulders were used more frequently than depths of greater than 1m or shade from vegetation (Nakamoto 1994).

Spawning habitat for summer steelhead is variable and their consequent temporal and spatial isolation from other steelhead runs may maintain low levels of genetic differentiation from winter steelhead (Barnhart 1986, Papa in press). Summer steelhead often spawn in intermittent streams, from which the juvenile emigrate into perennial streams soon after hatch (Everest 1973). In the Rogue River, Oregon, summer steelhead spawn in small headwater streams with relatively low (<50 CFS) winter flows (Roelofs 1983). 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.

Distribution: The KMP steelhead range includes the Klamath and Trinity rivers and other streams north to the Elk River near Port Orford, Oregon. Their range 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, as does the Rogue River, Oregon.

Abundance: We know little about the past abundance of these fish; quantitative records of summer steelhead numbers exist only for the recent few decades (Roelofs 1983). Given the habitat available, however, it is likely that summer steelhead in California today represent only a small fraction of their original numbers.

Summer steelhead populations have declined precipitously in the past 30-40 years. Snorkeling counts for summer steelhead are prone to numerous problems such as counting half-pounders as adult steelhead, incomplete spatial surveys, observational bias by surveyors, and low water clarity from suction gold dredging. Thus survey numbers likely represent the minimum fish present and so are still useful for trend analysis. However, the majority of estimates for California populations have been less than 100 fish each for the past decade (Appendix 1). In 1989-1991, the three-year average exceeded 500 fish in only two KMP streams: North Fork Trinity River and New River, which also had more than 500 fish in 1999-2001. Out of fifteen summer steelhead populations in the Klamath-Trinity basins, ten averaged <100 fish annually and five populations averaged <20 fish each for the years they were surveyed. Because the "effective" (breeding) population sizes are probably less than the actual counts, many populations may be close to or below the minimum size needed for long-term survival (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. 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 status of each major population is a follows:

Mainstem Trinity River: Moffett and Smith (1950) indicate that summer steelhead were common in the 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 in this section is not known.

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 conditional based upon sufficient flows, while temperature may be limiting in the reaches closest to the mainstem Trinity River confluence (Everest 1997). Given that this stream has been heavily altered by mining, it is likely that runs were much higher in the past (Roelofs 1983). Canyon Creek, a tributary close to the North Fork Trinity River, continues to see small numbers of summer steelhead and the average estimated adult population was 19 for 24 surveys over 30 years.

South Fork Trinity River: There is no historical information on summer steelhead in this stream. Recent counts were as lowed as 34 fish, although in 2006 and 2007 more than 100 fish were observed. Recent surveys on the South Fork Trinity River show summer steelhead were less common than half-pounder steelhead, although similarly distributed (Garrison 2002).

New River: This tributary to the Trinity River is the largest summer steelhead population in California, although it is highly accessible to humans and was heavily dredged for gold. The estimated average abundance for 1979-2006 was 647 summer steelhead, with an increase through the 1990s. The estimated abundance was 2108 fish in 2003, averaging 977 in 2004-2006.

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 have the largest summer steelhead populations on the Klamath River averaging more than 300 fishes annually during the years they were surveyed. 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 over the past few years and the 2004 and 2006 counts were 410 and 275, respectively.

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 fish per year. These watersheds were heavily mined during the late 19th century and smaller scale mining continues in the river during summer. 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.

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 100-300 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 and 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).

Factors affecting status: Summer steelhead are exceptionally vulnerable to human activities because adults are conspicuous in their summer pools, so vulnerable to poaching, and because all life stages are present in rivers for extended periods of time. Summer steelhead, like other salmonids, are subject to the legacy effects of 19th century hydraulic mining and logging, 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. Here we discuss some of the major factors causing declines, which are dams, logging, mining, harvest, and disturbance. There is no hatchery production of summer steelhead, so their populations truly reflect local conditions. 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: The construction of dams that have blocked access of steelhead to upstream areas on the Klamath, Shasta, and Trinity rivers diminished the total habitat available to them. These fish probably ascended higher in each watershed than any other salmonid.

Logging: 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 probably exacerbated by the effects of the 1964 floods in almost all drainages containing summer steelhead. 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 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 1964 flood is gradually being scoured out of the pools, but much of it still remains. The potential for further mass wasting along the Trinity 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 natural predation, particularly that of 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. E. Naylor, CDFG, 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 life 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 (Ricker 1997; Jong 1997; Borok and Jong 1997). Accumulation of gravel in stream beds in recent years has reduced the amount of suitable habitat for summer steelhead by reducing pools 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 destabilize the steep slopes of the coastal mountains. In more recent years, the upswing of suction dredge mining is creating problems for vulnerable over-summering fish (see UKTR spring Chinook account).

Harvest: Steelhead are harvested legally as they migrate upstream to spawn, as well as in the ocean. But perhaps the most immediate threat to summer steelhead is poaching during the summer in canyon pools. The steelhead are unusually vulnerable at this time because they are conspicuous, aggregate in pools, and are prevented from leaving by low stream flow. They can thus be snagged from the bank or speared by divers. Roelofs (1983) indicated that the most stable populations of summer steelhead are in the most inaccessible streams on public land, whereas those that are showing signs of severe decline are in areas that are most accessible to people. Roelofs (1983) indicated that poaching is 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. Summer steelhead in the South Fork of the Trinity are also heavily poached (P. Higgins, pers. comm.). In addition to summertime poaching, mortality of adults may occur during late season winter steelhead fishing, as the summer steelhead move upstream towards their holding pools, during spring. The high seas gillnet fishery for squid and other species may also be killing steelhead from California streams. The impact of marine fisheries on steelhead in general is poorly known, but such fisheries may be a source of ocean mortality.

Disturbance: Even where habitats are apparently suitable, summer steelhead may be absent because of continuous disturbance by humans. Heavy use of a stream by gold dredgers, swimmers, and rafters may stress the fish. This may make them less able to survive natural periods of natural stress (e.g., high temperatures), less able to spawn or to survive spawning, and more likely to move to less favorable habitats. Because disturbance makes the fish move around more, they are also more likely to be observed and captured by illegal anglers. Not surprisingly, summer steelhead tend to persist only in the most remote canyons in their watersheds.

Conservation: Conservation recommendations for summer steelhead have been developed for most populations (Jones and Ekman 1980, Roelofs 1983, McEwan and Jackson 1996), but management is not a high priority because they are not listed under state and federal endangered species acts.

Present management focuses on increased monitoring to assess if the populations are recovering naturally, presumably to the point where some harvest will be possible during their migratory period. Although KMP summer steelhead populations appeared to increase slightly through the 1990s, many now reflect their lower average numbers over the longer period of monitoring. Key elements of the *Steelhead Restoration and Management Plan for California* (McEwan and Jackson 1996) for the Klamath River included improved flow regimes in the Klamath and Trinity rivers, which may help increase survival of emigrating juvenile summer steelhead. The restoration plan recognizes the importance of protecting functioning subbasins, allowing natural processes to take precedence over human activities that cause degraded habitat conditions. Greater effort by managers to take measures focusing on restoring favorable instream conditions that benefit multiple species and desired ecosystem function, would help summer steelhead in the Klamath River basin. However, special, intense management is needed in the few watersheds where summer steelhead are most abundant; it should focus on reducing human impacts and improving habitats, especially in ways that keep water temperatures down.

Management plans for *each* population should be included in the Summer Steelhead Management Plan, which was once "being prepared by DFG" (McEwan and Jackson 1996, p 139). These plans should address (1) better enforcement of fishing and land use regulations in over-summering areas, (2) better watershed management to minimize sediment and maintain healthy water quality, (3) better regulation of adult harvest during the migrations, (4) better management of downstream reaches to favor out-migrating smolts, (5) rebuilding of present populations through natural and artificial means, including habitat improvement, (6) restoration of populations that have become extirpated, and (7) some protection of adults and juveniles from predation. Strategies should incorporate approaches from the Steelhead Restoration and Management Plan for California.

Improvement of summer steelhead habitat has not been a priority program for the Department of Fish and Game or other agencies, although reduction in summer carryover habitat has been repeatedly identified as a critical limiting factor. Land management which reduces sedimentation, increases cover, and minimizes changes to summer steelhead over-summering habitat is critical to recovering populations close to extirpation.

The problem with poaching has been reduced in recent years because of the interest of community groups in the plight of summer steelhead. However, another problem may be 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. Although fishing is prohibited in many areas and fines for violations are high, protection of summer steelhead populations may require special guards or stream keepers for a number of years. Where populations are exceptionally low, some relocation of natural predators, mainly otters, may be necessary until steelhead populations are large enough to withstand natural predation.

There is also a considerable need for research on summer steelhead populations in California, 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, gold dredging and disturbance from recreation on adults. For most populations, there is a need to accurately census populations and to identify the factors that limit their numbers.

Trends:

Short term: Summer steelhead populations have been reduced to levels far below historic levels and only 2-3 populations are large enough now to expect persistence for more than 10-25 years under present conditions. Most of the smaller populations are likely to disappear in the near future.

*Long term***:** The long term decline experienced by KMP summer steelhead seems to be continuing and their eventual extinction as a distinct life history strategy seems likely if present trends continue. Climate change will likely have significant impacts on summer steelhead because it will influence volume, temperature, and seasonal flow patterns of water in watersheds containing summer steelhead, which will likely lead to further reduction in suitable habitat for spawning and over-summering. 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 and keep temperatures cool.

Status: 2. KMP summer steelhead have a high likelihood of going extinct within the next 50-100 years because of lack of strong protection combined with climate change affecting adult holding and juvenile rearing habitat (Table 2). 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 were judged not warranted for listing by NMFS in 2002 because they are considered part of the larger KMP steelhead ESU and therefore not separated from the more abundant winter steelhead.

Metric	Score	Justification
Area occupied	2	Much diminished from historic distribution
Effective pop. Size	2	Populations are very small and isolated.
Intervention dependence	3	No intervention is being undertaken to assist in persistence, but it is badly needed.
Tolerance	2	Adults require cold water refuges
Genetic risk	2	Hybridization risk with winter steelhead, especially hatchery fish, is high.
Climate change	1	Highly vulnerable; temperatures and flows already marginal in many areas.
Average	2	12/6
Certainty (1-4)	3	Well documented

Table 2. Metrics for determining the status of KMP summer steelhead, where 1 is poor value and 5 is excellent.

Appendix 1. Observed number of adult summer steelhead in Klamath Mountain Province stream and rivers. Estimates were compiled from McEwan and Jackson 1996, Loren Everett, personal communication, and Leroy Cyr, personal communications.

Watershed	Source	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Bluff	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	41[3]	37	16	87	23	48	23[3]	73[2]	73[2]	91	44	91	212	149	31	15	20	15	2	15	5	9	9	35	31	20	10
Red Cap	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	45	12	11	18[2]	ns	29	25	25	7	2	31	8	4	3	6	1	6	3	0	2	9	23	20	10
Camp	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	18	ns	1	7	ns	2	2	1	0	4	0	0	2	4	5	3	13
Wooley	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	105	160	166	249	353	78	92	290	ns	285	362	245	73	25	38	112	54	42	15	54	41	30	49	214	288	288	110	50
Dillon	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	236	187	295	300	200[4]	162	ns	77	294[4]	38	74[2]	88	ns	161	ns	122	91	180	151	209	679	929	1108	576	437	216
Clear	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	1810	79	241	270	618	257[2]	156[2]	162[3]	428	524	693	934	117	39	100	178	134	175	102	85	68	65	186	538	1034	238	268	108
Elk	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	408	ns	90	47[2]	249	ns	18	ns	ns	31	69	150[2]	57	44	72	61	110	61	96	33	490	23	77	212	200	55	112	34
Indian	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	421	ns	ns	ns	15[2]	ns	ns	ns	ns	ns	46[2]	154[2]	21	8	271	67	117	39	ns	42	ns	ns	ns	ns	ns	4	ns	ns
Thompson	A, B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	4[P]	14[P]	13[P]	ns	ns	ns	ns	ns	46[P]	17[P]	9[P]
Salmon River	A,E	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	235	120	266	ns	ns	17	19	10	324[H]	47[H]	110[H]	67[H]	99[H]	179[H]	202[H]	175[H]	165[H]	141[H]	193[H]	172[H]	214[H]	338[H]	312[H]	192[H]	357[H]	324[H]
Grider	B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	no	ns	ns	ns	29	no	44	3
Canyon Creek	A,C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	6	3	20	3	20	10	ns	no	32	ns	15	3	6	24	45	23	5	26	42	16	27	33	13	40	24	7
North Fork Trinity	A,C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	200[4]	320	456	219	193[2]	160	180	57	ns	300	624	347[4]	554	1037	369	604	990	830	396	339	149	187	370	975	985	1042	453	443
New River	A,C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	341	320	236	114[3]	ns	335	ns	ns	ns	500[4]	699	381	748	358	368	427	817	307	651	495	538[4]	515	995	1500	2108	1156	843
South Fork Trinity	A, C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	6	ns	ns	26	ns	ns	15	73	ns	26	37	66	18	29	42	22	42	11	95	37	38	76	75	77	37	34	105
SF Smith	A, D	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	2	ns	ns	ns	ns	ns	12[2]	4[2]	8[2]	13[2]	8[2]	4[2]	5	4	9	no	ns	no	13	1		2	8	13
MF Smith	A,D	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	2[2]	ns	ns	ns	ns	ns	21	1[2]	18	11	13	5	2	11	11	6	6	no	6	no	ns	1	6	2
NF Smith	A,D	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	2[2]	ns	ns	ns	ns	ns	12	4	8	no	13	no	no	4	4	ns	ns	ns	ns	ns	ns	ns	ns	ns

ns= no survey
: none observed
cally how much adult holding habitat is surveyed
0 to 100% of the adult holding habitat , if known.
50 to 69% of the adult holding habitat, if known.
25 to 49% of the adult holding habitat, if known.
ased on expansion of partial count
lt spawning habitat or spot surveys, if known.
nder included in count.

Sources (if observed numbers differ, larger number of observations is reported)
A= McEwan and Jackson 1995
B= USFS, Klamath National Forest, Orleans/Happy Camp Ranger District Files
C= USFS, Trinity-Shasta National Forest, Files
D= Friends of the Smith River, Summer Diver Reports
E= USFS, Rebecca Quinones, Klamath National Forest

NORTHERN CALIFORNIA COASTAL WINTER STEELHEAD
Oncorhynchus mykiss

Description: Steelhead are anadromous rainbow trout which return from the ocean as large silvery trout with numerous black spots on their tail, adipose and dorsal fins. The spots on the tail are typically in radiating lines. Their back can be an iridescent blue to nearly brown or olive. Their sides and belly appear silver, white, or yellow with an iridescent pink to red lateral band. The mouth is large, with the maxillary bone usually extending behind 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. Resident adult trout may retain the color patterns of parr (Moyle 2002).

The various forms in California are identical morphologically and are distinguished mainly by genetics, although different populations may show some variation in the average size of returning adults.

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, rainbow trout, including steelhead, are 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 genetic units (ESUs and DPSs, Box 1) recognized by NMFS for California have more or less discrete geographic boundaries, with genetic similarities between adjacent populations across ESU boundaries. These units are used as the basis for independent steelhead accounts in this report.

The Northern California coastal winter (NCCW) steelhead is a well-supported, easily identifiable group of 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). The northernmost populations of NCCW steelhead show a 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 NCCW 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 NCCW steelhead populations in the Mad River and Redwood Creek cluster with steelhead populations from other NCCW steelhead streams, which either reflects ecotypes adapted to local conditions in these environmentally diverse basins or the intra-DPS transfer of NCCW 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, 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 NCCW 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 historic 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).

The larger watersheds within the range of this DPS also support summer run steelhead in Redwood Creek and the Mad, Mattole, Eel, and Van Duzen Rivers. We have a separate account for Northern California coastal summer steelhead because their distinct life history strategy requires different conservation frameworks.

Box 1: National Marine Fisheries Service (NMFS), ESUs and DPSs

In the federal Endangered Species Act (ESA) of 1972 a species is defined as a species, subspecies or distinct population segment of a species. The nature of a distinct population segment was not well defined, so agencies working with endangered species had to come up with their own definitions. The National Marine Fisheries Service (NMFS), as a consequence, created the Evolutionary Significant Unit (ESU) for the management of endangered salmonids. According to NMFS "An ESU is a population or group of populations that (1) is substantially reproductively isolated from other conspecific population units, and (2) represents an important component in the evolutionary legacy of the species." This definition arose in part to avoid having to list individual runs of anadromous fish, allowing groups of runs with a common genetic heritage to be treated together. Subsequently, most species listed by NMFS under the ESA were ESUs. In 2005, however, NMFS developed a joint policy with the US Fish and Wildlife Service (USFWS) to use the less restrictive Distinct Population Segment (DPS) for steelhead, in order to be able to list anadromous forms, while not listing resident forms. This allows sympatric, interbreeding resident and anadromous rainbow trout in the same stream to be treated as different DPSs. The DPS Policy states that a group of organisms forms 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 Fed. Reg. 4722 (Feb. 7, 1996)].

One reason that this works for steelhead is that NMFS has jurisdiction over anadromous fishes, while USFWS has jurisdiction over resident fishes, allowing resident and anadromous forms to be separated on the basis of who is allowed to manage them. This additional flexibility in implementing the Endangered Species Act for *O. mykiss* in California should benefit recovery of many populations in a biologically significant fashion. Ecologists have long recognized that *O. mykiss* populations within the same geographic range exhibit distinctive behavioral and physiological life-history traits, yet the ESU criteria has limited the precautionary application of an approach that embraces these distinctions for *O. mykiss*. With the use of DPS criteria for *O. mykiss* under the ESA, potential delineation of summer

Life History: In general, rainbow trout, which include steelhead, exhibit the largest geographic range and most complex suite of traits of any salmonid species. Anadromous steelhead and resident rainbow trout in many rivers are part of a single gene pool which contributes to the ability of coastal rainbow trout to adapt to systems that are highly unpredictable and undergo frequent disturbance. The life history of steelhead in California is covered in Moyle (2002). Basically, 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.

NCCW steelhead enter estuaries and rivers between September and March (Busby et al. 1996). Further migrations upstream occur as late as June, but timing depends upon rainfall and consequent stream discharge being suitable 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. 1996), though favorably wet conditions may lengthen the spawning period into May. These spawning steelhead arrive at spawning areas in reproductive condition. 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).

Unlike salmon, steelhead can spawn more than once. 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 (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. 1996). 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, 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 and recent studies have shown that some one year old smolts return as adults (Mike Sparkman, CDFG, pers. comm.). However, successful juveniles typically 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. NCCW steelhead grow from 0.24 to 0.37 mm/day in the Navarro and Mattole Rivers, respectively (Zedonis 1990; Cannata 1998). In Redwood Creek, growth rates were greater, ranging from 0.26 to 0.73 mm/day (M. Sparkman, CDFG, pers. comm. 2007). NCCW 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 depending on flows. Young-of-year steelhead will emigrate to estuaries as late as June or July (M. Sparkman, pers. comm. 2007). 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, a greater proportion of older (2+) juveniles reside in the estuary than in the river. 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 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 NCCW steelhead is the 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 NCCW 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 bar formation 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.

California steelhead can spend up to four years in the ocean, though many steelhead returning to the small coastal tributary, Freshwater Creek, 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).

NCCW steelhead were captured in August during trawl surveys north and south of Cape Blanco (Brodeur et al. 2004), suggesting much of their time in the ocean is spent fairly close to their natal streams. Steelhead grow rapidly at sea, feeding on fish, squid, and crustaceans taken in surface waters (Barnhart 1986). It is believed that steelhead use their strong homing sense to return to the same area in which they lived as fry to spawn (Moyle 2002).

In Redwood Creek and the Mad, Eel, and Mattole Rivers, a small number of "half pounder" steelhead are observed annually. These half pounders are likely distinct from the half pounder steelhead in the Klamath Mountain Province, which are reported to enter and leave the river as immature, subadult fish (Kesner and Barnhart 1972). The NCCW steelhead half pounders are generally larger (25-35 cm FL or larger) than Klamath fish but they are not well documented. The high phenotypic plasticity in juvenile and adult life histories demonstrated by NCCW steelhead suggest the 'half pounders' may represent small reproductive fish, large resident fish, or a mixture of different life history variations.

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 streams (Moyle 2002). Adult steelhead require high flows with water at least 18 cm deep for passage

(Bjornn and Reiser 1991). 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 the adults (Moyle 2002), although migrating winter steelhead usually do not encounter these conditions (Table 1). For spawning, 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) .

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 (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 (<2.0mm) 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 for steelhead and can lead to reduced growth and increased mortality. As temperatures become stressful, juvenile steelhead will move into faster riffles to feed due to increased prey abundance (see 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 temperature less than 17°C were "good", 17°-19°C were "marginal" and higher than 19°C were "unsuitable/poor" (Coates 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 (MSG 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 and bioenergetics box in SONCC coho account).

	Sub-Optimal	Optimal	Sub-Optimal	Lethal	Notes
Adult Migration	<10ºC	10-20°C	20-23°C	>23-24°C	Migration usually stops when temperatures climb above 21ºC. Lethal temperature under most conditions is 22- 24ºC. Fish observed moving at higher temperatures are stressed and searching for cooler refuges.
Adult Holding	<10ºC	10-15°C	16-25°C	>26-27°C	These temperatures are for summer steelhead, which survive the highest holding temperatures. If high temperatures are frequent, egg viability of females may be reduced.
Adult Spawning	<4ºC	4-11°C	12-19°C	>19°C	Egg viability in females 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 temperatures) 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 (a sign of stress) start being produced at 17°C.
Smolt-ification	<7ºC	7-15°C	15-24°C	>24°C	Smolts may survive and grow at suboptimal temperatures but have a harder time avoiding predators;

Table 1. Temperature requirements for steelhead, from Richter and Kolmes (2005), McEwan and Jackson (1996), and Moyle (2002). Values may vary according to acclimation history and strain of trout.

Steelhead have a body form adapted for holding in fast water, more so 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 to 3.9 m/sec (Reiser and Bjornn 1979 in Spence et al. 1996). Preferred holding velocities are much slower, and range from 0.19m/sec for juveniles and 0.28m/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.

Juvenile steelhead rear in the estuaries of Redwood Creek, Humboldt Bay, and the Eel, Navarro, Garcia, Gualala Rivers. 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 4.5mg/L negatively affect juvenile steelhead trout (Barnhart 1986), although they can survive DO levels as low as 1.5-2.0mg/L for short periods of time (Moyle 2002).

Distribution: Along the eastern Pacific, rainbow trout, including steelhead, are distributed from Southern California north to Alaska and range west to Siberia (Sheppard 1972). In California, steelhead occur in coastal streams from the Oregon border down to San Diego County and up to barriers to migration throughout their distribution. The NCCW steelhead DPS includes all naturally spawning populations of steelhead in California coastal river basins from Redwood Creek (Humboldt Co.) to just south of the Gualala River (Mendocino Co.) (Spence et al. 2007). 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. Spence et al. (2007) identified 32 historically self-sustaining populations in the DPS region based on habitat availability and gene flow among watersheds. An additional 33 small populations are likely dependent upon immigration of non-natal steelhead from the more permanent populations (Bjorkstedt et al. 2005). With few exceptions, NC steelhead are present wherever streams are accessible to anadromous fishes and there are sufficient flows. Big and Stone lagoons, between Redwood Creek and Little River, contain steelhead following their opening to the ocean in the early winter, although the source of these fish is unknown (M. Sparkman, pers. comm.).

Abundance: Little historical abundance information exists for naturally spawning populations of NCCW steelhead, but the current abundance of this species is apparently quite low relative to historical estimates (Figure 1).

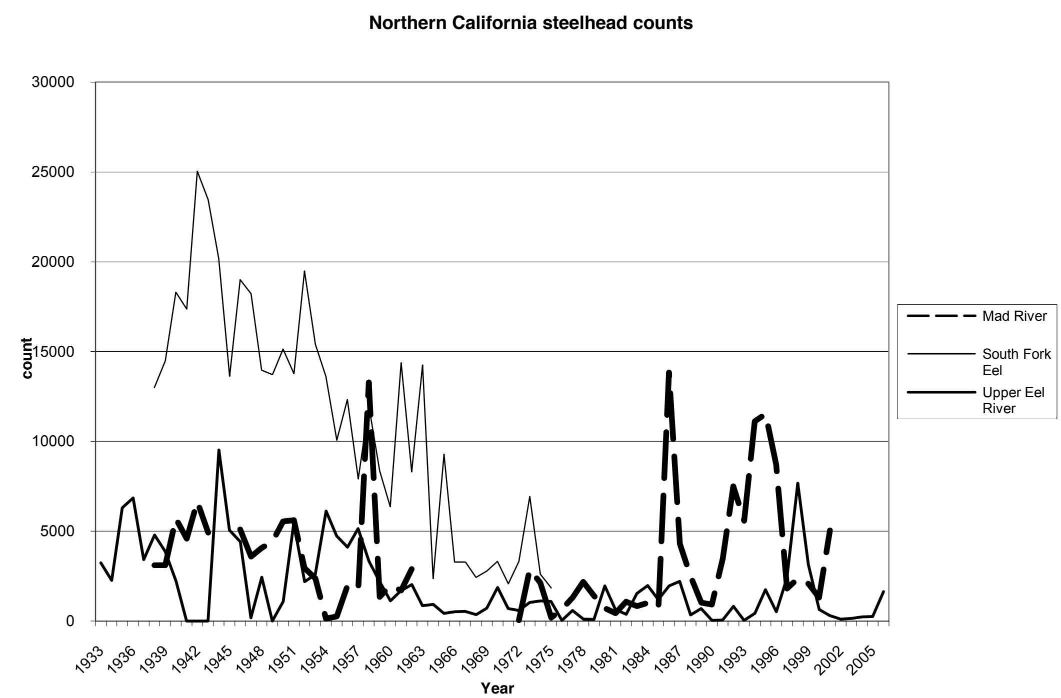


Figure 1. Northern California steelhead DPS counts from two locations in the Mad River (Sweasey Dam (pre-1964) and Mad River hatchery(1972+), South Fork Eel River (Benbow Dam), and Upper Eel River (Van Arsdale Station). Data from Taylor 1978, California Department of Fish and Game, and Grass 2007.

In the Mad River, CDFG (1965) estimated that about 6,000 steelhead spawned annually. 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 historic 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 only 8.3% (1,419) comprised of wild fish. The Eel River is the most important steelhead producing river in this DPS and once supported at least 82,000 steelhead with the South and Middle forks combining to hold 70% of these spawning fish (Taylor 1978). 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 Ardsdale Dam on the mainstem Eel River (Taylor 1978), which both show long-term declines in abundance (Figure 1). 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).

The Mattole, Big, Navarro, and Gualala Rivers were thought 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. 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 (Scott Harris, 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 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 (Scott Harris, per comm. 2007).

Overall, CDFG (1965) suggested that close to 200,000 NCCW steelhead once spawned the region's rivers combined. Optimistically, annual spawning returns today range from 25,000-50,000 fish. However, data sets that allow long term trends to be determined quantitatively are lacking and all that can really be said is that every indicator suggests that 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, the result is severe decline. Here we discuss only some of the more regional factors for NCCW steelhead; other cosmopolitan factors (e.g., freshwater and estuarine habitat degradation, water diversions, and gravel extraction) are discussed in accounts for Central California coastal steelhead and Central California Coast Chinook and Coho salmon.

Barriers: 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 Ruth Dam, while in the upper mainstem Eel River more than 90% of available habitat is blocked by Scott Dam (Spence et al. 2007).While these represent significant habitat constrictions, culverts and bridges are barriers to steelhead passage in numerous smaller watersheds across the NCCW steelhead region. In the Eel River, a more significant problem associated with Scott Dam is the reduction of flows into the mainstem Eel River. This flow reduction negatively impacts mainstem water quality during summer and fall, reduces stream complexity, and constricts 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 (next paragraph). Even in this reach it is not certain if the higher flows and colder temperatures help steelhead populations. Barrier inventories have been completed in NCCW steelhead counties, but most are still in place because considerable effort is required to eliminate even the priority barriers.

Flow releases in the reach between Scott Dam and Cape Horn Dam have improved summer flows and temperatures since 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 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 NCCW steelhead landscape is industrial timberlands, both private and public, which have already undergone one or more cycles of tree removal, include intense no-holds-barred 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 (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 the 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 coastal 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.

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 temperature in the streams (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 NCCW 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 (Harvey et al. 2002).

Together, poor land use practices associated with logging, agriculture, gravel mining, road construction, and vineyard construction negatively impact instream and upslope conditions for steelhead in most watersheds in which they occur. 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 the suitability of impacted reaches for steelhead and numerous populations inhabit impaired watersheds where TMDL Basin Plans are being developed.

Fisheries: 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. Fishing is allowed on the Mad River for ten months, is directed towards hatchery steelhead, 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 hatchery and the catch rate for wild and hatchery fish did not show any clear relationship in these three basins.

Hatcheries: No studies have been carried out to evaluate the impact of hatchery releases on wild steelhead and other salmonids in the northern California coastal region, but studies elsewhere have shown that releases of large numbers of fish result in negative competitive interactions between wild steelhead and hatchery fish for food, habitat, and mates (Nickelson et al. 1986). Also, carrying capacity of rivers is often exceeded during the outmigration of hatchery smolts decreasing food availability (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 (Waples 1991).

The principal steelhead hatchery in the region is the Mad River Hatchery, using stocks that were originally from the South Fork Eel River. These fish have been widely planted throughout the NCCW steelhead region and may account for some of the genetic ambiguity that exists (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.

Alien species: Non-native species are present in many of the watersheds used by NC steelhead, but the biggest problem has been created by the invasion of the Eel River system by Sacramento pikeminnow (Brown and Moyle 1997). Pikeminnow not only prey directly on juvenile steelhead but they displace them from pool habitat into less desirable riffle habitat, presumably resulting in reduced growth and survival.

Conservation: The current unfocused 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 may still not conserve 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. Thus steps necessary to restore steelhead to historic numbers and protect all life-history types potentially requires management and societal changes within each watershed. This need had been recognized in the *Steelhead Restoration and Management Plan for California* (McEwan and Jackson 1996), which has largely not been implemented, as shown here for NCCW steelhead.

Listing of this DPS under the Endangered Species Act was influenced by the failure of the State of California to follow guidelines agreed upon in a 1998 NMFS/California Memorandum of Agreement (MOA), particularly improvements to the California Forest Practices Act. The objectives of this MOA remain critical to the recovery of NCCW steelhead, yet almost a decade later, most of them have not been enacted. Many of the guidelines specifically addressed the factors affecting the status of steelhead described above. As part of the Pacific Coast Salmon Recovery Fund, 83% of the funded restoration activities in the North-Central California Coast Recovery Domain addressed habitat limiting factors for steelhead (NMFS 2006) .

Critical habitat was delineated on September 2, 2005 (NMFS 2005) and includes approximately 4075 km (3,028 mi) of stream habitat and 65 square km (25 square mi) of estuarine habitat, primarily in Humboldt Bay. Critical limiting factors that need to be addressed for recovery of the steelhead DPS include degraded estuarine, riparian, and in channel habitats; fish passage; hatchery-related effects, harvest, and predation, competition, and disease (NMFS 2006). Current state and federal conservation measures cumulatively do not provide the necessary social, fiscal, or regulatory support necessary for long term protection and recovery, though efforts underway will influence the trajectory of recovery activities (NMFS 2007).

Steelhead abundance has been impacted by poor water quality and sediment in most basins, even though these pollutants can be regulated through the Clean Water Act's Basin Plan framework. Activities of the California Regional Water Quality Control Board to reduce sediment and temperature impairment in many of the streams as part of the Total Maximum Daily Load reduction effort may benefit NCCW steelhead if reductions are successful.

CDFG and NMFS have been developing a statewide coastal salmonid monitoring program for a number of years, yet it has not been implemented. Developing comprehensive abundance and trend data for coastal salmonids is essential for assessing the viability and recovery of NCCW 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 CDFG Fisheries Grant Restoration Program, although limitations in funding have never allowed the Grant Program to meet the identified habitat restoration needs of NCCW steelhead.

Currently, a majority of timberlands along California's coast have or are developing Habitat Conservation Plans (HCP) for listed species, including NCCW steelhead. 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. The potential direct and cumulative negative effects of logging are well documented (Spence et al. 1996) Ongoing HCP planning efforts should be vetted by the new viability and recovery framework for NCCW steelhead developed by Spence et al. (2007), and there is need to better integrate HCPs with other watershed-based management actions.

Funds generated by sales of the Report Card purchased by anglers have not been used much in the region although allocations for development of Fishery Management and Enhancement Plans for NCCW steelhead fisheries are needed. In any case, a more intensive creel survey of the Eel River, similar to those completed historically (Puckett 1978) would prove more useful than the current information derived from the Report Card, if combined with effective monitoring of wild productivity and escapement.

Hatcheries can play a significant role in conservation of these steelhead but only with careful monitoring. Further monitoring, including development of a hatchery genetic management plan, should be undertaken to minimize the risks associated with the operation of hatcheries on naturally-produced NCCW steelhead.

Overall, conservation efforts for NCCW steelhead have been minimal compared to the size of the problems they face. Much has been planned but little has been implemented in the past 10 years. Industrial logging has left significant legacy problems and contemporary protective measures are not being undertaken quickly enough to conserve upslope and riparian habitats that affect steelhead and to preserve favorable instream conditions. Selective logging, protection of erosion-prone slopes, environmentally-sensitive road construction, and ecologically sustainable water management are new paradigms for best management practices in the NCCW steelhead region necessary for recovering these fishes, but seem to be little used. The Eel River was the main steelhead producer among the NCCW steelhead streams, but now is heavily impacted by sedimentation by roads and logging, flow reduction, habitat barriers, alien species, and water quality impairment. An ecosystem approach to managing salmonid and nonnative fish in the Eel River will be necessary to maintain the steelhead population in the tributaries and forks of this basin in the long term. Any such improvements will need to consider climate change scenarios for at least the next century, to consider maintaining the abundance of NCCW steelhead around 50,000 spawners (25% of 1965 levels).

Trends:
Short term: NCCW steelhead continue to occupy a large portion of their historic distribution although dams, culvert, and other barriers limit their distribution in most watersheds. Population abundances are largely unknown, but estimated to be low in comparison to historic estimates and recent analyses have shown a downward trend (Good et al. 2005). Until better regulation of in-channel and upslope land practices influencing steelhead populations and increasing restoration efforts are able to provide habitat for steelhead, this steelhead group is likely to continue to decline, if not as rapidly as many other anadromous fishes in California.
Long term: While NCCW steelhead have a long-term declining trend in numbers, they may be able to continue to occupy much of their historic range, if regulatory and restoration efforts are effective. Without such efforts, populations are likely to be lost one after another, first in the smaller streams, then in the larger rivers.

Status: 3. NCCW steelhead have a low risk of extinction in the next 50-100 years although better information could change 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. Populations of NCCW steelhead are large enough and appear to be declining slowly enough so that there is no immediate threat of extinction throughout the region, although smaller populations may disappear soon. However, this status could deteriorate rapidly if restoration and protection efforts are not put into effect. NCCW steelhead currently have no special conservation status with the state of California beyond being a fishery species. Due to their continuing decline, NCCW 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.

Metric	Score	Justification
Area occupied	3	Multiple watersheds in CA
Effective pop. Size	3	About 1000 wild spawning steelhead present annually in the Mad and Eel Rivers, and other populations (Redwood Creek, Mattole, and Garcia) may contain as many though information is lacking
Intervention dependence	3	Require continuous monitoring and improvement of habitat 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	3	Coast range has cooler temperatures and more consistent flow in most basins, but effects can be high in altered watersheds
Average	3.3	20/6
Certainty (1-4)	2-3	Actual numbers of fish poorly known

Table 2. Metrics to determine the status of Northern California Coast Winter Steelhead, where 1 is poor value and 5 is excellent.

NORTHERN CALIFORNIA COASTAL SUMMER STEELHEAD
Oncorhynchus mykiss

Description: Summer steelhead are morphologically similar to Northern California coastal (NCC) winter steelhead (see account).

Taxonomic Relationships: For general relationships, see NCC winter steelhead account. NCC summer steelhead are found in a small number of streams that also contain populations of winter run NCC steelhead. These populations are isolated from each other principally by life history differences. The differences include: (1) time of migration (Roelofs, 1983), (2) state of gonadal maturity at migration (Shapovalov and Taft 1954), and (3) location of spawning (Everest 1973, Roelofs 1983).

In an early genetic study on summer NCC steelhead, differences were observed in the Middle Fork Eel River between summer and winter run steelhead when compared with other coastal winter run populations (Nielsen and Fountain 1999). Clemento (2006) evaluated the genetic relationships among winter and summer steelhead in the Middle Fork Eel River over multiple years. He found that fish of both types were genetically from steelhead in the South Fork Eel River, Lawrence Creek (Van Duzen River tributary), and Willits Creek (upper Eel River tributary). Among multiple years of summer steelhead collections there was little genetic differentiation but these samples were distinct from winter steelhead from Black Butte River, the main lower tributary of the Middle Fork. Genetic analyses among juvenile steelhead in upper Middle Fork Eel River tributaries (North Fork and Cutfinger Creek) showed significant differences, indicating isolated, small, resident populations (Clemento 2006). These genetic studies all suggest that NCC summer steelhead within various basins are most closely related to proximate NCC winter steelhead stocks.

We nevertheless treat NCC summer steelhead as a distinct entity because of its striking differences in life history and ecology from NCC winter steelhead. An alternative based strictly on genetics would be to treat all three of the remaining NCC summer steelhead populations as separate Distinct Population Segments.

Life History: The basic life history of summer steelhead is (1) adults migrate upstream in spring to holding pools in headwaters as immature adults, (2) adults hold through the summer in deep pools, (3) adults spawn in fall and survivors migrate back to the ocean, and (4) juveniles rear in headwater streams as well as streams lower in the watershed for 1-3 years, and (5) smolts migrate out to sea during high winter flows. Very few studies have been carried out on NCC summer steelhead, though some research has been completed on these fish in the Middle Fork Eel River population. NCC summer steelhead migrate into the upper Middle Fork Eel River from mid-April through June (Puckett 1975; Jones and Ekman 1980). Migration may extend into July, but fish are increasingly less likely to make it to upstream areas as mainstem flows decrease and stream temperatures increase. Returning adult summer steelhead have an age composition of 1% 2 year olds, 46% 3 year olds, 44% 4 year olds, and 9% five year olds; with 13% of the fish spawning more than once (Puckett 1975). Oversummering summer steelhead have been observed to migrate among pools (Nielsen et al. 1994), though later in the season the pools are often hydrologically disconnected. It is possible that steelhead from large populations also enter smaller rivers (i.e., Mad River and Redwood Creek) following the first fall rain and contribute to other summer populations (T. Weseloh, California Trout, pers. comm.). Spawning timing has not been well documented for NCC summer steelhead and may occur at the same months as winter steelhead. However, it is presumed that temporal and spatial isolation of reproductive fish from sympatric winter steelhead runs serves to maintain the integrity of summer steelhead (Barnhardt 1994). The mountainous high gradient stream reaches inhabited by summer steelhead in the Middle Fork Eel River likely reinforces their spatial isolation from winter steelhead. Spawning habitat is likely similar to that of KMP summer steelhead (see description).

Juvenile and ocean life history of NCC summer steelhead is undocumented, but it is presumably similar to KMP summer steelhead. In the Mattole River, a small number of "half pounder" steelhead are observed during annual summer steelhead dive surveys. This phenotype in NCC summer steelhead is not well documented and they may be subadult 'half-pounders' similar to those observed further north. Alternatively, these fish may represent large resident trout or small returning adult summer steelhead. Greater monitoring and research is necessary to adequately describe this life history variation of the NCC summer steelhead.

Habitat Requirements: Basic habitat requirements of NCC summer steelhead are generally similar to those of other steelhead, though their over-summering in rivers requires ability to survive in a special set of conditions. Due to their long migration through mountainous terrain into the Middle Fork Eel River, NCC summer steelhead require adequate flows to reach optimal over-summering habitats. Water depth does not seem to be critical to migrating fish because they usually migrate when stream flows are high, but a minimum depth of 13 cm is required (NOAA 2005). Water velocities greater than 3-4 m sec^{-1}, however, may impede their upstream progress. Lack of spring rain and a poor snow pack will curtail migration of summer steelhead and isolate these fish in the lower reaches of the Middle Fork, which have warmer, potentially lethal, stream temperatures (Scott Harris, CDFG, pers communication).

Temperature requirements for NCC summer steelhead are presumed to be similar to KMP summer steelhead because both stocks live in similar mountainous habitats. For most adult steelhead temperatures of 23-24°C can be lethal (see NCC winter steelhead account) but summer NCC steelhead likely regularly encounter temperatures in this range. Jones (1980) reported summer temperatures in the Middle Fork Eel River of 17-24ºC. Cold tributary confluences are critical oversummering location for NCC summer steelhead. Steep, well-shaded, narrow tributaries contributed as much as 95% of the stream flow during the late summer in the river and are often 3- 4ºC cooler than the mainstem (Jones 1980). Additionally, snowmelt lasting into the spring and temperature stratification in deep pools provides cool habitats for summer adult steelhead to oversummer in mountainous watersheds. In the Middle Fork Eel River, 93% of the summer steelhead occupy pools deeper than 1.6m with cover such as underwater ledges, caverns and bubble curtains which they seek when disturbed (Puckett 1975; Roelofs 1983). Jones (1980) characterized these pools to be thermally stratified, with the average difference between the bottom and the surface being 1.8ºC. In 2004, 66% of the Middle Fork summer steelhead occupied pools between 3.1 and 6.1m (Scott Harris, pers. comm.).

In watersheds inhabited by NCC summer steelhead, complex and well-shaded habitats with appropriate depths and temperatures are important for oversummering of adult fish (Nakamoto 1994). These features and alluvial recharge (Nielsen et al. 1994) via springs and seeps provide cool areas for fish. Dissolved oxygen requirements for spawning fish generally need to be at least 80 percent of saturation, with temporary levels not less than 5.0 mg l^{-1} (Reiser and Bjornn 1979).

Distribution: NCC summer steelhead are patchily distributed in a small number of watersheds. Populations appear to remain in Redwood Creek and the Mad, Van Duzen, Middle Fork Eel, and Mattole rivers, although only the Mad and Middle Fork Eel populations are likely to persist. Other populations exist or did exist in the North Fork Eel, Upper Mainstem Eel, and South Fork Eel rivers

A survey of the Middle Fork Eel River drainage indicated that the best steelhead spawning gravels are located at Balm of Gilead Creek, North Fork of the Middle Fork Eel River, and in the Middle Fork from Hoxie Crossing to the North Fork of Middle Fork (Jones 1980), though other areas also appear to support spawning (S. Harris, pers. comm.). Redds have been observed in the Middle Fork approximately 0.5 km below the North Fork (Jones 1980).

Abundance: Little historical abundance information exists for NCC summer steelhead. However, it appears that a majority of NC summer steelhead populations have declined precipitously since initial recognition of these fishes' presence in occupied watersheds 30 to 40 years ago. Extirpation of most remaining populations is a serious threat with a majority of populations declining to extremely low populations since the 1980s. The majority of these populations appear to remain at levels below the critical threshold necessary for persistence and further research is required to determine the reasons for this. Adult summer steelhead estimates are typically of fish holding in some portion of possible pool habitat during midsummer and indicate general trends of abundance although likely many of these fish do not survive to spawn. However, the counts may also represent an unknown of the total number of summer steelhead actually in the stream. Typically the counts are based on unreplicated observations, do not contain entire watersheds, and lack reference reaches.

The longest set of population estimates goes back to 1966 for the Middle Fork Eel River (Table 1). Recent efforts have included surveys in the Mattole, Van Duzen and Mad Rivers and Redwood Creek, but survey effort has often been inconsistent. The Middle Fork Eel River appears to contain a sufficient number of summer steelhead to maintain a viable population. The number of adult summer steelhead counted in the Middle Fork Eel River has ranged from 198 to 1601 during the annual summer dives (Table 1, Scott Harris, pers. comm.) and was lowest following the 1964 flood. This flood likely caused loss of deep, complex pools needed for over-summering habitat. The effect of this flood, compounded by continued sedimentation from logging and road building in the latter part of the 20th century, have reduced NCC summer steelhead to abundances below population viability in most watersheds.

Table 1. Number of summer steelhead observed in Northern California steelhead DPS streams. All attempts have been made to exclude half-pounders from these counts. Data from Scott Harris, CDFG.

Watershed	Source	1966	1967	1968	1969	1970	1971	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985
Redwood Creek	B	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	16[3]	2[3]	7[3]	44[3]	44[3]
Mad River	A, C	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	2[2]	6[3]	166	31[3]	134[3]	52[3]
Van Duzen	A	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	31[2]	25[2]	6	8	13	58	ns
Middle Fork Eel	D	198	241	335	ns	865	997	502	1422	1522	1149	792	654	377	1298	1052	1600	1051	666	1524	1463
Mattole River	E	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	44[4]	ns	ns	ns	ns	ns

Watershed	Source	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Redwood Creek	B	19[3]	15[2]	8[3]	no[3]	14[3]	15[3]	7[3]	8[3]	36[1]	18[2]	23[2]	37[2]	25[2]	10[3]	3[3]	1[3]	3[3]	6[3]	15[3]	22[3]
Mad River	A, C	5[3]	18[2]	60[3]	20[3]	33[3]	66[3]	34[3]	48[3]	305	564	427	230	178	78	80	171[P]	185	480	209[P]	210[P]
Van Duzen	A	ns	52	42[2]	4[2]	ns	31[2]	no	ns	ns	2	ns	3	11	7	11	18	30[P]	80[P]	26[P]	30[P]
Middle Fork Eel	D	1000	1550	711	726	449	691	516	622	701	1148	771	513	527	451	306	422	418	657	731	629
Mattole River	E	ns	ns	ns	ns	ns	ns	ns	ns	ns	ns	12	16	30	16	17	17	15	9	16	20

ns= no survey
no= none observed
In many cases, it is unknown specifically how much adult holding habitat is surveyed
1= Estimated from dive surveys of 70 to 100% of the adult holding habitat , if known.
2= Estimated from dive surveys of 50 to 69% of the adult holding habitat, if known.
3= Estimated from dive surveys of 25 to 49% of the adult holding habitat, if known.
4= Cursory estimate based on expansion of partial count
P= Estimated from <24% of adult spawning habitat or spot surveys, if known.
H= half pounder included in count.

Sources (if observed numbers differ, larger number of observations is reported)
A= McEwan and Jackson 1995
B= Dave Anderson, Redwood National Park, Files
C= Fisheries Division, California Fish and Game File
D=Scott Harris, California Department of Fish and Game
E= Mattole Salmon Group (2005)

Factors affecting status: NCC summer steelhead have declined from a combination of factors including habitat loss, water management, disturbance, hatcheries, and poaching. Recent changes in sportfishing regulations and hatchery operations have reduced some of these threats. Discussion of these problems for the entire DPS can be found in the NCC winter steelhead account. Here we only discuss issues specific to NCC summer steelhead

Logging and other land use: The scattered distribution of NCC summer steelhead suggests that stochastic events can have drastic consequences to local populations. Natural disturbance can be synergistic with the decades of poor watershed management, mainly in association with logging, which has occurred in many of the summer steelhead watersheds. The potential for further mass wasting along Redwood Creek, Mad, Eel, Van Duzen, and Mattole Rivers is high, because logging is still occurring on steep slopes and recent fires may be contributing to soil instability (aggravated by road building for salvage logging). These activities intensify peak flows and accumulation of gravels in stream beds, thus reducing the amount of suitable habitat for summer steelhead potentially below amounts necessary for viable populations. It is likely that effects of the 1952 and 1964 floods were exacerbated by land use practices in almost all drainages containing NCC summer steelhead. These floods deposited enormous amounts of gravel into pools that originated from landslides and mass wasting, especially from areas with steep slopes that had been logged. The floods not only filled in pools, but widened stream beds and eliminated riparian vegetation that served as cover and kept streams cooler. The gravel accumulated from the 1964 flood is gradually being scoured out of the pools, but much of it still remains.

Diversions: Increased spring withdrawals from the Upper Eel River at Scott Dam likely reduces the time available for migrating juvenile and adult summer steelhead to move through the mainstem river. Ruth Dam on the Mad River presumably decreases stratification by maintaining flows greater than the natural hydrograph, which are removed by 5 collector wells operated by the Humboldt Bay Municipal Water District in the lower river. In numerous watersheds including the Mattole, Mad, Van Duzen rivers and Redwood Creek, rural landowner water use for residential and agricultural purposes significantly curtail flows in the mainstem river. This reduces habitat availability and truncates migration patterns. In an effort to reduce water intake during the summer, the Mattole Restoration Council has assisted landowners with changing their water withdrawal patterns by filling off-channel storage tanks during the winter.

Disturbance: Even where habitats are apparently suitable, summer steelhead may be absent because of continuous disturbance by humans. Heavy use of streams by gravel mining, swimmers, and rafters may stress the fish. This may make them less able to survive natural periods of stress (e.g., high temperatures), less able to spawn or to survive spawning, and more likely to move to less favorable habitats. Because disturbance makes the fish move around more, they are also more likely to be observed and captured by illegal poachers.

Hatcheries: Hatchery-reared salmonids have adverse effects on wild populations. Summer steelhead were brought into the Mad River Hatchery from the Washougel River, Washington in 1971 (Roelofs 1982) and likely impacted wild summer steelhead. The specific consequences of these hatchery fish on wild stocks of summer steelhead are not known. Summer steelhead are no longer intentionally produced at the Mad River hatchery, so this problem has presumably been alleviated.

Poaching: Illegal harvest of summer steelhead remains a persistent threat to these fish due to lack of adequate game warden or other law enforcement staffing in many of the rural locations occupied by these fish. Reports of poaching are sporadic in the Middle Fork Eel River, with poaching activity likely being common and signs of it observed as recently as 2005-2006 (Scott Harris, CDFG, pers. comm. 2007). Fishing tackle and other evidence of poaching has been found in Redwood Creek and the Mad, Van Duzen, and Mattole Rivers recently (T. Weseloh, California Trout, pers. comm.).

Conservation: The listing of the NCC steelhead DPS, including summer steelhead, as threatened in 2000 was influenced by the failure of the State of California to follow guidelines agreed upon with 1998 NMFS/California Memorandum of Agreement. The objectives of this MOA remain critical to the recovery of NCC summer (and winter) steelhead almost a decade later, yet not all of them have been enacted. Very little management effort is directed specifically at NCC summer steelhead. Comprehensive management recommendations have been made by Jones and Ekman (1980) and Roelofs (1983). These recommendations should be rapidly developed into a new California summer steelhead Conservation Plan, similar to efforts completed to protect California golden trout. This effort is critical and education and outreach needs to be initiated to inform the public and important stakeholders about the status of this imperiled fish. Summer steelhead numbers have not increased in response to limited management efforts over the past two decade. Improvement of summer steelhead habitat has simply not been a priority program for state and federal agencies. The dearth of summer holding habitat is a critical limiting factor, which can be restored through in-channel habitat restoration. The completion of difficult passage projects in the Middle Fork Eel River and mainstem habitat restoration projects in some occupied watersheds have been steps in the right direction and such efforts should receive continued funding and encouragement.

The problem with poaching continues to plague summer steelhead due to the absence of adequate law enforcement. Although fishing is prohibited in many areas and fines for violations are high, remaining summer steelhead populations require special guards or streamkeepers. Moyle et al. (1995) suggested management plans for each population need to be formalized and this still needs to be done. Management needs to move from neglect to adaptive solutions that increase passage and habitat, protect flows, and identify strategies to prepare for stochastic events and climate change. Management should consist of a mixture of (1) better protection of summering areas from poachers, (2) better watershed management to keep summer flows up and temperatures down, (3) better protection from potential poaching of adults during late season catch-and-release winter steelhead fisheries, (4) better management of downstream reaches to favor outmigrating smolts, (5) rebuilding of present populations through habitat improvement, (6) restoration of populations that have become extinct, and (7) some protection of adults and juveniles from predation.

If instituted, none of these recommendations are likely to mean much without monitoring. Monitoring of each population should be continued and formalized as part an interagency management and conservation program. Historically, summer steelhead monitoring occurred through an annual coordination meeting, which has stopped taking place. These annual, or even more frequent meetings, should be reinitiated and taken advantage of to adequately identify problems and reinvigorate monitoring efforts. Occupied basins are principally on public lands and a coordinated effort to monitor these fish would yield valuable insights into their viability.

There is also a considerable need for research on summer steelhead populations, especially to (1) determine the genetic relationships among each population and to the winter run steelhead in these watersheds, (2) determine the extent of possible summer holding areas and potential cool water refuges not being used, (3) determine the distribution of spawning areas and whether they may require special protection, (4) determine the habitat requirements of out-migrating smolts, and (5) determine the effects of disturbance from recreation on adults, and (6) standardize protocols for surveys. For most populations, there remains a need to accurately determine the populations and to identify the factors that limit their numbers.

Trends:

Short term: It appears that summer steelhead populations have continued to decline during the past decade. All occupied basins are subject to water diversions for rural, municipal, or agricultural purposes, and alteration of summer flows likely has a significant impact of these fish. While CDFG has continued to fund passage and habitat projects that can increase access by summer steelhead to preferred oversummering and spawning habitats, the number of fish being observed in monitoring studies has not increased, suggesting these efforts are not adequate. The impact of poor ocean conditions on these populations in unknown, but could be significant.

Long term: Most NCC summer steelhead populations have likely been extirpated in the past 75 years. Long term monitoring for summer steelhead has only occurred in the Middle Fork Eel River basin, which indicates that no real recovery has occurred since their habitats were decimated in 1964. In the future, climate change will increase the variability in the amount of precipitation runoff in the Middle Fork Eel and will likely increase water temperatures in some of the coastal watersheds occupied by summer steelhead. These changes may have drastic consequences for the over-summer survival of summer steelhead without compensatory actions.

Status: 2. Persistence of NCC summer steelhead for more than 50 years seems unlikely if present trends continue. Only the Middle Fork Eel population seems likely to remain viable beyond the next 25 years, although changes in flows and hatchery practices in the Mad River may provide an opportunity for summer steelhead restoration. NCC summer steelhead are part of the NCC steelhead DPS, which was listed as threatened on June 7, 2000 (NMFS 2000) and reaffirmed on January 5, 2006 (NMFS 2006a). Summer steelhead are also considered a Species of Special Concern by the California Department of Fish and Game and a Sensitive Species by the USFS. All remaining populations are declining or small and isolated, although data is generally inadequate (Table 2). While some summer steelhead populations are monitored, information is sparse and not synthesized at a regular interval to assess overall condition of the populations.

Metric	Score	Justification
Area occupied	2	Of five remaining populations, only two (Mad, Middle Fork Eel) seem to be viable
Effective population size	2	Amongst all populations, there are likely ~1000 spawners, but only the Middle Fork Eel has enough fish to persist for more than 25-50 years.
Intervention dependence	3	No intervention currently undertaken but it is needed to maintain populations.
Tolerance	2	Require cold water refuges in summer
Genetic risk	2	Small populations, winter steelhead interactions may reduce viability
Climate change	1	Climate change will severely impact all populations
Average	2	12/6
Certainty (1-4)	3	

Table 2. Metrics for determining the status of NCC summer steelhead, where 1 is poor value and 5 is excellent.

CENTRAL VALLEY STEELHEAD
Oncorhynchus mykiss

Description: Steelhead and rainbow trout are very plastic 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. Adult CV steelhead rarely exceed 60 cm FL, appear silver, sometimes showing an iridescent pink to red lateral line, and have a square-shaped tail 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. 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: For a general discussion of steelhead systematics, see North Coast winter steelhead. Central Valley steelhead are part of the coastal rainbow trout complex that exists in the Central Valley. NMFS (1998) found that Central Valley steelhead formed an Evolutionary Significant Unit (ESU) that was genetically distinct from the Central Coast ESU, which includes fish found in streams tributary to San Francisco Bay. Because an ESU can also include non-anadromous rainbow trout, to clarify the situation the ESU was changed in 2005 to a Distinct Population Segment (DPS), which included only the anadromous forms (see explanation in North Coast winter steelhead account). Nevertheless, there is a conundrum in the relationships among sea-run steelhead and various other rainbow trout in the Central Valley, such as fish that migrate between the Sacramento River and tributaries, resident fish in the main river, and resident fish in tributaries, including those above the major dams. In some instances (e.g., Berryessa Reservoir) there are steelhead-like fish that migrate from the reservoir into tributaries to spawn. There appears to be no major genetic separation among these forms, but there are also no studies to demonstrate conclusively they are all part of one population. However, above-dam forms are now isolated from below-dam forms and presumably are on their own evolutionary pathway, although individuals may be washed downstream from dams. Wild forms may also interbreed with hatchery fish planted in each reservoir although there is not much direct evidence for this. In other systems, it has been demonstrated that there is no reproductive barrier between resident and migratory fish (Zimmerman and Reeves 2000). According to NMFS (2006) "It is unclear how long an *O. mykiss* population can persist if 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.) " Curiously, in an Argentina river, steelhead have developed from resident fish (apparently of California origin) with resident and migratory fish remaining one interbreeding population (Pascual et al. 2001). In some cases (e.g., Calavaras River), the anadromous forms may be mainly female, while males remain as resident fish (McEwan 2001).

The genetic structure of Central Valley populations is complex. In the Bay Area and coastal streams, the above barrier and below barrier populations are more closely related to each other than to those from adjacent drainages. In the Central Valley the above-dam populations apparently show a closer relationship to each other than to populations below the dams (Nielsen et al. 2005). This could indicate a separation from anadromous origin for the above-barrier populations, perhaps from a common hatchery strain being stocked in streams above dams (Lindley et al. 2006). No clear genetic division exists between Sacramento and San Joaquin river populations, indicating a common ancestry in steelhead in these two river systems. However, fish in the American and Mokelumne Rivers reflect a partial Eel River origin of fish propagated in the Nimbus Hatchery (Nielsen et al. 2005).

In this account, we follow NMFS in considering only anadromous rainbow trout as Central Valley steelhead. We do this with some reluctance, because we also recognize that there are interactions among anadromous and non-anadromous segments of Central Valley rainbow trout populations that reflect adaptations to a rapidly changing environment.

Life History: Central Valley steelhead (CV steelhead) exhibit flexible reproductive strategies, which allow for persistence in spite of variable conditions with California's Central Valley (McEwan 2001). The general aspects of steelhead life history are portrayed in Moyle (2002) and in the North Coast winter steelhead account. At present the winter-run steelhead is the only form of steelhead found in the Central Valley of California. There is indication from fish counts before the era of large rim dams that summer-run 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 that are found in mid- to high elevation streams, they were probably extirpated by large dams blocking 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).

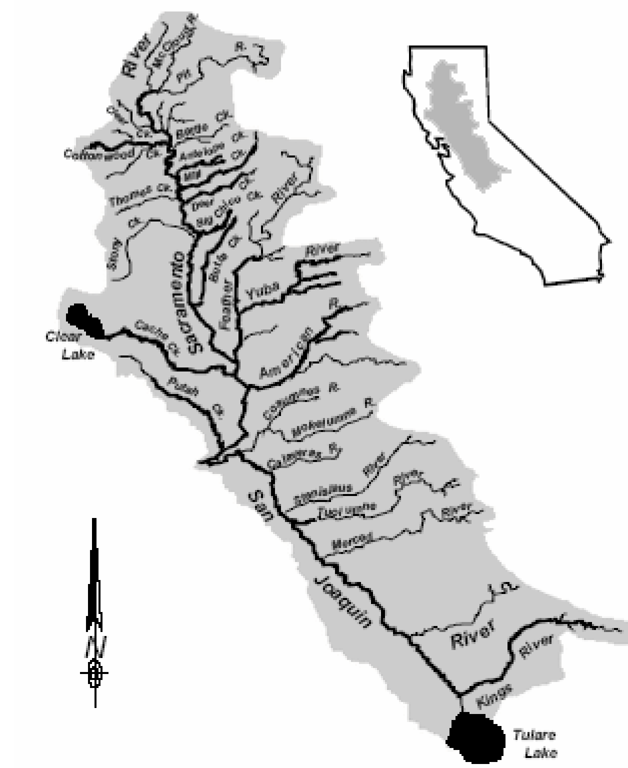
Juvenile CV steelhead are opportunistic, voracious predators on anything available in their rearing streams, from aquatic and terrestrial insects, to small fish, to frogs and mice (Merz and Vanicek 1996, Merz 2002). However, benthic aquatic insect larvae are the mainstay of their diet, especially those of caddisflies (Trichoptera), midges (Chironomidae), and mayflies (Ephemeroptera). Below reservoirs, zooplankton may be important as well. Diets shift with season and size of the juveniles. At times, salmon eggs, juvenile salmon, sculpins, and suckers may be important prey for yearling steelhead (Merz 2002) and may be 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 individual juveniles tended to have relatively limited movement within the rearing areas.

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 if this anadromous life history diversity is still true today or if some steelhead progeny do not go to sea at all. It is possible that some steelhead have adapted to the improved conditions in the Sacramento River for rearing (cold water in summer, abundant food in the form of hatchery salmon fry) and just migrate between river and tributaries, rather than risking migration through the Delta and adverse conditions in the ocean. And some 'steelhead' may not migrate all but remain in the rivers as resident fish. As discussed in other steelhead accounts, the relationship between anadromous and resident rainbow trout is complex but populations that have both forms are likely to have an evolutionary advantage. Anadromous steelhead produce many more eggs than resident fish and improve gene flow among rivers, maximizing genetic diversity. 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 or other natural disasters.

Habitat Requirements: The habitat requirements of CV steelhead are similar to those of Central Coast steelhead, where they are presented in detail. Water quality is a critical factor during the freshwater residence time with cool, clear, and 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 the gravel usually migrate into shallow (<36 cm) areas such as the stream edge 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 in the late summer and fall move into higher-velocity, deeper, mid-channel areas (Hartman 1965, Everest and Chapman 1972, Fontaine 1988). Fry prefer water depths of 25 cm (10 in) to 50 cm (20 in) 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 stronger preference for pool habitats, especially deep pools near the thalweg of the river with ample cover, as well as higher-velocity rapid and cascade habitats (Bisson et al. 1982, 1988; Dambacher 1991). In general, juveniles prefer complex habitat with large physical structures such as boulders, undercut banks, and large woody debris that provide feeding opportunities, segregation of territories, refuge from high water velocities, and cover from fish and bird predators. These features are most characteristic today of small tributaries and they are uncommon in rivers below the major dams. However, it is worth nothing that much of the complex cover in the Sacramento and San Joaquin rivers, and their tributaries, was removed in the 19th century, as part of the 'desnagging' effort to improve channels for navigation. While CV steelhead have been observed spawning in mainstem rivers, it is likely that such habitat is suboptimal for both embryos and young. Nevertheless, Merz (2002) observed good growth and feeding in the Mokelumne River below Camanche Dam and the Sacramento River above Red Bluff supports 'resident' rainbow trout all year around. Thus the generality that CV steelhead primarily used tributaries for spawning and rearing may be at least partially an artifact of large-scale relegation to low elevation rivers before any one was really studying the fish.

Distribution: CV steelhead historically occupied the Sacramento and San Joaquin Rivers and most of their associated tributaries, although they did not occur as high up in many San Joaquin tributaries because of lower natural barriers (Figure 1). Lindley et al. (2006) modeled the likely distribution of steelhead in the Central Valley based on habitat characteristics and concluded that 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 were clearly in areas not accessible to anadromous fish.

The distribution of steelhead 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 spawning and rearing areas (Figure 2). Estimates on the loss of habitat for Central Valley salmonids ranges from 80 to 95 percent (Clark 1929, CACSST 1988, Yoshiyama et al. 2001, Lindley et al. 2006). Non-hatchery stocks of 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, 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).



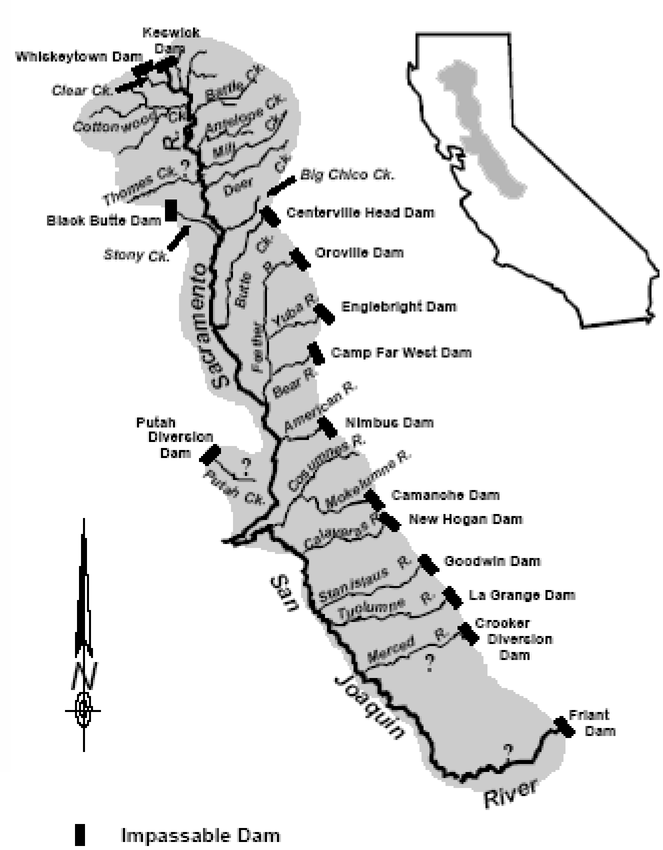


Figure 1 Historical distribution of steelhead in Central Valley drainages. Thick lines represent streams and stream reaches that have documented historical evidence of steelhead. Thin lines represent likely distribution of steelhead based on documented occurrence of Chinook salmon or lack of natural 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).

Abundance There is no good way to accurately estimate the current abundance of CV steelhead today with existing information. Nevertheless, estimates were made in the early 1990s that included hatchery and wild fish (based on Red Bluff Diversion counts, hatchery counts, and past estimates from some tributaries); the estimate was about 10,000 adult fish (McEwan and Jackson 1996). An idea of the apparent precipitous decline of steelhead can be 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 present (Figure 3). If the same trend is happening throughout the Sacramento-San Joaquin system, which is likely, steelhead have declined significantly in the Central Valley. However, the accounts are not particularly reliable as estimates of total numbers.

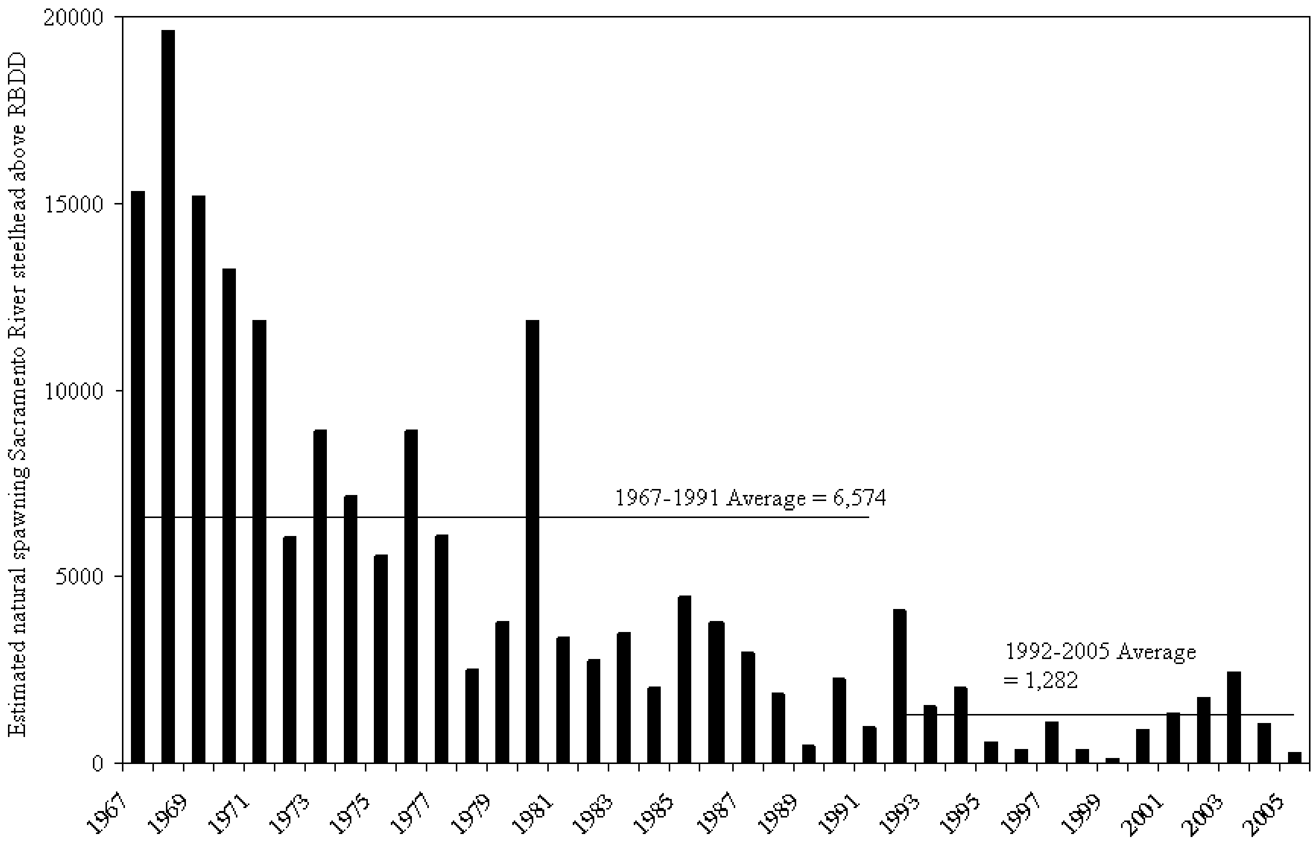


Fig Figure 3. Estimated annual numbers of naturally spawning steelhead in the Sacramento River, upstream of Red Bluff Diversion Dam. Source: http://www.delta.dfg.ca.gov/afrp/

Factors affecting status: Many stressors have contributed to the declining abundance, persistence, and recovery efforts for steelhead 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. 1990, 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 mainly in the Central Valley fall Chinook salmon account. Here only more steelhead-specific causes are treated.

Dams: Probably the single greatest stressor to steelhead has been the loss of access to habitat for spawning and rearing, now above impassable dams. It is likely that somewhere between 80 and 95% of steelhead habitat has been lost. This habitat was mainly smaller tributary streams at higher elevations but steelhead also likely ascended many mainstem rivers to higher elevations than Chinook salmon (McEwan 2001). Even though many dams provide downstream releases for fall Chinook salmon, most do not provide cool temperatures for steelhead during summer and fall months, especially during critical dry periods (drought). The reasons are often complex, but many are just not able to do so because of inadequate release structures or lack of adequate pool storage (McEwan 2001). Where cold releases are present throughout the summer, often resident populations of trout develop, which support tailwater fisheries. Most dams had been built by the early 1960s, so the amount of rearing habitat was static until dam removals on Butte Creek and Clear Creek added a few km of habitat. The more recent declines are most likely a reflection of declining habitat quality, increased water exports, and land use practices that have reduced the relative capacity of existing steelhead rearing areas (McEwan 2001).

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, Nimbus Hatchery on the American River, and Mokelumne Hatchery (McEwan 2001). The fish produced by these hatcheries can have negative effects on CV steelhead in three major ways: displacing wild steelhead juveniles through competition and predation, competition of hatchery adults with wild adults for limited spawning habitat, and hybridization of CV steelhead with fish from outside the basin. The first two effects are well documented for salmonids and may be responsible for estimate that only 10-30% of returning steelhead in the upper Sacramento River are of wild origin (Reynolds et al. 1990). However, it is likely that, in the long run, hatcheries will cause 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.

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) found that Coleman Hatchery and Feather River Hatchery fish are genetically most similar to wild Central Valley steelhead but 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. The extent to which Eel River steelhead have genetically influenced wild populations of CV steelhead is not well documented, but the evidence in Busby et al. (1996) suggests that it is surprisingly small.

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 a detriment to the steelhead population as a whole. It is not clear what effect the incidental catch and release of wild steelhead has on the CV steelhead population as a whole, but some mortality is most likely occurring, which could be deleterious as wild fish numbers continue to decline and a greater percentage of the fish are caught and released.

Conservation: The management of steelhead in the Central Valley is difficult because there is considerable variation in life history patterns as well as interactions with native resident trout and with hatchery steelhead. It has been generally assumed that managing Central Valley rivers for Chinook salmon will benefit steelhead, so steelhead-focused actions are not needed; as a result steelhead management has been relatively neglected (McEwan 2001). Nevertheless, management of rivers to benefit naturally spawning Chinook salmon, especially late fall, winter, and spring runs does usually benefit steelhead by providing both habitat protection and cold-water flows. Nevertheless, management measures that focus on salmon may not fully benefit steelhead, given differences in spawning times and rearing habitats, the apparent need for steelhead to have access to smaller tributaries for spawning, and the effects of 'trout' fisheries.

The lack of solid information on CV steelhead life history, abundance, and interactions with resident and hatchery rainbow trout is a major obstacle to effective and adaptive management of steelhead populations. These problems suggest a conservation program needs to include the following steelhead-oriented elements:

1. Develop a monitoring program that will reliably estimate numbers of CV steelhead entering the Sacramento-San Joaquin River System. This requires special effort because adults migrate over a longer time period than to individual runs of salmon and tend to move upstream when water is high and turbid. Likewise, juveniles often move out to sea at large enough sizes so they avoid screw traps and other standard salmonid sampling devices. In particular, in order to properly analyze steelhead restoration efforts, accurate estimation of wild smolt emigration is needed.
2. Develop a research program that includes:
   a. comprehensive analysis of genetics of rainbow trout above and below barriers in watersheds known to contain CV steelhead. Although some knowledge of sub-basin genetics has been acquired (Nielson et al. 2005), further genetic analysis could benefit managers by helping to determine the origins of migrating fish.
   b. studies of interactions between resident and anadromous rainbow trout in the upper Sacramento River (see below).
   c. studies to determine where steelhead are presently spawning and rearing in the Central Valley and to determine how this habitat has changed with climate change.
   d. use of the San Francisco Estuary by steelhead and ways to increase survival of fish that pass through it.
3. Develop a comprehensive habitat improvement program for steelhead that contains:
   a. flow regimes below dams that have been modified to address the needs of steelhead.
   b. habitat improvements to enhance spawning and rearing habitat, including improved temperature conditions and improve riparian conditions to increase habitat complexity, in both major rivers and tributary streams.
   c. removal of barriers or improvement of passage over barriers to provide more upstream habitat.
   d. improved management of existing streams used for spawning and rearing such as Deer and Mill Creeks in Tehama County, as well as of streams that have potential for use by steelhead.
   e. Improve hatchery management practices to reduce negative interactions among hatchery and wild steelhead, including eliminating use of Eel River strains steelhead in the Nimbus Hatchery.
   f. Improve management of wild resident rainbow trout stocks below the rim dams on the assumption that they are an important component of the steelhead population.

In recent years, funding from CALFED has focused on restoring Clear Creek and Battle Creek tributaries to the Sacramento River, as restoration demonstration streams with a high probability of success. In Clear Creek, barriers have been removed and passage to upstream areas improved. For Battle Creek, dams have been slated for removal, with those left 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. The remediation of passage, diversions, instream gravel mining, instream flows, summer water temperatures, grazing, and riparian restoration should all be considered within this process.

Trends

Short term: The best, if limited, evidence suggests that in the past twenty years the steelhead life history is a declining phenomenon among wild rainbow trout in the Central Valley. How this relates to the status of related wild resident rainbow trout is not known.

Long term: If the short term trends in wild CV steelhead continue, anadromous rainbow trout in the Central Valley may face extirpation as a major phenomenon in the next 50 years. Much depends, however, on their relationship to wild resident fish and on improved management and understanding of both habitat and hatcheries. While resident rainbow trout can apparently redevelop anadromy as a life history strategy (Pascual et al. 2001), much depends on appropriate environmental conditions being present that favor the strategy.

Status: 3. CV steelhead do not appear to be in immediate danger of extinction, although this judgment could change with better information. The score could also be either 2 or 4, depending on the importance of the connection between anadromous and resident population segments in maintaining the steelhead life history pattern and the status of resident populations below the major dams. The high degree of uncertainty suggests that scoring a “2” might be the more conservative option. The DPS was first listed as a threatened species under the ESA by NMFS in 1998 and was revaluated and confirmed in 2005. It is managed by CDFG as a sport fish with limited take.

Metric	Score	Justification
1B Area occupied	4	Multiple populations present in Central Valley but individual viability is not known.
2 Effective pop. Size	2	Does not include resident fish in Sacramento River and tributaries.
3 Intervention dependence	2	Intensive effort required to maintain steelhead life history with appropriate genotype
4 Tolerance	3	Broad physiological tolerances but conditions often unfavorable in big rivers and estuary.
5 Genetic risk	2	Hybridization risk high with hatchery steelhead of Eel River origin and other non-native strains of trout.
6 Climate change	2	Climate change will likely reduce populations but not eliminate many of them, but inability to access historic cold water tributaries makes them more vulnerable
Average	2.5	15/6
Certainty (1-4)	2	Unequivocal data are hard to come by for this taxon

Table 1. Metrics for determining the status of Central Valley steelhead, where 1 is poor value and 5 is excellent.

CENTRAL CALIFORNIA COAST STEELHEAD
Oncorhynchus mykiss

Description: Central California Coast steelhead are anadromous coastal rainbow trout. A description of juveniles and adults is similar to that of steelhead in the Northern California coastal winter steelhead DPS account.

Taxonomic Relationship: The Central California Coast (CCC) steelhead DPS is a complex group of populations inhabiting a region that has been the recipient of 100,000s of out-of-basin juvenile steelhead releases. CCC steelhead have also been used as a source for numerous transfers into the South-Central Coast steelhead DPS (Bjorkstedt et al. 2005). Using microsatellite markers Garza et al. (2004) found collections of juvenile steelhead from CCC streams clustered separately from northern California DPSs, with a closer relationship between these collections and more southerly steelhead populations. Within the Russian River, samples of steelhead show two genetic patterns: mainstem river and headwaters. In the mainstem, collections of steelhead from below natural barriers were not different from each other or from collections from above recently constructed dams. However, six steelhead collections from above natural barriers were significantly different genetically from other populations, suggesting long term isolation and limited genetic diversity (Deiner et al. 2007). Other populations from watersheds north of and including San Francisco Bay clustered together in an analysis of microsatellite DNA variation. Further to the south, CCC steelhead samples have been shown to be phylogenetically intermingled with South-Central California Coast steelhead, likely due to out-of-basin transfers and translocations between these DPSs (Bjorkstedt et al. 2005).

Life History: CCC steelhead trout show a tremendous amount of juvenile and adult life history variation, though all adult runs occur during the winter. Shapovalov and Taft (1954) identified 32 different combinations in the amount of time 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 April. CCC steelhead enter rivers in reproductive condition and spawn soon after reaching spawning grounds. Most spawning, however, typically occurs during late spring, avoiding damaging effects of winter floods, common to the coastal watersheds 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 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. CCC 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 in the gravels and Shapovalov and Taft (1954) estimated hatch time to be 25-35 days, with emergence of fry after 2 to 3 weeks for alevin development. When steelhead spawn later in the winter, warmer water temperature promote rapid alevin development, reducing redd stranding as stream flows drop. 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 trout move into the water column and utilize deeper water as they grow.

On Waddell Creek, Shapovalov and Taft (1954) observed a bimodal emigration pattern by juveniles, although they moved downstream during all seasons of the year. Peaks in emigration were in early January and mid-March. Older age classes of juvenile migrated earlier than young-of-year trout. (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. Both of these types of fish typically spent one to ten months rearing in the estuary prior to ocean entry. The third pathway observed by Hayes et al. (2008) was for juveniles to rear 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.

Smoltification of juvenile steelhead often occurs after fish reach a large size (100mm FL). Smith (2002) found favorable conditions for rapid growth in productive lagoons at the mouths of streams and in stream reaches with high summer flow in Waddell Creek and the San Lorenzo River steelhead typically had to reach age 1+ years before they were large enough to become smolts. Due to potentially restrictive summer habitat requirements, 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 1+ steelhead present in the upper Scott Creek watershed, possibly due to low flows and nutrient inputs found under the redwood canopies (Romero, Gresswell et al. 2005; Hayes 2008). Based on size data (Hayes et al. 2008), juvenile steelhead in Scott Creek appeared to emigrate out of the upper watershed before age 2 (150mm), although these fish often took advantage of the rapid growth achievable in estuaries.

Estuaries along the Central California Coast are variable in size, but tend to undergo sandbar formation and become seasonal freshwater lagoons during summer low flow conditions. These areas constitute small portions of steelhead habitat, but seem to be a critical nursery habitat for juvenile steelhead. The Russian River estuary does not always close to the ocean and juvenile steelhead increase in size until mid summer then decrease in size, suggesting young-of-year (YOY) continue to enter the estuary while larger smolts either emigrate or move upstream (Cook et al. 2005). In the Russian River estuary, steelhead preferred the middle and upper portions of this habitat and were almost exclusively captured at confluences with tributaries (Cook 2005). In Scott Creek, Bond (2006) found juveniles emigrated into the estuary at all sizes, but larger smolts had a higher survival rate. YOY juvenile trout remained in the estuary until it became a closed freshwater lagoon. These fish experienced high growth rates, which resulted in a doubling of fork length (mean FL of fall lagoon resident- 206mm FL). The growth rate of juvenile trout in the estuary varied among years and appeared to be density-dependent (Hayes 2008). Juvenile steelhead in Scott Creek that are larger than 150 mm FL have significant survival advantage in the ocean. Bond (2006) found they comprised 85% of the returning adult population though they comprised less than 50% of the juvenile population in the estuary, which included 0+ and 1+ fish.

Habitat Requirements:

CCC Steelhead require similar freshwater spawning and rearing sites as described in NC steelhead account. Leidy (2007) found the abundance of CCC steelhead juveniles in the San Francisco Bay Area was *positively* correlated with elevation, stream gradient, dominant substrate size, and percent native species, but *negatively* correlated with stream order, average and maximum depth, wetted channel width, water temperature, water clarity, percent open canopy, conductivity, percent pool habitat, and the total number of fish species (Leidy 2007). This indicates that they were mainly found in small, cold water streams, where pools were few, which may be partially an artifact of the urbanization of the lower reaches of the streams. Apparent limiting habitat in streams is often over-summering habitat for yearling steelhead. These fish require deep water with overhead cover for protection from predators. Stream flows must provide for annual lagoon bar failure so adult spawners can migrate upstream to reproduce and juveniles can emigrate for foraging in the estuarine and ocean environments.

Like other salmonids, CCC steelhead require cool water, though these fish manage to grow in warmer water conditions. The optimal temperature range for juvenile steelhead growth is 15-18ºC (Moyle 2002). While cool water is typically found in headwater regions of CCC, steelhead distribution and within the marine-influenced coastal regions of watershed, 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 habitat presumably provides heterogeneous thermal habitats, where steelhead can move between cooler and warmer habitats. 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 in California streams in the Russian River and south to Aptos Creek. 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. Currently, steelhead remain in 82% of historically occupied watersheds in the CCC steelhead DPS region. Spence et al. (2007) identified five regions within the CCC steelhead DPS with similar basin-scale environmental and ecological characteristics. Eleven watersheds across these regions were found to historically have contained sufficient habitat and limited level of gene flow to support independently viable populations, while another 26 watersheds had conditions which may have support independently viable populations.

The CCC steelhead DPS is dominated by two large populations centered on the Russian River and San Francisco Bay (Spence et al. 2007). In the *Interior Region*, the upper Russian River mainstem reaches above 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 by 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 San Leandro, San Lorenzo, Coyote, and Alameda Creeks. Additionally, functionally independent populations are found south of the Golden Gate in the *Santa Cruz Mountains Region* including the San Lorenzo River and San Gregorio and Pescadero Creeks. In the San Lorenzo River, a majority of spawning occurs above the town of Boulder Creek (Johansen 1975). Numerous small coastal and San Francisco Bay tributaries contain historically small populations but lacked sufficient habitat for a self-sustaining population.

In the ocean, CCC steelhead presumably stay close to their home coastline, though evidence is limited. Only a few CCC steelhead have been captured in trawl surveys along the Oregon and California coast (Brodeur et al. 2004), but this may be due to the lack of tagging efforts for CCC steelhead. If the southern steelhead populations are similar to northern steelhead populations, which are highly pelagic, it is possible that these fish migrate into the north Pacific as well.

Abundance: Information about CCC steelhead abundance is very limited but numbers appear to be considerably lower than historic estimates throughout the region. Current estimates are approximately 14,100 adult steelhead per year, on average (NMFS 2006). During the early 1960s, CDFG (CDFG 1965) estimated 94,000 steelhead spawned in this DPS, with the majority of spawning occurring in the Russian River (50,000) and San Lorenzo River (19,000). Tributaries in Marin and San Mateo Counties were estimated to each contain 8,000 spawning steelhead annually, while Sonoma and Santa Cruz Counties contributed about half as many steelhead each annually. The Russian River was probably once the third largest steelhead river in California. Steelhead abundance in the Russian River declined from an estimated 50,000 in the 1960s to 1,750-7,000 in the 1990s (Busby et al. 1996; Good et al. 2005), indicating a potential decline of at least 89%. The steelhead run in Lagunitas Creek is believed to have been about 500 fish annually during the early 1990s (McEwan and Jackson 1996) and between 15 and 136 redds were observed between 2001 and 2005 (Ettlinger et al. 2003; 2004; 2005).

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, abundance also appears to be less than 15% of levels from only thirty years ago (Good et al. 2005). Creel surveys along the San Lorenzo ranged from 1,895 to 5,645 steelhead caught in 1953 and 1954, and between 1035 and 1816 captured between 1970 and 1973 (Johansen 1975). Information about run sizes in other watersheds is sketchy; the most recent estimates for San Vincente, Scott, Soquel, and Aptos Creeks are all below 300 fish annually.

Juvenile abundances are highly variable annually and geographically. In the Lagunitas Creek drainage, 1.51 steelhead trout per meter were found on electrofishing surveys (Emig 1985). Further south in the DPS, juvenile trout sampling in Waddell Creek found densities to range from 4 to 33 fish/m^2, depending on location. Although there are numerous difficulties with using juvenile data, Good et al (2005) reviewed trend data for juvenile steelhead trout from the San Lorenzo River, Scott Creek, Waddell Creek, Gazos Creek, and Redwood Creek. All of these populations except the San Lorenzo were classified by Spence et al. (2007) as potentially dependent populations, thus the trend observed in these data likely does not reflect demographically independent populations. Overall, all five datasets demonstrated downward trends in juvenile abundance (Good et al. 2005).

Factors affecting status:

Small populations of steelhead still occur in watersheds throughout the DPS range, but they are limited by a wide variety of factors included in four broad categories (1) dams and other barriers, (2) degradation of stream habitat, (3) degradation of estuarine habitat and (4) hatcheries. Other factors not discussed here include pollution, gravel mining, fisheries, and a litany of other factors that affect many steelhead streams up and down the entire California coast. All of these factors combined make it much more difficult for CCC steelhead populations to cope with the high natural variability of rainfall and other climatic conditions, as well as fluctuating ocean conditions. These cumulatively will make it much more difficult for the fish to resist the potential effects of climate change.

Dams and other barriers: Across the CCC steelhead DPS, barriers have reduced the amount of accessible habitat for juvenile and adult habitats. This is important because steelhead tend to rear and spawn in smaller headwater tributaries in upper portions of watersheds (Bjorkstedt et al. 2005). For the DPS as a whole, 22% of historical habitat is estimated to be behind recent (usually human-made) barriers (Good et al. 2005). In the Russian River, Coyote and Warm Spring dams both block historic habitat. On the Russian River, 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%). North of the Bay, accessibility is also a problem in Lagunitas Creek (49%) and Walker Creek (26%).

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. All diversions typically reduce summer flows, reducing habitat and increasing water temperatures, making it more difficult for steelhead to survive through the warmer months. 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 but the effects of this on steelhead are not known.

*Degradation of stream habitat***:** Degradation of habitat in most watersheds and estuaries supporting populations 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 (CWA) due to high levels of sedimentation, aggravated water temperatures, presence of pathogens, and generally poor water quality. Similar conditions exist in the San Francisco Bay area where CWA-listed impaired watersheds include Guadalupe, San Francisquito, Stevens, and Sonoma Creeks as well as the Petaluma and Napa Rivers. Similar sedimentation problems due to agricultural and logging practices have led to the CWA listing of San Mateo County coastal steelhead creeks (Pomponio and Pescadero Creeks). 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, the necessary inputs of large wood are being eliminated further reducing cover and pool formation, and increasing conditions unfavorable to juvenile steelhead. Degraded habitat can favor alien species, which can increase predation pressures on juvenile steelhead (Leidy 2007).

Degradation of estuarine habitat: CCC steelhead seem unusually dependent on the estuaries (lagoons) at the mouths of their streams for growth and survival. These habitats are shrinking as they fill with sediment from upstream and are encroached by urbanization and agriculture. 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. In addition, the natural summer sand barriers are frequently artificially breached, resulting in sudden draining of lagoons and large-scale reduction in habitat (Moyle and Smith 1995). Highway 1 also impacts almost every estuary in the CCC steelhead DPS, due to channelization and bridge construction for roadways.

Hatcheries: There are currently two artificial propagation programs for CCC steelhead: the Don Clausen Fish Hatchery (Dry Creek, Russian River) and Kingfisher Flat Hatchery (Scott Creek). While these may contribute to future abundance and spatial structure, neither are located in watersheds that supported steelhead populations that are viable in isolation (functionally independent populations), thus the success of these operation in supporting recovery goals is questionable. Due to the low number of wild spawners expected in the limited available natural habitat, it is more likely that domestication selection will reduce genetic diversity and effective population size in these watersheds, and then in locations where the natural spawning population is larger. Additionally, the influence of past frequent plants of hatchery steelhead from out-of-basin is not well understood.

Conservation: The Federal Recovery Outline for CCC steelhead was released in 2007 (NMFS 2007). Previous work designated approximately 1,465 miles of stream and 386 miles of estuary as Critical Habitat for CCC steelhead (NMFS 2005). This draft recovery plan recognizes the diversity of factors causing decline of the CCC steelhead and indicates that a lack of state protection efforts is a factor influencing their status. For example, the plan identifies the California Forest Practices Rules as being inadequate for protection of riparian habitat, which shades the streams, naturally limits sedimentation, and provides inputs such as large woody debris for habitat. Likewise, the plan identifies the stalling of CDFGs statewide coastal salmonid monitoring program as a factor preventing the gain of comprehensive abundance and trend information for the DPS. The CDFG salmon and steelhead stock management policy is identified as an important conservation document, though its work plan has yet to be accomplished. Essentially significant protection for CCC steelhead can be accomplished by state agencies moving forward with actions based on programmatic documents already developed.

The solutions needed are simultaneously local and widespread, small-scale actions in the context of improved watershed management, such as addition of large wood into a stream reach, maintaining adequate riparian buffers, and limiting sediment and other pollutants flowing into stream to cumulatively benefit recovery of CCC steelhead. Thus, restoration guidelines have been developed by NMFS for bank stabilization, road maintenance, and instream gravel mining. To enhance the summer and overwintering survival of CCC steelhead, improvement in the stream complexity, as well as, recruitment and retention of large wood is important. At a larger scale, 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. At a still larger scale is the need to manage entire watersheds in a coordinated fashion to reduce human impacts on the streams and estuaries. It is especially important to regulate releases from dams so that a more natural flow regime can be instituted in creeks and rivers. This is a large undertaking but without it, the populations will continue to decline.

Trends:

Short term: Juvenile abundance data indicate a downward trend in populations in recent years at five locations: the San Lorenzo River, Scott Creek, Waddell Creek, Gazos Creek, and

Redwood Creek in Marin County (Good et al. 2005). Although an overall reduction in juvenile abundance is implied by this analysis, it is unclear how such a reduction ultimately affects numbers of returning adults. In lieu of abundance data, information on available habitat can provide insight about population status and most streams in the CCC steelhead region are listed as impaired in one way or another. There is little sign of major habitat improvement, despite many local efforts, so CCC steelhead populations must be assumed to still be declining.

Long term: It is clear that CCC steelhead runs have declined by 80-90% in the past 50 years and that the decline is continuing. The NMFS draft recovery plan states that CCC steelhead have only a low to moderate potential for recovery due to urbanization across their range. In the Russian River, agriculture continues to require more water, which is delivered via the inter-basin transfer of instream flows from the Eel River to the Russian River. The pressures of agriculture and urbanization are not likely to be reduced. 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. The plasticity of life history strategies observed in CCC steelhead will likely guarantee their presence in the larger watersheds they inhabit, but it is likely extirpation of steelhead from most currently occupied watersheds will occur over the next 25-100 years, unless large-scale actions are taken. Climate change will exacerbate the decline by increasing temperatures beyond lethal limits in unprotected streams and increasing demand for scarcer water.

Status: 3. This is an optimistic designation because some populations (e.g., Russian River) seem to be large enough to be sustainable. However, every indication is that trends in all populations are downward and will be accelerated by climate change. CCC steelhead were listed as a threatened species on August 18, 1997; their threatened status was reaffirmed on January 5, 2006 (NMFS 2006). They have no special status in California except as a sport fish with limited take.

Metric	Score	Justification
1B Area occupied	3	Multiple watersheds occupied in California but probably <10 viable populations still exist.
2 Effective pop. Size	3	The Russian River likely contains >1000 spawners annually with smaller contributions from other populations but numbers are declining
3 Intervention dependence	3	Habitat restoration and barrier removal are critical to increasing habitat availability
4 Tolerance	4	Able to live in freshwater and estuarine environments
5 Genetic risk	3	Widespread but populations increasingly fragmented and isolated, with potential for interbreeding with non-native strains.
6 Climate change	1	Extremely vulnerable in all watersheds because of stress from other factors (urbanization, etc.)
Average	2.9	17/6
Certainty (1-4)	3	Hard numbers are few but status is fairly certain.

Table 1. Metrics for determining the status of Central California Coast steelhead, where 1 is poor value and 5 is excellent.

SOUTH-CENTRAL CALIFORNIA COAST STEELHEAD
Oncorhynchus mykiss

Description: South-Central California Coast (SCC) 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 by their genetic identity and distribution.

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, which is evident within the SCC steelhead DPS. Garza et al. (2004) studied 41 collections of steelhead from across California and constructed genetic trees showing that collections from each basin were fairly distinct with relatively small amounts of genetic exchange with neighboring basins. However, clear genetic differences between South-Central Coast Steelhead and southern steelhead were not apparent. Unlike other regional-scale genetic differentiation where each DPS occupies a relatively distinctive branch of the steelhead family tree, the sample collections from the SCC DPS and further south in the southern steelhead DPS were genetically intermixed (Girman and Garza 2006), suggesting that these DPSs are more similar to each other than to steelhead DPSs further north. The SCC DPS therefore seems to exist mainly for management convenience, breaking along the historic boundaries of the original ESUs (Evolutionarily Significant Unit) used to describe the forms.

Aguilar and Garza (2006) used a molecular marker to evaluate natural selection within 24 collections of steelhead from along the coast of California. They observed that a genomic region associated with thermal tolerance and spawning time may have been under selective pressures in collections from Waddell Creek and Chorro Creek in the SCC steelhead DPS. Boughton et al. (2006) reported that rainbow trout are found above artificial barriers in 17 of 22 basins in the South-Central/Southern California Coast steelhead DPS. In the Salinas and Arroyo Grande watersheds, a genetic comparison of trout above barriers and juvenile steelhead below barriers demonstrated these collections were closely related and that there was not substantial divergence above and below recent barriers (Girman and Garza 2006).

Based on this genetic information and distributional information, Boughton et al (2006) identified 41 historically independent populations of SCC steelhead in the DPS, including three populations in the Salinas River. Three populations are recognized in the Salinas River due to its large size, which likely allows sufficient geographic isolation to maintain multiple populations (Boughton et al. 2006). These three populations each contain spawning areas separated by the mainstem Salinas River, and one grouping includes steelhead found in the Nacimiento, San Antonio, and upper Salinas rivers. These 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 SCC steelhead, although they appear to express a diversity of life history patterns similar to other steelhead (see NC and CCC steelhead accounts). SCC 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. SCC steelhead and CCC steelhead encounter similar physical habitat features that bound the trajectories of their juvenile life history. These features include principally small, steep coastal watersheds that reduce juvenile growth and age at outmigration, as well as seasonally-open estuaries, which influence smoltification, marine survival, and migration patterns.

The potential for steelhead to make life history switches between adult life histories has been demonstrated for anadromous and resident fish in Oregon populations (Zimmerman and Reeves 2000). Observations of SCC steelhead's phenotypic plasticity include inland resident juvenile trout exhibiting smolt characteristics and the production of smolts in watersheds without returning adult steelhead (Boughton et al. 2007). Adult steelhead likely are found as far south as northwestern Mexico in the ocean and appear to be more solitary than other salmonids (Busby et al. 1996; Good et al. 2005). Adult steelhead return from the ocean to enter watersheds to spawn in SCC stream between January and May (Boughton et al. 2006). SCC 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 water will be impossible. Presumably under such circumstances the adults spend another year in the ocean before returning to try again and older juveniles suffer high mortality.

SCC steelhead display a high degree of life history plasticity. Beyond the three categories of juvenile steelhead life history strategies discussed in the CCC steelhead account, SCC steelhead may use finer-scaled habitat switching, making intraseasonal movements between lagoons and freshwater and within freshwater movements between reservoirs and tributaries (Boughton et al. 2006). Immature steelhead may spend several weeks to months in estuaries prior to entering the ocean. In cases where larger basins are occupied by SCC steelhead (e.g. Pajaro, Salinas Rivers), juvenile life history patterns are influenced by the necessity to emigrate due to desiccation of tributary streams in dry years, which eliminates low elevation reaches of these streams as over-summering habitat. Fish may be forced to move upstream into headwater areas with perennial flows or to emigrate downstream to the estuary. Mainstem rivers in the SCC steelhead DPS are too warm for steelhead from the late spring through summer and are primarily used as migration corridors. SCC steelhead juveniles presumably grow more during the winter and spring in fresh water when temperatures are optimal bioenergetically, while summer and fall seasons see little growth due to water temperatures being at the upper limits of their physiological tolerance.

Habitat Requirements: South-Central California coast steelhead have habitat requirements similar to those of steelhead populations further north. They need cool, flowing waters, access to the ocean, and available food items. These requirements can be difficult for SCC steelhead to find. Optimal mean monthly temperature in potential rearing areas are between 6ºC and 10ºC, with temperatures over 13ºC being considered poor (NMFS 2007, see southern steelhead for more details on temperature requirements). Thompson et al. (unpublished) studied juvenile steelhead habitat in the Salinas River found no steelhead at sites where the maximum temperature exceeded 26ºC or where the mean temperature exceeded 21.5ºC. A key component of SSC steelhead habitat in the Salinas basin is large woody debris, made up mostly of hardwood trees, often still alive. Often, mainstem river and lower reaches of tributary creeks are seasonally dry and these reaches are primarily used as migratory corridors. In cases when large wood provides oversummering habitat, SCC juvenile steelhead will use mainstem creeks and rivers with perennial flows. These creeks may be important 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 possible. Boughton et al (2006) presented a similar result that after rains subsequent to a drought juvenile steelhead were observed virtually immediately in wetted segments using snorkel presence/absence surveys. Thus, sufficient habitat with perennial flows and cover are critical requirements for juvenile rearing and full expression of life history variation.

The potential for catastrophic natural events, including wildfire, drought, and debris flows, to negatively impact habitat availability for SCC steelhead is considerable. Since these events have the potential to extirpate populations within the SCC steelhead DPS, they each directly affect the viability of steelhead within the four SCC steelhead DPS biogeographic groups (Boughton et al 2007).

Distribution: SCC steelhead are distributed between the Pajaro River south to (but excluding) the Santa Maria River. This is nearing the southern limit of anadromous rainbow trout in North America. Although habitat quality is low and population sizes in coastal basins seem small for persistence, steelhead are currently found in almost all SCC DPS coastal watersheds in which they were historically present (Boughton et al. 2005). Steelhead have also been found in a number of basins in the SCC DPS with no recent historic records of steelhead, including Los Osos, Vincente, and Villa Creeks, illustrating the opportunistic nature of the species in an unpredictable landscape (Boughton et al. 2005).

Watersheds in this DPS occupied by steelhead are separated into four biogeographic regions that are categorized by migration connectivity and reliability, summer climate refugia, intermittence of stream flow, and winter precipitation (Boughton et al. 2007). In the *Big Sur Coast* and *San Luis Obispo Terrace regions*, 37 streams contain steelhead and bear more ecological resemblance to steelhead streams in northern California (J.J. Smith, pers. comm.) than to streams in the interior regions of the DPS. These watersheds are ocean-facing and subject to marine-based weather patterns. The other two SCC steelhead regions include rivers which cut across the coastal ranges and extend inland through valleys. These include the Pajaro River, Gabilan Creek, Arroyo Seco, Southwest Salinas Basin, and Carmel River. These watersheds are part of the *Interior Coast Range and Carmel River* regions, are principally in coastal rain shadows, and have warmer seasonal climates.

Abundance: Historically, annual runs totaled more than 27,000 adults (NMFS 2007) in the SCC steelhead DPS. CDFG (1965) suggests that the DPS-wide run size was as high as 17,750 adults in 1965. Good et al. (2005) reported less than 500 adults returning annually to each of the Pajaro, Salinas, Carmel, Big Sur, and Little Sur Rivers in 1996. CDFG (1965) estimated these same runs consisted of about 4,750 adults annually during the 1960s. Thus, it appears interior regions of the SCC steelhead DPS including the Pajaro, Salinas, Nacimiento/Arroyo Seco, and Carmel Rivers have experienced declines in run sizes of 90% or more (Boughton et al 2007).

Very little population monitoring data exists for SCC steelhead. The one time series that exists is from the Carmel River. Adult steelhead counts on the Carmel River at San Clemente Dam have ranged from 0 to 1350 between 1962 and 2002 with an average run size of 821 adults (Good et al 2005, MPWMD 2007). Although steelhead in the Carmel River underwent a drastic decline that lasted into the late 1980s, the recent trend data for this population indicates it is rebounding, apparently due to intensive habitat management efforts that improved juvenile growth rates as well as the propensity of fish to smolt at a larger size (Good et al. 2005). These larger smolts may then have benefited from higher ocean survival with positive influences on the next generation of steelhead (Bond 2006). Overall, it is reasonable to assume that the total number of SCC steelhead spawners throughout their range in a fairly wet year is considerably less than 5,000 fish, probably more on the order of 2,000 spawners.

While data from surveys of juvenile steelhead are difficult to evaluate in the context of run size and viability, this type of data exists from a number of watersheds and is an indication of habitat integrity. In Big Creek, a spring fed watershed along the Big Sur coast, juvenile steelhead data indicate that a fair number of 1+ and 2+ steelhead inhabit the lower reaches during summer. It is possible that higher rainfall, lower air temperatures, and perennial flows in watersheds of the Big Sur Coast Region allow populations in this region to persist despite limited habitat area in the smaller watersheds. Thus, although steelhead populations within the SCC steelhead DPS have declined dramatically, about 90% of historic habitat continues to be occupied. The resilience of SCC steelhead along the South-Central California coast may reflect favorable oversummering conditions with sufficient cover and perennial flows in coastal populations benefiting 1+ and 2+ steelhead survival.

Factors affecting status: NMFS identified seven principal natural threats to steelhead in their Draft Recovery Outline for the SCC steelhead DPS (NMFS 2007): (1) alteration of natural stream flow patterns, (2) physical impediments to fish passage, (3) alteration of floodplains and channels, (4) sedimentation, (5) urban and rural waste discharges, (6) spread and propagation of alien species, (7) and loss of estuarine habitat. For more specific discussion of these and additional factors such as fire and drought, see the southern steelhead account.

The threats posed by alteration of the terrestrial and aquatic systems are principally associated with human activities in the larger watersheds in the range of the SCC steelhead such as the Pajaro, Carmel, Salinas, Arroyo Seco, San Antonio, and Nacimiento Rivers. There has been extensive loss of habitat in these areas due to agriculture and urbanization, resulting in the dewatering of streams, modification of river and creeks channels, and addition of toxic materials. Water development (surface and groundwater) has reduced the frequency, duration, timing, and magnitude of flows. High flows in particular are critical for breaching of lagoon mouths, adult steelhead spawning migration and timing, and juvenile steelhead emigration. The encroachment of agricultural, industrial, and residential developments into riparian and floodplain channels of SCC steelhead rivers and creeks has caused serious declines in population due to loss of riparian cover, modification of river channels, and lack of vegetation to maintain suitable stream temperatures, food resources, and oversummering habitats for juveniles. A significant portion of spawning and rearing habitat has been rendered inaccessible as a result of dams and diversions on most of the rivers, which reduce flows, alter downstream habitats, and block or impede migration. While many of these threats also influence the smaller coastal tributaries in the SCC steelhead DPS, many of these watersheds are on public lands or in areas with less human development so are more able to maintain populations of steelhead.

Conservation: Critical habitat listing for the SCC steelhead was issued on September 2, 2005 (NMFS 2005). Within 30 occupied watersheds, 2000 km (1,250 miles) of stream habitat and 7.7 square km (3 sq miles) of estuarine habitat were designated as critical habitat. Despite identification of critical habitats, continued human population growth continues to intensify development of land and water resources within them. The inadequacy of federal and state regulatory mechanisms has allowed steelhead habitat to be damaged repeatedly, protected ineffectively, and managed inconsistently for recovery of the steelhead (NMFS 2007). Here are three examples:

1. The Los Padres National Forest Plan does not include sufficient provisions for protection and restoration of aquatic habitats important for all life history stages of steelhead. This is essential given the importance of resident fish on public land to the viability and recovery of SCC steelhead.

2. In an effort to protect residential development, federal agencies which influence the development of waterways and floodplains have set standards which do not reflect the highly variable geomorphic and hydrologic nature of South-Central California watercourses. SCC steelhead are adapted to persist in these highly variable physical environments with wide riparian buffers and floodplain channels. Residential development has heavily encroached into this area, which reduces habitat and increases risk to humans. Agencies such the U.S. Army Corps of Engineers and Federal Emergency Management Agency do not have a process in place to effectively balance the continual development of water resources with recovery of SCC steelhead and a healthy, natural, and variable aquatic ecosystem.

3. Although NMFS and the California Department of Fish and Game have produced a Coast-Wide Anadromous Fish Monitoring Plan it remains unfinished and funding has not been identified or secured to support this program. This monitoring plan is critical to data collection necessary for the assessment of SCC steelhead populations and habitat.

Not surprisingly, NMFS (2007) gives the SCC steelhead DPS only moderate potential for recovery. A critical step in the recovery strategy for SCC steelhead will be securing passage and refuge habitat for a core set of populations (NMFS 2007). Additional steps in a recovery strategy include:

- Secure extant parts of the Interior Coast Range and Carmel Basin regions.
- Identify and maintain sustainable refugia against severe droughts and heat waves.
- Collect annual population data.
- Secure and improve estuarine/lagoon habitat.
- Develop a strategic balance and timeline for investment in better information vs. investment in more recovery activities.
- Establish programs for ecosystem-based management of sediment regimes and hydrographic regimes.

Although a number of small populations seem to persist along the Big Sur Coast, any recovery effort will need to focus on larger watersheds within the SCC steelhead range because viability of a population increases with population size; these are the core populations most likely to meet viability criteria (Boughton et al. 2006). Core populations should be multiple and well dispersed. Smaller non-core populations are also needed for aiding in dispersal and connectivity across the SCC steelhead DPS. The limited number fish returning to streams within the *Interior Coast Range* and *Carmel Basin regions* indicates that mainstem restoration may be necessary for maintaining viability among the core populations in the DPS. In particular, recovery will require providing sufficient flow and perennial fish passage in these streams.

Climatic change and stochastic events (e.g., wildfires) will have an influence on SCC steelhead recovery. Although a majority of extirpations in the SCC and southern steelhead DPSs have been associated with anthropogenic barriers, 32% appear correlated with mean annual air temperature (Boughton et al. 2005). As air temperature increases into the future, extirpations related from this factor will shift northward into the SCC steelhead DPS and the likelihood of wildfire will likely also increase. Increasing monitoring of SCC steelhead populations will assist with development of a strategy to adapt to climate change.

Another potential impact of climate change is 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. J. Smith, San Jose State University, pers. comm.). Research on CCC steelhead indicates these habitats are critically beneficial to productive steelhead runs (Bond 2006; Hayes in press) and similar research should be undertaken to assess the importance of the lagoon-anadromous life history form to the viability of SCC steelhead. Due to the small size and coastal location of estuaries in the SCC steelhead DPS, these areas have been subject to intense pressures from human developments, water use, and pollution.

Numerous beneficial actions can be taken fairly quickly to reduce the threats of limited spatial distribution and low productivity of SCC steelhead (NMFS 2007). For example:

- Further research on SCC steelhead life history and habitat requirements can guide recovery actions and provid3 a basis for hypothesis-driven understanding of the biological and physical constraints for steelhead recovery.
- Completing and implementing fish barrier removal projects in smaller coastal streams (i.e. Arroyo Grande Creek) and larger interior rivers (Carmel, San Antonio, Nacimiento Rivers) will provide access to historic habitat and increase population sizes.
- Providing flows in the Salinas and Pajaro River systems to support establishment of functioning riparian corridors and floodplain habitats should greatly increase the spatial distribution and productivity of SCC steelhead.
- Additional training of regulatory agencies and biologists working in the SCC steelhead region to aid recovery by protecting stream corridors, facilitating assessment of waste discharges (sediment, pesticides, and other non-point source pollutants), and by reducing the filling in, artificial breaching, and draining of estuaries.

Trends:

Short term: SCC steelhead continue to persist in most of their historic watersheds. In fact, three basins with no historic record of steelhead have been shown to be occupied (Good et al. 2005). However, most populations are very small and may not be able to persist in the long term (50-100 years). While the amount of habitat available for oversummering is greatly reduced during dry periods in numerous watersheds, evidence suggests that during wet years spawning may occur in a broader range of tributary streams than initially believed, based on seasonal drying of streams (L.C. Thompson, UC Davis, pers. comm. 2007). The DPS's sole time series of adult returning steelhead, from the Carmel River, shows an overall downward trend in returning adults, although a recent positive trend suggests the Carmel River steelhead population is rebounding. It is unclear what mechanism is driving the Carmel River's population increase, but it may be due to a substantial immigration of straying steelhead and/or intensive fisheries management that has included greater stream flows, improved passage, and recovery of riparian habitats, which may have improved reproduction and survival.

Long term: It is a tribute to the resilience of SCC steelhead that populations have managed to persist in the face of rapidly increasing human populations, accompanied by increased demand for the water they require for persistence. Limited data from the larger watersheds suggest that in past 50 years or so, total steelhead numbers have declined by 90% or more. Climatic regimes will heavily influence oversummering juvenile survival in interior regions of the SCC steelhead DPS and these pressures will intensify with rural instream withdrawal and groundwater pumping. The continuing increase in human populations in the region, coupled with climate change changing rainfall patterns and increasing water temperatures, means that long term (>100 years) persistence is most streams is not likely without large-scale intervention. A possible exception may exist in the larger streams along the Big Sur Coast (e.g., Big Creek, Big Sur River) which still benefit from the summertime cooling effect of ocean proximity.

Status: 2. A majority (possibly all) of SCC steelhead populations are likely to be extinct within 50 years without serious intervention (Table 1). SCC steelhead were listed as a threatened species by NMFS in 1997. They are considered to be a Sensitive Species by USFS and a Species of Special Concern by the California Department of Fish and Game. SCC steelhead are threatened by increasing human land and water development, as well as climate change, wildfire, and drought. These impacts may be insurmountable without both 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 agriculture need to be implemented by private landowners and industrial water users to conserve and restore floodplain and riparian habitats along mainstems and tributaries. NMFS (2007) identified extensive public education, development of cooperative relationships, and interagency collaboration as critical to recovery of SCC steelhead. These steps are necessary to ensure that funding and strategic planning result in effective, sustained implementation of SCC steelhead recovery efforts.

Metric	Score	Justification
1B Area occupied	3	Multiple watershed occupied, although not indefinitely
2 Effective pop. Size	2	Most populations probably contain <100 spawners
3 Intervention dependence	2	Habitat restoration and barrier modification projects critical for recovery. Most populations will require reconnection of resident and anadromous populations in the near future to boost them to sustainable levels.
4 Tolerance	3	Moderate physiological tolerance, iteroparity uncommon
5 Genetic risk	3	Limited gene flow among populations may make them vulnerable to inbreeding and other effects.
6 Climate change	1	Affects will be exacerbated by human population growth
Average	2.3	14/6
Certainty (1-4)	3	Little monitoring of most populations

Table 1. Metrics for determining the status of South Central California coast steelhead, where 1 is poor value and 5 is excellent.

SOUTHERN STEELHEAD
Oncorhynchus mykiss

Description: Southern steelhead (Southern California Coast steelhead DPS) are similar to other steelhead. For a full description see the Northern California coastal winter steelhead DPS account.

Taxonomic Relationships: Southern steelhead are anadromous coastal rainbow trout and are the southernmost anadromous salmonid in the United States. The southernmost rainbow trout are populations of resident trout in headwaters of the Rio Santo Domingo in Baja California, Mexico and in several watersheds of north-central Mexico (Behnke 2002, Miller 2005). For a general discussion of California steelhead systematics, including the significance of their designation as a Distinct Population Segments (DPS) (rather than an ESU) see the Northern California coastal winter steelhead DPS account.

Steelhead populations in California appear to follow a pattern of geographic isolation, with populations in proximity to each other generally being most closely related. However, the limited genetic analyses completed on southern steelhead do not follow this pattern. Girman and Garza (2006) found that populations of southern steelhead and South-Central California Coast steelhead do not partition themselves into independent lineages in neighbor-joining gene trees. The genetic relationships among putative populations suggest that southern steelhead are intermixed with steelhead from other DPSs in California. Southern steelhead watersheds have been the focus of decades of hatchery planting of rainbow trout from outside the region, although there appears to be very little genetic mixing of wild steelhead with these hatchery strains with the exception of a few populations south of the Santa Clara River basin (Girman and Garza 2006). However, steelhead of genetically native ancestry occupy some basins south of the Santa Clara River such as Malibu, San Gabriel, and San Mateo Creeks. Many collections of rainbow trout in the Girman and Garza (2006) study were from above dams and these fish were observed to be most genetically similar to anadromous fish in the same watersheds; this indicates recent ancestry of freshwater-resident trout from anadromous southern steelhead. The close genetic relationship between anadromous and resident rainbow trout in streams appears to be a widespread phenomenon (Docker and Heath 2003).

Boughton et al. (2007) used distributional information to identify 46 southern steelhead populations in five biogeographic regions. It is unclear if each population or each region containing multiple populations is capable of supporting viable populations as they once did. The Santa Monica Mountains and Santa Catalina Gulf Coast regions may have historically supported only ephemeral populations subject to recolonization from neighboring metapopulations in the northern Monte Arido Highlands and Conception Coast regions. The Mojave Rim Region, which is positioned between the Santa Monica Mountains Region and Santa Catalina Gulf Coast Region, is hypothesized to have had unreliable flows to the ocean and likely contained mostly freshwater resident trout (Boughton et al. 2006).

Life History: The ecology of southern steelhead has not been well studied but is presumed to be similar to that of the better documented steelhead populations further north (see NC coast winter steelhead account). Differences mainly relate to the variable environment in which southern steelhead evolved. Southern steelhead are dependent on winter rains to provide upstream passage through seasonally opened estuaries and flowing mainstem rivers. The reliance on rainstorms for permitting passage through the lower portions of southern California watersheds suggests a restricted and rapid spawning period for steelhead. This spawning period typically occurs between January and May, with a peak in February through mid-April (SYRTAC 2000). Recent summer observations of adult steelhead holding in the lower Ventura River following a temporary sandbar breach due to large swells and high tides suggest movement into fresh water is extremely opportunistic (Matt Stoecker, pers. comm. 2007). Rivers within the range of southern steelhead are presumably warmer than streams further north and these warmer temperatures likely decrease incubation time for alevins. For example, in the Ventura River, with 15.6ºC water temperatures, embryos can hatch and alevins emerge from the gravel in as little as three weeks (Barnhart 1986). Adult steelhead are iteroparous but it is not known if repeat spawning is common among southern steelhead. Larger steelhead are commonly observed isolated in mainstem rivers and estuaries during late spring and early summer, suggesting the late spawning period of southern steelhead may lead to late out-migration of spawned adults, with many adult fish not able to exit fresh water subsequent to bars forming over estuary entrances. If spawned-out adults are unable to return to the ocean, they may attempt to return to cold water habitats upstream to over-summer and perhaps spawn again.

Three life history patterns have been described for South-Central Coast steelhead which are also likely important for southern steelhead: fluvial anadromous, freshwater resident, and lagoon-anadromous (Boughton et al. 2007). Juvenile steelhead usually remain in freshwater for 1 to 3 years before emigrating (Shapovalov and Taft 1954). Southern steelhead, however, probably spend less time in fresh water because of the often inhospitable conditions (low flows, warm temperatures) in the lower reaches of southern California streams. Thus, southern steelhead may migrate to the ocean or have greater dependence on coastal lagoons during their first year compared to other stream-oriented northern steelhead populations. Southern steelhead outmigration is dictated by the breaching of estuary sandbars, typically between January and June, with a peak from late March through mid-May (SYRTAC 2000). Ocean swells and high tides can lead to temporary sandbar breaching during the summer and fall, draining lagoons and allowing juvenile trout to emigrate from the streams to the ocean. While barriers may limit the upstream immigration of anadromous steelhead, outmigrating juveniles originating from upstream of barriers are often found downstream of these barriers in the Santa Ynez River (A. Clemento, University of California, Santa Cruz, pers. comm. 2007). Juvenile and adult life history pattern plasticity ostensibly occurs in some portion of each southern steelhead population.

Smolts in the Santa Clara River outmigrated between mid-March and early May and fish 15-20 cm FL were typically 1 year old (Stoecker and Kelley 2005). In southern steelhead streams, estuaries at the mouths of watersheds typically turn into lagoons during the summer. These lagoons can be highly productive environments where juvenile steelhead grow quickly, leading to fish entering the marine environment during their first winter. Early smoltification may occur because rapid growth in these productive environments allows fish to reach a smolt size at a younger age (Bond 2006). In contrast, freshwater environments during the summer may have limited food resources, resulting in slow growth for southern steelhead (Boughton et al. 2007).

Because of frequent 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) evidently 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 (E. Gerstung, memorandum to R. Rawstron, CDFG, November 22, 1989). When droughts last over multiple years and anadromous steelhead are unable to spawn, the freshwater-resident populations are essential for the long-term viability of populations within some watersheds. Likewise, when catastrophic events (i.e., fires, landslides) extirpate steelhead from a watershed, the anadromous fish are presumably critical for the recolonization of the streams. It is likely that during wet years, a high percentage of the southern steelhead returning to spawn have spent only one year in the ocean. This "bet-hedging" strategy of attempting to spawn every year is adaptive to the unpredictable environmental conditions of southern rivers (J. J. Smith, CSU San Jose, pers. comm.).

Habitat Requirements: The basic environmental requirements for southern steelhead are similar to those of other California steelhead (see Northern California coast winter steelhead account). Southern steelhead require cool, clear, well-oxygenated water with ample food, but they have adapted to living under highly variable environmental conditions. Thus their physiological tolerances may be broader than other steelhead. The incipient lethal level of dissolved oxygen for adult and juvenile rainbow trout is approximately 3 mgL^{-1} (Matthews and Berg 1997). Egg mortality begins at 13.3ºC, and juveniles have trouble obtaining sufficient dissolved oxygen at temperatures greater than 21.1ºC (McEwan and Jackson 1996). Southern steelhead prefer higher elevation headwaters as spawning and rearing areas, although a majority of these areas have been blocked by human-made migration barriers. Lowland reaches contain a more restricted distribution of potential perennial habitats and the importance of lagoons for rearing habitat presumably has been amplified due to reduction of access to upstream habitats. Channel connectivity is critical for steelhead to access spawning areas and 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 in the Ventura, Santa Clara, and Santa Ynez Rivers (M. Capelli, in USFWS 1991). Adult steelhead require a minimum depth of around 17-20 cm to move upstream and a long reach of shallow water may be therefore be a barrier until higher flows arrive (McEwan and Jackson 1996).

Preferred temperatures of juvenile steelhead are reported as 10-17ºC, but southern steelhead seem to persist in environments outside this range. Carpanzano (1996) found steelhead trout in the Ventura River persisting where temperatures peaked daily at 28ºC and 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 areas of pools that had lower temperatures despite their associated low oxygen levels. Spina (2007), in contrast, found that thermal refuges were often not available to juvenile southern steelhead and that 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.

Within the riverscape, reaches with subtle patterns of temperature heterogeneity have an important influence on the growth of juvenile steelhead (Boughton et al. 2007; Spina 2007) and the patchy distribution of fish within reaches may be indicative of the influence of local temperature conditions. Geomorphology has an essential role in development of temperature heterogeneity because it influences pool depth, shading, and cooling of water through subsurface flows. To minimize thermal stress, southern steelhead often seek out areas with cool seeps, although thermal stratification of pools may be important if seeps are not present (Matthews and

Berg 1997). In Topanga Creek, where peak daytime temperatures regularly are above 21ºC, trout were more often found in habitats associated with cooler ground water, although these habitats made up only 16% of the available habitat (Tobias 2006). In streams without such refuges, steelhead persist by adopting different bioenergetic strategies (Spina 2007). Tobias (2006) found groundwater discharge areas typically had greater surface area, greater depth, and more shelter than other nearby areas, although Spina (2007) indicated that steelhead preferred such areas even without cool groundwater discharges. Trout densities were negatively correlated with aquatic macrophyte densities, likely due to low dissolved oxygen concentrations in these areas and the density and richness of non-salmonid fish species (Douglas 1995).

Different size classes of juvenile steelhead use different parts of the habitat available. In one stream, Spina (2003) found YOY 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.

Distribution: The southern steelhead DPS includes all naturally spawned anadromous rainbow trout populations below natural and human-made impassable barriers in streams from the Santa Maria River, San Luis Obispo, California (inclusive) to the U.S.-Mexico Border. Populations from over half of the 46 watersheds historically supporting steelhead runs have been extirpated (Boughton et al. 2005). All of the four largest watersheds (Santa Maria, Santa Ynez, Ventura, and Santa Clara Rivers) in the northern portion of the DPS are estimated to have experienced declines in run sizes of 90% or more. More recently, adult steelhead have been documented in San Juan Creek, San Luis Rey, and San Mateo Creek in Orange and San Diego counties (Hovey 2004). These southernmost populations are separated from the northern populations by 130 km (80 mi) (NMFS 2007). Broughton et al (2007) divide the range of the southern steelhead into five biogeographic regions (next paragraph).

Resident rainbow trout occupy numerous watersheds in the southern steelhead DPS region. These fish may be offspring of either anadromous steelhead or freshwater-resident trout, although many basins have barriers restricting anadromous adults from reaching optimal spawning habitat in their headwaters. The fires and droughts so common in southern steelhead range suggest that intermittent connectivity between the extant populations within each biogeographic region is critical for viability. In the most southern biogeographic region, the *Santa Catalina Gulf Coast*, resident trout are reported to recently have occurred in a majority of streams above barriers including San Mateo Creek, San Onofre Creek, Santa Margarita River, San Luis Rey River, San Diego River, and Sweetwater River (Boughton et al. 2007). A similar pattern was reported by Good et al. (2005) in the adjacent *Mojave Rim* biogeographic region's watersheds with resident trout being observed recently upstream of barriers in the Los Angeles, San Gabriel, and Santa Ana Rivers. Steelhead have reappeared in the past decade in the *Santa Monica Mountain* region, likely due to colonization events (Good et al. 2005). Malibu Creek also seems to have a small steelhead population, while Big Sycamore Creek's population seems to have been extirpated. Boughton et al. (2005) found steelhead trout in numerous watersheds in the *Conception Coast* biogeographic region, which are still 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 2002).

Abundance: Southern steelhead have been either significantly depleted in or extirpated from all rivers and streams in which they historically occurred. There are still important populations in the Santa Ynez, Ventura, Santa Maria, and Santa Clara Rivers. Remnant or ephemeral runs seem to occur in multiple DPS biogeographic regions including Gaviota, Arroyo Honda, Goleta Slough Complex, Mission, Malibu, San Gabriel, and San Mateo Creeks. In all these waters, estimates of historical run size estimates were highly subjective and based on very sparse data (Good et al. 2005). In the Santa Ynez River, which probably supported the largest historical run of southern steelhead, runs may have been as large as 20,000 to 30,000 spawners (Busby et al. 1996). However, this may be an overestimate based on evidence from 1944 (see Good et al. 2005). The minimum number of steelhead in the Santa Ynez River was 13,000-14,500 fish following a favorable wet period (Good et al. 2005). While the 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.

Historic run estimates on the Ventura River were 4,000-5,000 steelhead, but the estimates followed a decade with numerous plantings of fish into the basin (Good et al. 2005). Steelhead runs in the Matilija basin (part of the Ventura watershed) were 2,000-2,500 steelhead, but were also based on surveys following a period of numerous plantings (Good et al. 2005). In the Santa Clara River, historic runs have been estimated at 7,000-9,000 fish, and were based upon extrapolations of Clanton and Jarvis's (1946 and Moore 1980 cited in Good et al. 2005) estimates in Matilija Creek. However, the Santa Clara River is one of the largest watersheds in southern California (ca. 1600 square miles), so it was presumably once capable of supporting large numbers of steelhead (12.5 time that of the Ventura River, based on watershed size). Good et al. (2005) noted that anecdotal accounts indicate a precipitous decline in run sizes during the 1940s and 1950s, possibly due to drought and dam construction. In May 1991, 14-25 adult steelhead were observed in the upper estuary of 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, CDFG, memorandum to B. Bolster, CDFG, October 13, 1993). These observations are similar to more recent sightings that have occurred in the Ventura River and San Antonio Creek (Good et al. 2005). Fish from upstream of Bradbury Dam have been found downstream and rainbow trout in this basin appear to persist mainly as resident fish (A. Clemento, pers. comm.). Good et al. (2005) estimated a run of less than 100 steelhead annually, indicating this population may no longer be viable.

In the Santa Maria River, historic 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 steelhead trout to be have 52% 0+ fish, 24% 1+ fish, 17% 2+fish, and 7% 3+fish. A fourth remaining population exists in the Santa Clara River drainage. There are 129 natural and human-made fish migration barriers in the Santa Clara River watershed (Good et al. 2005). The Vern Freeman (VF) Diversion Dam, which has had a dysfunctional fish ladder since 1997, blocks access to 99% of the watershed (Good et al. 2005). When functioning, the fish ladder passed one fish in 1994 and 1995, two in 1996, and none in

1997 (Good et al. 2005). The VF Diversion Dam is downstream of the major southern steelhead spawning tributaries such as Piru and Sespe Creeks. Sespe Creek provides a large amount of high quality habitat, but also contains nonnative predatory fish. Though smaller than the above drainages, Santa Paula Creek provides some of the highest quality habitat in the watershed (Stoecker and Kelley 2005).

Overall, southern steelhead numbers have declined dramatically from estimated annual runs totaling a minimum of 30,000 adults to less than 500 returning adult fish combined in the past 50-75 years. Girman and Garza (2006) using genetic techniques, determined that populations in the Santa Ynez, Ventura, and Santa Clara Rivers had all gone through recent declines in effective population size. There have been no comprehensive surveys conducted in recent years to provide a reliable estimate of total population size for southern steelhead but numbers in most years are likely less than 500 spawners.

Factors affecting status: NMFS (2007) identified eight primary threats to southern steelhead viability which are associated with each of the four major river systems that still support small populations: these are the Santa Maria, Santa Ynez, Ventura, and Santa Clara Rivers. These four populations most likely serve as source populations for populations further south, which do not have steelhead currently or only small numbers of fish. Southern steelhead watersheds with only resident freshwater populations of rainbow trout likely continue to produce smolts and with adequate flows, mainstem habitat restoration, and barrier removal should provide opportunities for restablishment of natural anadromous populations. The primary factors impacting southern steelhead include: (1) urbanization, (2) dams and other barriers, (3) stream habitat loss, (4) estuarine habitat loss, (5) species interactions, (6) hatcheries, (7) drought and climate change, and (8) wildfire.

Urbanization: Most watersheds containing southern steelhead south of Santa Barbara County are heavily urbanized. Not surprisingly, the four largest watersheds containing them 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. The impacts of urbanization in the major watersheds (i.e., San Gabriel, Santa Ana, San Juan, Santa Margarita, and Sweetwater Rivers) of the DPS reduce perennial flows and decrease connectivity among habitats; this is turn reduces the persistence of steelhead in the streams. Urban and rural waste discharges are also widespread, which degrades water quality and create habitat conditions that favor alien aquatic organisms.

Dams and other barriers: 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 2002, NMFS 2007). Of the larger dams, Matilija Dam blocking the Ventura River and Rindge Dam on Malibu Creek are being considered for removal. 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. Diversion dams and poorly functioning fish ladders on the Santa Clara River have denied steelhead access to spawning habitats and reduced available rearing habitat for steelhead offspring. Twitchell Dam eliminated half of the Santa Maria River's historically accessible habitat and water diversions continue to reduce connectivity among critical lower watershed tributaries (i.e., Sisquoc River) and the estuary.

Stream habitat loss: Southern California steelhead streams have suffered major loss of physical habitat of all types from diverse sources, including channelization, road crossings, stream bank stabilization, sedimentation, and many other abuses. In addition, diversion of water and increases in non-permeable surfaces (e.g., roads, parking lots) have made the hydrograph more extreme in many streams, with flashier winter flows and lower summer flows, greatly reducing habitat quality and amount. Floodplain development has also altered natural fluvial processes and reduced riparian habitats, which facilitate adult migration and juvenile rearing. Associated flood control structures (e.g., levees) and activities have further disrupted the natural fluvial processes. Increases in residential structures (and associated roads) on steep sided erosive slopes has accelerated erosion and sedimentation of river and stream channels.

Loss of estuarine habitat: Southern steelhead are likely similar to South-Central Coast steelhead in their use of estuaries (see South-Central Coast steelhead account for more details). Estuaries are essential for juvenile rearing, adult migration, and occasionally adult oversummering (Bond 2006). Many southern California estuaries/lagoons have disappeared due to human activities , while others are functionally degraded (Lafferty 2005). Many are much shallower and warmer than they were originally, due to altered stream and sediment flows and this influences their temperature and salinity. Overall, Southern California has lost approximately 90% of its historical estuarine habitat through dredging and filling. 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 and possibly because Highway 101 provides some protection for them. The degradation of remaining estuarine habitat as a result of both point and non-point sources of pollution and artificial breaching of sand-bars has reduced the suitability of these habitats for steelhead rearing and as transition zones between marine and freshwater environments.

Species interactions: The presence of alien fishes, both predators (e.g., smallmouth bass) and competitors (e.g., arroyo chub) is pervasive in Southern California streams. Although habitat may exist for southern steelhead in some watersheds from which they are currently missing, the presence of non-native fishes can make reestablishment of steelhead in these basins difficult. Stoecker (2005) found steelhead and arroyo chub densities had a strong negative relationship, possibly due to competition and/or different optimal water temperatures.

Hatcheries: Stocking of non-native strains of rainbow trout to support recreational fisheries has been a common practice in current and potential steelhead habitat in both the northern (Stoecker 2002; Stoecker 2005) and southern (USFWS 1998) portions of the Southern steelhead range. While there appears to be very little genetic mixing with hatchery strains, genetic analyses of some juvenile trout from watersheds collected south of the Santa Clara River showed genetic signals of hatchery ancestry (Girman and Garza 2006). This includes fish from Topanga Creek, which seem to be intermediate between wild and hatchery fish and from the Sweetwater River and San Juan Creek, which appear to be primarily of Fillmore Hatchery origin. The negative genetic consequences associated with hatchery plantings suggests reliance on broodstock with origins other than native wild stocks; this is likely to lead to modification of genetic diversity and rapid fitness declines in planted stocks (Kostow 2004; Araki et al. 2007). Stocking of hatchery steelhead is also a threat to southern steelhead because of competition with wild fish, introduction of disease, and a tendency for managers to rely on artificial culture as a substitute for the maintenance of self-sustaining steelhead populations in their native ecosystems (NMFS 2007).

Drought: Droughts have a profound influence on southern steelhead by eliminating passage during the spawning and smolting season and by reducing summer freshwater habitat. Tree-ring records suggest long periods of historical drought in Southern California which would have affected all populations of southern steelhead. Steelhead must have either survived in drought-resistant refuges or been extirpated regionally during these periods (Boughton et al. 2007). The development of southern California watersheds by humans has essentially made droughts more frequent and more severe from a fish perspective, decreasing likelihood of survival through dry years. In addition, human-caused climate change is likely increasing the natural frequency and severity of droughts, exacerbating the problem.

Wildfire: Periodic wildfires are an integral ecological feature of Southern California. 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 (Keller et al. 1997), covering habitats and fish alike. Following a wildfire, if winter rains do not mobilize sediment but do increase runoff, then favorable characteristics such as increased scour and nutrients may benefit steelhead trout. As with drought, the severity and presumably frequency of wildfires is increasing in southern California, making it more difficult for steelhead to persist in some watersheds.

Conservation: The final critical habitat designation for the endangered southern steelhead was made in 2005 (NMFS 2005) and 1133 km (708 miles) of stream habitat within 32 watersheds were designated as critical habitat. Conservation of southern steelhead will require the (1) immediate protection and expansion of habitat for steelhead within each of the five biogeographic regions and (2) reestablishment of large runs in streams that historically were highly productive for steelhead (i.e., Santa Maria, Santa Ynez, Ventura, and Santa Clara Rivers). Both of these conservation goals should include directed research into life history diversity and adaptations of southern steelhead, as well as increased monitoring of existing populations. Public education and increased intergovernmental cooperation among local, county, state, and federal agencies are essential to long-term success of restoration and management actions.

Restoration efforts focused at the watershed level, particularly dealing with ensuring adequate flows and passage to historical spawning and rearing areas that are most likely to result in increases in the number of steelhead. Numerous local restoration fixes are needed to provide for re-establishment and expansion of southern steelhead populations, including providing connectivity among populations in different streams. Many extant southern steelhead populations are on public lands, and effective management of these waters by state and federal managers is needed to benefit these populations.

Expansion of southern steelhead populations in each biogeographic region is important to guarantee sufficient redundancy to reduce the extinction risk of steelhead within these groups due to wildfire and other natural factors. NMFS modeled the necessary number of steelhead populations in each biogeographic region based on the geographic extent of a 1000-year fire, similar to what was observed in the fire of 2003 (Boughton et al. 2007). They determined that at least twenty populations were needed spread among the regions. The ability to protect southern steelhead from catastrophic fire is limited and a stochastic event such as this could lead to extirpation within a large portion of the DPS.

Changes in water management are critical to restoring habitats and geomorphic processes important to southern steelhead. The feasibility of reintroduction and suggested plans of action are discussed in detail by Higgins (1991) for San Mateo Creek and the Santa Margarita River. Water removal from streams now containing critically low numbers of steelhead should be restricted or enhanced in order to leave minimum flows for fish in streams and lagoons. The environmental impact of future development projects should be carefully evaluated and appropriate alternate measures reviewed by state and federal regulatory agencies (e.g., CDFG, RWQCB, NMFS) prior to accepting mitigation approaches. Restoration techniques that can increase habitat fairly rapidly for southern steelhead may include groundwater recharge projects, removal of barriers in watersheds with high habitat quality, and enhancement of instream and riparian habitats. Return water from sewage 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 juvenile steelhead during the fall and early winter in lower reaches of Southern California streams.

Culverts, road crossings, and bridges are a significant impediment for steelhead migration in many southern steelhead streams and their removal or modification provides an opportunity for increasing connectivity within watersheds for different steelhead life history types and among the different populations within biogeographic groups. In many cases, barriers have been identified and assessed so planning and implementation of these projects can occur quickly. Further studies are needed on how southern California estuaries are used by steelhead as rearing habitat and measures for restoring estuarine habitat need to be developed and implemented.

Dams and fish passage facilities provide numerous opportunities for restoring southern steelhead into portions of watersheds with optimal spawning and rearing habitats. In many cases, resident trout persist upstream of these barriers. Considerable planning has gone into removal of Matilija Dam on Matilija Creek, a tributary of the Ventura River, and Rindge Dam on Malibu Creek, as well as construction of fish passage facilities on the Ventura River (Robles Diversion Dam). Implementation of these projects should be more expeditious in order to benefit southern steelhead as soon as possible. Evaluation of fish passage barriers and associated water operation facilities in the Cuyama, Santa Ynez, Santa Margarita, 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 Cayuma River can be operated more effectively to permit re-establishment of flows during periods critical for steelhead survival, especially during migration and periods when fish are rearing in estuaries and lower river reaches. Use of trap and haul techniques to move steelhead into upstream areas may also be needed in dry years.

Opportunities for recovery in the Southern California Coast steelhead DPS are limited due to increasing effects of climate change anticipated over the next 100 years. NMFS (2007) identified strategic recovery actions for southern steelhead. They were:

- Identify and commit to a core set of populations (anadromous and resident) on which to focus recovery efforts.
- Secure extant parts of the inland populations in the Monte Arido Highlands and Mojave Rim biogeographic regions.
- Identify and maintain sustainable refugia against severe droughts and heat waves.
- Collect population data annually.
- Secure and improve estuarine/lagoon habitat.

- Decide on a strategic balance and timeline for investment in better information vs. investment in recovery activities.
- Establish programs for ecosystem-based management of sediment regimes and hydrographic regimes.

Trends:

Short term: An absence of monitoring data makes understanding trends in southern steelhead difficult. Development of a baseline monitoring plan for steelhead and steelhead habitat in Southern California watersheds is an essential task. Despite the paucity of data, it appears that southern steelhead populations have declined in the last 25 years and are continuing to decline, with many headed towards extinction in the near future.

Long term: The long-term historical trend for southern steelhead has been one of continuous decline, with present populations probably 10-20% of historical populations on average. The decline is likely to continue. While there is considerable interest in restoring southern steelhead, increasing human populations and water consumption combined with the effects of climatic change are making southern California's streams increasingly less habitable for steelhead. Extirpation of southern steelhead populations has already occurred in watersheds where barriers have eliminated connectivity between resident and anadromous populations (Boughton et al. 2005). Loss of longitudinal connectivity is an increasing threat as water demand increases, flows are reduced in stream reaches needed for passage, and wildfires and droughts eliminate upstream segments of populations. In addition, climate change, with increases in temperature and variability in rainfall, is likely to reduce habitat for southern steelhead to levels less than what is necessary to support viable populations in all streams. A conscientious effort is required to maintain or increase stream flows in key areas and otherwise improve habitats. Further efforts that may be necessary include a conservation hatchery program for populations in danger of extirpation.

Status: 2. Southern steelhead are in danger of extinction within the next 25-50 years, due to the growing human population of Southern California and climate change (Table 1). Southern steelhead were listed as an endangered species by NMFS in 1997 and endangered status reaffirmed on January 5, 2006. They are considered a Species of Special Concern by the California Department of Fish and Game. Urbanization, land disturbance, and water associated impacts will continue to threaten their persistence into the future and a number of populations have already been extirpated. Other populations are blocked from reaching much of their critical upstream spawning and rearing habitats. NMFS concludes there is moderate potential for recovery of southern steelhead (NMFS 2007). If resident rainbow trout populations are considered part of the southern steelhead complex (they are not at present), then the extinction threat of the genetic population is somewhat less. The steelhead life history strategy, however, is essential for connecting and maintaining the isolated resident trout populations, so considering the two forms as one just puts extinction a bit further into the future.

Metric	Score	Justification
Area occupied	3	Found in most of native range, if scattered.
Effective pop. Size	2	Limited availability of habitat annually likely leads to limited spawning. Each population appears to be small and independent.
Intervention dependence	2	Intensive efforts such as barrier modification, habitat restoration, and restoration of instream flows are essential to maintenance of populations.
Tolerance	2	Moderate physiological tolerance to existing conditions, although limits are being reached; semelparity probably the rule.
Genetic risk	2	Limited gene flow among populations; some hatchery hybridization. Populations small.
Climate change	1	Climate change likely to impact them throughout their range, exacerbating other factors.
Average	2.0	12/6
Certainty (1-4)	3	

Table 1. Metrics for determining the status of southern steelhead, where 1 is poor value and 5 is excellent.

RESIDENT COASTAL RAINBOW TROUT
Oncorhynchus mykiss irideus

Description: Resident coastal rainbow trout 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 typically silvery in color, white on the belly, with black spots on the tail, adipose fin, dorsal fin, and back; tail spots are placed in radiating lines. There is a pink to rosy lateral band on each side and the gill covers are usually also pink. Color is highly variable, however, so trout from small streams may be fairly dark on the back with a yellowish belly. 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. 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. See Moyle (2002) for a more detailed description.

Taxonomic relationships: Under this name are many different populations of rainbow trout that presumably had independent origins from steelhead, including some that may naturally interbreed with steelhead or produce young that go out to sea, as well as populations established through introductions. These populations include (1) those in upstream areas, usually above natural barriers, in coastal watersheds, (2) those in Central Valley streams, and (3) those established through introductions above barriers (e.g., in the Sierra Nevada) and into non-native watersheds. The boundary between steelhead and resident rainbow trout is fuzzy; for example, reservoirs often develop steelhead-like runs of fish that spawn in tributary streams. Such runs may or may not have been derived from steelhead trapped behind the dams. In addition, many resident trout populations, especially those resulting from introductions, may have originated from hatchery strains, of mixed stock, although traits of wild native fish would presumably be selected for under natural conditions. We follow Behnke (1992, 2002) in using *O. m. irideus* to refer to all non-redband trout, both resident and migratory. Resident coastal rainbow trout have multiple origins from steelhead, so represent a taxon of convenience. For further discussion, see Moyle (2002).

Life history: Coastal rainbow trout have a high diversity of life history strategies which is a principal reason for their success. The classic pattern for resident fish, however, 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, The trout mature in their second or third year of life, spawn 1-3 times, but rarely live more than five or six years. Spawning takes place in spring (February to June, depending on flows and temperatures). Each female digs a series of redds and buries the fertilized embryos. The embryos hatch in 3-4 weeks (at 10-15° C) and the fry emerge 2-3 weeks later. The 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, the fish may be territorial or partially so, but fish in pools tend to hang out in the water column in groups, albeit with some sorting by size. Diets of stream-dwelling trout are primarily aquatic and terrestrial insects that are drifting in the water column, although frogs and fish may also be consumed on occasion, and benthic feeding also occurs. 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 streams and secondarily in lakes and reservoirs. They typically thrive in the tailwaters of large dams. Rainbow trout are among the most physiological tolerant of salmonids, which is why they are often the only salmonid found in streams that are thermally marginal. They can live in waters that reach 26-27° C in summer for short periods of time, provided there is sufficient acclimation time and plenty of food available (see Box 1 on bioenergetics in SONCC coho salmon account). Thermal refuges (e.g. upwelling ground water) are also important in marginal situations. Optimal temperatures for growth (and preferred temperatures) under 'normal' circumstances are usually 15-18° C. At low temperatures, rainbows can survive relatively low dissolved oxygen concentrations although saturation is needed for most activities. They also can survive and grow in a wide range of water chemistry, including water with pH values between 6 and 9. As indicated under life history, 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, 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 for the most part resident fish are/were more 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 found wherever there was an evolutionary advantage to being resident, usually above barriers difficult or impossible for steelhead to pass. Today, thanks to thousands of official and unofficial introductions, resident trout with coastal rainbow origins, are found in virtually all streams where habitat is suitable. Their expanded range includes most of the lakes and streams in the once-fishless Sierra Nevada, north of the Upper Kern basin. For more details, see Moyle (2002).

Abundance: Wild, naturally spawning resident coastal rainbow trout are undoubtedly much more abundant than they were historically in California because of their introduction into most suitable waters, including reservoirs, and their high abundance in tailwaters below large dams. While local populations in urban and heavily agricultural areas may be diminished or even eliminated, total abundance statewide is high. "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)."

Factors affecting status: At one time or another virtually every factor discussed for other salmonids in this report have reduced local resident rainbow trout populations: over-exploitation, water diversions, dams, pollution, poor watershed management (through logging, agriculture, over-grazing, road building), mining, channelization of streams, introductions of alien species, and so forth. Because of their hardiness and value to recreational fisheries (increasingly, catch-and-release fisheries for wild fish), many local populations have persisted and have become the focus of restoration programs. Hybridization of locally-adapted strains with fish of hatchery origin 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.

Part of the success story of resident rainbow trout is their wide introduction outside their native range, all over California, North America, and the world. Most of these populations are at least partially, if not wholly, derived from California coastal rainbow stocks. Of course, 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 in the eastern Sierra Nevada). They are considered worldwide one of the hundred worst invaders in the World by the International Union for the Conservation of Nature (Lowe et al. 2000).

Conservation: Conservation efforts mostly center around improving existing populations to increase wild trout populations for recreational fisheries. In fact, increasing the number of stream miles devoted to thriving wild trout populations is now a major goal of the California Department of Fish and Game, mandated by state law. Maintaining such populations even at present levels, however, is going to be an increasing challenge as climate change results in warmer water, reduced summer flows, and increased frequency of large floods throughout California. In addition, there will be continuing conflicts with protecting endangered fishes and other aquatic species.

Trends:

Short term: In recent years, resident wild trout populations have probably at least held their own ,with decreases in some areas due to urbanization and other intense human use of watersheds and increases in other areas, thanks to conservation efforts by agencies, local watershed groups, and organizations such as California trout.

Long term: Since the 19th century, resident rainbow trout populations, presumably mainly of coastal rainbow trout, have increased in distribution and abundance thanks to introductions. Starting roughly in the 1950s, however, increasing emphasis was placed on supporting fisheries with domestic trout from hatcheries. While planting domestic trout for put-and-take fisheries is still an important activity of the California Department of Fish and Game, 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, by protecting riparian corridors, by improving flow regimes below dams and other actions. Climate change effects (above), however, may reduce these gains in the next 50 years without continuous action to protect trout streams and their cold-water flows.

Status: 5. Despite all the damage done to trout streams in the past 150 years, resident coastal rainbow trout continue to thrive in many areas. Populations are presumably expanding at the present time due to conservation efforts.

Metric	Score	Justification
Area occupied	5	Abundant and widely distributed around the world
Effective population size	5	Many fish in many populations
Intervention dependence	5	While stream improvements and other activities greatly improve the habitat of native and introduced populations, most populations can at least persist on their own with existing protective laws and regulations.
Tolerance	4	Physiological tolerance rarely an issue.
Genetic risk	5	Lots of gene flow among populations.
Climate change	4	Management can help make up for habitat losses due to climate change.
Average	4.6	28/6
Certainty (1-4)	4	Well documented

Table 1. Metrics for determining the status of resident rainbow trout , where 1 is poor value and 5 is excellent.

SOUTHERN OREGON-NORTHERN CALIFORNIA COASTAL CHINOOK SALMON
Oncorhynchus tshawytscha

Description: Chinook salmon from the various ESUs differ only slightly in basic morphology and meristics, so see the Upper Klamath-Trinity Rivers fall Chinook salmon account for a description of the species.

Taxonomic Relationships: The Southern Oregon- Northern California coastal Chinook salmon (SONCC) ESU can be distinguished from other California Chinook ESUs with molecular techniques(Banks et al. 2000, Waples et al. 2004). Within the ESU, genetic analyses with microsatellite loci and reanalysis of older allozyme datasets demonstrate that fish from the Klamath River and fish from Blue Creek (Lower Klamath River) form two genetic clusters within the Klamath Basin (Myers et al 1998). The Blue Creek sample clustered with collections from further north of the Klamath River. Banks et al. (2000) used microsatellite DNA to show that Blue Creek Chinook salmon were the most genetically divergent of the collections from the Klamath River and were most similar to southern Oregon and California coastal Chinook collections. Snyder (1931) noted SONCC Chinook salmon from the Smith River and Blue Creek were similar in morphological and reproductive maturity. Spring Chinook runs can also be found on the Smith River, but the relationship of these fish to Fall SONCC Chinook is unknown. It is possible they are strays from the more abundant Spring SONCC run in the Rogue River or perhaps Smith River fall Chinook that simply return early. Very little information exists about SONCC spring run in California.

Life History: SONCC Chinook salmon are principally late fall-run Chinook salmon that have adapted to coastal watersheds in the Klamath Mountains. They enter tributaries in the lower Klamath River from September through December, a broader period than is found in Upper Klamath Trinity River Chinook salmon; spawning activity typically occurs later, continuing into January (Leidy and Leidy 1984). Spawning has been observed between November and February in Mill Creek (Smith River). In Blue Creek, Gale et al. (1998) observed SONCC Chinook entering in September, with peak entry occurring in November following fall rains. Spawner migration continued into Blue Creek through December and multiple distinct pulses of spawning fish have been observed. Gale et al (1998) hypothesized that early entering Chinook may be less sexually mature than later-entering fish, which spawn lower in Blue Creek than the earlier arriving Chinook. Increased stream discharge is critical for SONCC immigration into coastal tributaries. Waldvogel (2006) observed that the time females spent at the redd decreased as the spawning season progressed from 10-21 days for early spawners to 5-10 days for late spawners.

Fry emerge in lower Klamath tributaries from February through mid-April (Leidy and Leidy 1984). SONCC Chinook salmon principally demonstrate an “ocean-type” juvenile life stage (See Sacramento River spring Chinook account for a discussion on stream-type vs. ocean-type Chinook). On Blue Creek in 1995-96, juvenile emigration started prior to placement of outmigrant traps in mid-March. Juvenile emigration peaked in late April and late May, respectively, before tapering off during mid-August (Gale et al. 1998). The mean fork length of Chinook captured increased throughout the trapping season and attained 103 mm FL during late August (Gale et al. 1998). The early migrants apparently spent little time rearing in their natal streams but moved out quickly to the estuary. The larger Chinook fingerlings rear for several months in their natal streams prior to seaward migration (Sullivan 1989). This strategy likely increases ocean survival and 28% of Chinook juveniles emigrating from Blue Creek in 1996 displayed this life history variation. McCain (1994) studied juvenile rearing in Hurdygurdy Creek in 1987 and 1988 and found that approximately 5% of the fish produced from redds in both years remained in the stream to rear after spring flows receded. However, the total number of fish observed and their length of residency time differed between the two years, possibly due to high flows in spring of 1988 that forced emigration of a higher proportion of the juveniles from the creek. Reimers (1971) studied the length of residence of juvenile chinook salmon in tributaries of the Sixes River, Oregon (northern section of SONCC) and reported that some fish moved directly downstream after emergence and into the ocean within a few weeks, while others reared in the streams for periods ranging from two months to over a year. Scale analysis of spawners revealed that most of the adults that survived to return reared in fresh water for two to six months. Reimers' study implies that although a large percentage of a cohort may move directly to the ocean after emergence, the fraction that rear in fresh water for an extended period (two to six months) may contribute most to the long term viability of the population.

Juvenile SONCC Chinook likely do not require extended estuarine residence and can immediately enter the ocean. In the Smith River estuary, Quiñones and Mulligan (2005) found that Chinook were most commonly observed rearing in the stream-estuary transition zone (<5‰ salinity), though some individuals did occupy lower estuarine waters. Klamath River Chinook salmon are found in the California Current off the California and Oregon coasts. Salmon seem to follow predictable ocean migration routes and Chinook recaptured in the Klamath River altered their ocean behavior to use habitats that exhibited temperatures of 8°-12°C (Hinke et al. 2005). Chinook salmon identified as originating from Southern Oregon stocks, which the SONCC ESU contains, were found north and south of Cape Blanco in June but made up the majority of the identified stock groups of Chinook encountered south of Cape Blanco in August (10%) (Brodeur et al. 2004). A majority of SONCC Chinook spawners in Blue Creek were age 3 fish, though age 4 and age 5 fish were observed (Gale et al. 1998). In Mill Creek (Smith River) 3 year old fish made up the majority (62%) of spawners in 1993-2002, though 4 year old fish (66%) dominated female spawners in 1981-1992 (Waldvogel 2006). Grilse (jacks), age two fish that return to spawn, constituted a smaller proportion of Chinook in Blue Creek than in other more interior Klamath tributaries. In 1995-96, approximately 7 percent of the annual Klamath River Chinook salmon observed were grilse.

Habitat Requirements: Spawning is primarily in habitats with large cobble and sufficient flows causing subsurface infiltration to provide oxygen for developing embryos. For SONCC Chinook, a majority of spawning habitat was found in the middle reaches of coastal tributaries. In Blue Creek, large numbers of spawners were observed holding in deep pools and swift run and pocket-water habitats (Gale et al 1998). During 1995-96, a majority of spawning activity in Blue Creek was observed in run habitats (Gale et al 1998). 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 (Healey 1991). Preferred spawning habitat seems to be at depths between 25 to 100 cm and water velocities of 30-80 cm/sec. Regardless of depth, the key to successful spawning is having adequate flow of water and redds are constructed in areas of 2-10m^2, where the loosened gravels permit steady access of oxygen-containing water. For maximum embryo survival, water temperatures must be 5°-13°C and oxygen levels must be close to saturation. For more details on temperature requirements of Chinook salmon, see the Central California Coast Chinook account.

With optimal conditions, embryos hatch in 40-60 days and remain in the gravel as alevins for another 4-6 weeks, usually until the yolk sac is fully absorbed. Where summer temperatures remain fairly cool (<20°C), juveniles will remain in stream habitats through the summer (Gale et al. 1998). No relationship was observed between emigration peaks and stream discharge in Blue Creek in 1995-96 (Gale et al 1998). Riparian vegetation that hangs over shallow water habitats is an important feature of juvenile freshwater and estuarine rearing habitats; the trees and bushes provide food (insects), cover, and habitat complexity for foraging and territoriality.

Distribution: The ESU includes all naturally spawned populations of Chinook salmon from Cape Blanco, OR (south of the Elk River) to the Klamath River. Coastal tributaries of the Klamath River up to the Trinity River confluence are included in this ESU. In California, SONCC Chinook salmon are distributed primarily in relatively small watersheds that are heavily influenced by maritime climate and were historically found in the numerous small coastal tributaries of the Lower Klamath River (USFWS 1979) . Surveys reported in USFWS (1979) completed during 1977-78, found Chinook salmon in Hunter, Terwer, McGarvey, Tarup, Omagar, Blue, Surpur, Tectah, Johnson, Mettah, and Pine Creeks. More recent surveys found Chinook in 8 of 10 (not present in Omagar and Surpur, Pine unsurveyed) of the earlier examined watersheds and also Hoppaw, Saugep, Waukell, Bear, Pecwan, and Roaches Creeks (Gale and Randolph 2000). Gall et al. (1989) indicated that the ocean migration patterns of populations from SONCC basins (Rogue and Smith Rivers) are different from those of populations in the more southern coastal rivers of the California Coastal Chinook salmon ESU. Apparent SONCC spring run Chinook have been observed in both the Middle Fork and South Fork of the Smith River (Reedy 2005).

Abundance: The vast majority of Chinook in the SONCC Chinook ESU originate from the Rogue River in Oregon with the Lower Klamath tributaries and Smith River contributing relatively small numbers of fish. USFWS (1979) cited a 1960 report that estimated that 4,000 Chinook salmon spawned in tributaries downstream of the Trinity River confluence. Spawning ground surveys in 1978-79 revealed numerous Chinook salmon and USFWS (1979) estimated annual runs to be approximately 500 fish. Spawner surveys in Blue Creek during 1995 and 1996 found 236 and 807 Chinook, respectively (Gale et al. 1998). Historic numbers were likely in the range of 2,000 or 3,000 returning spawners in most years (Moyle 2002).

In the Smith River, annual estimates of spawner abundances were estimated by CDFG to be around 15,000 in the 1960s (Moyle 2002), although this crude estimate is likely high. The Smith River remains undammed and there is no evidence of a long-term change in habitat, so runs sizes have presumably not changed much on average. Waldvogel (2006) surveyed chinook spawners in a small tributary (Mill Creek) over a 22 year period (1980-2002) and found numbers were highly variable (average return, about 160 fish) but with no trends. Spring Chinook in the Smith River have probably always had low numbers. Surveys in recent years for spring Chinook found just 5-21 individuals (34 to 53 miles surveyed; Reedy 2005).

Factors affecting status: The factors affecting the abundance of SONCC Chinook salmon overall are similar to those affecting Central Coast Chinook salmon, but in California the main factors seem to be habitat alteration, hatcheries, and fisheries, although the effects are poorly documented..

Habitat alteration: Although portions of the Blue Creek and other lower Klamath watersheds are not managed as industrial timberlands and although a majority of the Smith River is protected as a Wild and Scenic River, upslope land practices and road building likely have impacted the SONCC Chinook populations (USFWS 1979). As elsewhere in the region, landslides from road construction and clear-cutting on young coastal geologic formations cause chronic siltation and reduce the ability of spawning areas to support fish. In the Smith River estuary, land reclamation through construction of dikes and levees has reduced the amount of juvenile rearing habitat by up to 40% (R. Quiñones, pers. comm. 2007).

Hatcheries: Although no hatcheries are operated on Lower Klamath SONCC Chinook streams, there are potential interactions among hatchery and natural SONCC Chinook in the Lower Klamath as juvenile and adults, because of the abundance of fish from upstream hatcheries on both Klamath and Trinity Rivers. USFWS (2001) noted hatchery fish emigrated through the middle Klamath later than natural Chinook juveniles and these fish may potentially compete with SONCC Chinook, which also seem to exit natal watersheds later in the midsummer. The numerous returning hatchery spawners undoubtedly "stray" into Lower Klamath spawning areas, and may obscure the genetic distinctiveness of the SONCC Chinook in the Lower Klamath River. On the Smith River, the Rowdy Creek Hatchery spawns about 100 Chinook salmon a year and the juveniles are released in the spring. There seems little reason for this hatchery, given the pristine nature of the river and the self-sustaining nature of the run.

Fisheries: Commercial, sport, and subsistence fisheries have presumably significantly reduced SONCC Chinook abundance within the Klamath Management Zone (KMZ) in the past. Recent fishing reductions to protect the weaker Upper Klamath-Trinity River fall Chinook salmon presumably reduced harvest of Chinook salmon from the Lower Klamath and Smith Rivers as well.

Conservation: The SONCC Chinook ESU was separated from a larger Southern Oregon and California Coastal Chinook ESU in 1999 based on genetic and ecological differences from the more southerly California Coastal Chinook populations (NOAA 1999). While overall ESU abundance remains large, the California portion has presumably been reduced from historic numbers, although recent abundance seems stable in the Lower Klamath and Smith Rivers. It would seem desirable to close down the Rowdy Creek Hatchery to prevent possible negative influences of hatchery fish and to increase the value of the Smith River as a hatchery-free reference stream. At the very least, an intensive evaluation program of the hatchery should be initiated (e.g., marking all fish).

The persistence of SONCC Chinook salmon in their most important watersheds in California (e.g., Smith River, Blue Creek) suggests that protecting spawning and rearing habitats in these streams is important for conserving the ESU in the state. If SONCC Chinook remain abundant in these watersheds, then recolonization of other recovering watersheds, which were subject to historic degradation from logging and road building, is more likely.

The low abundance of spring-run Chinook in the Rogue and Smith Rivers may represent a threat to the total life history diversity of fish in the ESU. Special efforts should be made to document this run, determine its genetic history, and to find ways to increase its abundance.

Trends:

Short term: SONCC Chinook in California are currently limited to a few small Lower Klamath tributaries, Blue Creek, and the Smith River although the abundance of these populations seems stable. Fall run Chinook salmon appear to be persisting in Blue Creek and the Smith River, while smaller Lower Klamath tributaries have reduced populations as the result of land use practices, especially logging. Spring run Chinook have virtually disappeared from the SONCC ESU in California.

*Long term***:** The proximity of California SONCC Chinook population to the ocean and influence of cooler temperature in coastal California may provide the necessary conditions to minimize population impacts by climate change. Neither Blue Creek nor the Smith River suffer from water withdrawal from headwater areas and so maintain a natural flow regime. Continued efforts to protect these key SONCC Chinook watersheds and to minimize riparian disturbance and sedimentation suggest that these populations are likely to remain stable or increase.

Status: 4. No extinction risk (Table 1), although distribution is limited in California to a few fairly wild watersheds which are primarily in public and tribal lands. This ESU was determined to not warrant listing under the Endangered Species Act on September 16, 1999 by NMFS, although it is considered a Sensitive Species by the US Forest Service, Pacific Southwest Region.

Metric	Score	Justification
Area occupied	4	Blue Creek and Smith River are stable populations with additional populations in Oregon.
Effective population size	4	A couple hundred fish exist in the Lower Klamath tributaries and at least 1000 in the Smith River.
Dependence on intervention	5	California populations are largely self-sustaining.
Tolerance	3	Multiple juvenile life histories and spawner age diversity demonstrate physiological tolerances.
Genetic risk	4	Limited hatchery operations in California portion but some concern for hybridization with hatchery ‘strays’ from other ESUs
Climate change	4	Fall run is least vulnerable to climate change in North coastal environment of California since they spawn later and scouring of redds is less likely to influence juveniles. Their streams are close to the coast and likely to stay cool in most scenarios.
Average	4	24/6
Certainty (1-4)	3	Least studied of Klamath River Chinook runs

Table 1. Metrics for determining the status of SONCC fall run Chinook salmon, where 1 is a poor value and 5 is excellent.

UPPER KLAMATH-TRINITY RIVERS FALL CHINOOK SALMON
Oncorhynchus tshawytscha

Description: 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. Klamath River Chinook possess significant differences from Sacramento River Chinook in the number of their 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 1923b). Dorsal fin ray, anal fin ray and branchiostegal counts are significantly different from Columbia River Chinook (Snyder 1931, Schreck et al. 1986). 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.

Spawning Chinook adults are the largest Pacific salmon, typically 75-80 cm SL, but lengths may exceed 140 cm. Klamath River Chinook spawning adults are considered to be smaller, more rounded, and heavier in proportion to their length compared to Sacramento River fish (Snyder 1931). In 2004, Trinity River fall run Chinook averaged 69 cm FL with a maximum grilse size of 56 cm FL (CDFG 2006a). Adults are olive brown to dark maroon without streaking or blotches on the side. Males are often darker than females and developed a hooked jaw and slightly humped backs during spawning. Juvenile Chinook have 6-12 parr marks often extending below the lateral line, and they are typically equal to or wider than the spaces between. Occasionally, parr will have spots on their adipose fin, but a more distinguishing adipose fin character is a pigmented upper edge and clear center and base.

Taxonomic Relationships: The Upper Klamath-Trinity Rivers Chinook salmon 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 Upper Klamath-Trinity Rivers Chinook salmon (UKTR Chinook) ESU is genetically distinguishable from other California Chinook ESUs (Banks et al. 2000, Waples et al. 2004). Although fall-run and spring-run Chinook salmon are both part of this ESU, we treat the two runs as separate taxa due to the distinctive adaptive components characterized by these two groups.

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 appeared more closely related than fall-run Chinook from adjacent basins. This pattern is distinct from Chinook of different run timings in the Sacramento and Columbia Rivers, which show deeper temporal divergences than geographic divergences (Waples et al. 2004). Thus, fall run Chinook populations from both the Upper Klamath and Trinity Rivers appear more similar genetically to spring populations in the same subbasin than to fall Chinook salmon in Lower Klamath River tributaries.

Life History: UKTR fall Chinook salmon express 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 Central Valley spring Chinook account for a more detailed discussion of ocean-type vs. stream-type life histories). UKTR fall Chinook salmon enter the Klamath Estuary from early July through September. They often hold in the estuary for a few weeks and initiate upstream migration as early as mid July and as late as late October. Migration and spawning both occur under decreasing temperature regimes. Fall UKTR Chinook seem to hold extensively in and travel slowly through the Lower Klamath River (Strange 2005). Between 1925 and the early 1960s, the Klamathon Racks provided a counting facility as an egg collection station close to the current location of Iron Gate Dam. The earliest Chinook salmon date past this location between 1939 and 1958 was recorded as August 18, 1940; peak daily fish counts occurred during mid and late September and tapered off by late October (Shaw et al. 1997). More recent peak migration appears to occur one to four weeks later than the historic run timing on the Shasta and Klamathon Racks (Shaw et al 1997). In 2006, Chinook entered the Shasta River between mid September and mid December (Walsh and Hampton 2006) and Bogus Creek, adjacent to Iron Gate Hatchery between September 18 and November 25 (Hampton 2006). They reach spawning grounds in the Shasta and Scott Rivers as early as September. Spawning there 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 Chinook salmon migration occurs on the Trinity River between September and December with early migrating fish entering the larger tributaries first and use of smaller streams for spawning occurring later. Spawning on the Trinity River begins earliest downstream of Lewiston Dam but extends into late November downstream in the mainstem. Spawning in the South Fork began in mid-October (LaFaunce 1967). Spawning peaks during November in most Klamath and Trinity basin tributaries before tapering off in December (Leidy and Leidy 1984a).

Klamath River Chinook salmon have a lower fecundity and larger egg size compared to Chinook from the Sacramento River (McGregor 1922, 1923a). The average fecundity of Lewiston Hatchery fish is 3,732 eggs for 4-kg fish (Bartholmew and Henrikson 2006). Fry emerge from the gravel in the late winter or spring. The timing of emergence of fry is dictated by water temperature so the beginning of emergence may differ among years by over four weeks in the mainstem (Shaw et al. 1997).

Emigration timing of juveniles is highly variable and is dependent on river rearing conditions, which are controlled by water temperature. High winter flows, snowpack and subsequent spring runoff, summer weather conditions and smoke from forest fires (which can cool the water) all contribute to the 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. 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).

Sullivan (1989) examined scale growth patterns to study fry emigration patterns of returning fall run adults. Three distinct types of juvenile freshwater life history strategies for UKTR fall Chinook were identified by (Sullivan 1989): (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 the spawning areas as soon as they can and forage along the tributary and mainstem rivers for a short period, prior to emigrating during summer months into the estuary. Peak outmigration of fry occurs in March or early April in the Shasta River and between the middle of April to the middle of May in the Scott River. Historically, in the main Klamath River Chinook juvenile emigration started in mid-March before peaking in mid-June and decreasing by the end of July (Shaw et al. 1997). More recently (1997-2000), wild juveniles were not observed at 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 into the main stem during the spring and summer and rear there or in the estuary until ocean entry. Multiple juvenile fish kills in July and August (1997, 2000) highlight the extensive use of the middle and lower Klamath River during the summer by these juveniles (USFWS 2001). On the Lower Trinity River (0.4 rkm upstream of Weitchepec) naturally produced Chinook salmon emigration peaked around April 21. The first hatchery produced Chinook salmon were not observed until six weeks later in 2001 and emigration of these fish peaked in mid-October on the Lower Trinity River (Naman et al. 2004). Juveniles using this strategy may remain in the tributaries until autumn rains. The first two types of juvenile rearing strategy are likely influenced by mainstem flows. Wallace and Collins (1997) found that in low flow years Chinook salmon were more abundant in the Klamath River estuary than during high flow years, suggesting that the second strategy may be move fish into the cooler estuarine water sooner under low flow conditions.

Although the vast of majority of UKTR Chinook salmon use one of the two strategies described above, a small portion of juveniles spend an entire year in the river, mainly in the larger tributaries. Sullivan (1989) defined this third type of juvenile Chinook life history as individuals who reared in fresh water through their first winter before entering the ocean the following spring as yearlings. Between 1997 and 2000, these yearlings typically emigrated as smolts through the middle Klamath River between early May and mid-June, before the peak of 0+ wild juveniles in mid June (USFWS 2001). Yearling Chinook were captured in Bogus Creek between mid-January and mid May and at Big Bar, Presido Bar, and below the Scott River through mid-June (Shaw et al. 1997).

In the ocean, Klamath River Chinook salmon are found in the California Current system off the California and Oregon coasts. Salmon seem to follow predictable ocean migration routes and Chinook recaptured from the Klamath River generally use ocean areas that exhibit temperatures between 8° and 12°C (Hinke et al. 2005). Chinook salmon from the Klamath and Trinity hatcheries were observed in August south of Cape Blanco (Brodeur et al. 2004).

While there is significant variability in the age composition of Chinook spawners returning to the Klamath basin, typically a majority are age 3 fish, reflecting heavy mortality of older and larger fish in ocean fisheries. Some age 4 and age 5 fishes are observed, but they make up a smaller proportion of the total escapement than grilse. Grilse are small, two-year-old spawners. They constituted 2-51 percent of the annual Klamath River Chinook salmon numbers between 1978 and 2006 (Game 2006). Sullivan et al (1989) observed that a larger proportion of four year old Chinook returned to the Salmon River (24%), than other subbasins in 1986. In 1986, 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 2006, the Klamath River fall Chinook run was composed of two (31%), three (21%), four (47%), and five (1%) year old returns (KRTAT 2007). In 2004, the age structure of TRH fall Chinook run was composed of two (8%), three (78%), four (13%), and five (1%) year old fish (CDFG 2006a).

Habitat Requirements: The general habitat requirements of Chinook salmon are provided in the California Coastal Chinook account, including temperature requirements. UKTR fall-run Chinook salmon enter the Klamath estuary for only a short period prior to spawning. However, unfavorable temperatures can be found in the Klamath estuary and lower river during this period and chronic exposure of migrating adults to temperatures of even 17°-20°C is detrimental.

However, UKTR Fall run Chinook will migrate upstream in water as high as 23.5°C, if water temperatures are decreasing; 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 in UKTR fall run Chinook than in Columbia River fall run Chinook (>21°C; McCollough 1991). Optimal spawning temperatures for Chinook salmon are reported as less than 13°C (McCollough 1991) and fall temperatures are usually within this range in the Trinity River (Quilhillalt 1999). Magneson (2006) reported water temperatures up to 14.5°C during spawner surveys in 2005. The Shasta River historically was the system's most reliable spawning tributary from a temperature perspective (Snyder 1923), but diversions of cold water have greatly diminished its capacity to support salmon. Additionally, it is impaired by sediment. In six out of seven locations, Ricker (1997) found that levels of fines in potential Shasta River and Park Creek spawning habitats were high enough to significantly reduce fry emergence rates and embryo survival.

In the UKTR Chinook ESU, a majority of spawning habitat for fall run fish is found in larger tributaries and in the mainstem of the Klamath and Trinity Rivers. Spawning is primarily in habitats with large cobbles loosely imbedded in gravel and with sufficient flows for subsurface infiltration to provide oxygen for developing embryos. On National Forest land in the Scott River basin, a significant portion of such Chinook spawning habitat was generally in poor condition in 1990 (Olson and Dix 1992). In a survey of Trinity River Chinook redds, Evenson (2001) found embryo burial depths averaged 22.5 -30cm suggesting minimum depths of spawning gravels needed. Regardless of depth, the key to successful spawning is having adequate flow of water. Redds in the mainstem Trinity River averaged 14.5 long and 7.45ft wide (Moffett and Smith 1950) where the loosened gravels permitted access of oxygen-containing water. For maximum embryo survival, water temperatures must be between 6-12°C and oxygen levels must be close to saturation (Myrick and Cech Jr. 2004). 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. Water temperatures of 8°C were associated with initiation of fry emergence in the Scott and Shasta Rivers (Bartholow and Hendrikson 2006).

Water temperatures greater than 15°C stimulate juvenile emigration although temperatures above 15.6°C can increase risk of disease (McCollough 1999). Daily average temperatures above 17°C increase predation risks and impair smoltification while temperatures over 19.6°C decrease growth rates (Marine and Cech Jr. 2004). Temperatures up to 25°C are commonly encountered in the middle Klamath River during spring and summer juvenile emigration, so cool water areas at tributary confluences are important habitats during the day (Belchik 1997). Elevated river temperatures (>16°C) increase the mortality from *Ceratomyxa shasta* infection in Chinook salmon released from Iron Gate Hatchery, due to lethargy, reduced body mass, and co-occurring bacterial infections. Belchik (1997) identified 32 cool water areas in the middle Klamath River basin. Twenty-eight of these spots were tributary junctions, including that of the Scott River. These habitats have temperatures of 10°-21.5°C and provide refugia from temperatures lethal to emigrating juvenile Chinook (Belchik 1997). Belchik (1997) determined that the number of fish in these cool water areas was significantly related to the distance from Iron Gate Dam, proximity of the nearest other cool water area, and the minimum temperature of the areas.

Distribution: The range of the UKTR Chinook fall run includes three ecoregions (Coastal Range, Sierra Nevada, and Eastern Cascade; Meyer et al. 1998). UKTR Chinook salmon are found in all major tributaries above the confluence of the two rivers and two terminal hatcheries at Iron Gate and Trinity Dams. UKTR fall run Chinook salmon historically ascended to spawn in middle Klamath tributaries (Jenny Creek, Shovel Creek, and Fall Creek), 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 Dam and further blocked by the completion of a series of dams on the Klamath, concluding with construction of Iron Gate Dam in 1964. As a result, 663 km of migration, spawning, and rearing habitat in the Upper Klamath River basin was eliminated. Along the lower Klamath River numerous middle Klamath 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 each historically contained large numbers of spawning Chinook salmon and they are still among 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 between Bogus Creek and the Shasta River and between China Creek and Indian Creek (Magneson 2006).

UKTR 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). Historically, the majority of Trinity River UKTR fall-run Chinook spawning was located between the North Fork Trinity River and Ramshorn Creek; currently it is confined to the approximately 100 km between Lewiston Dam and Cedar Flat. Above Lewiston Dam, the Stuart Fork was an important historic spawning tributary, as were Browns and Rush Creeks (Moffett and Smith 1950). The distribution of redds in the Trinity River is highly variable. While the reaches closest to the Trinity Hatchery contain significant spawning, there is great variability in use of spawning habitat in reaches between the North Fork Trinity River and Cedar Flats (Quihiullalt 1999**).** Additional tributaries that contain spawning Chinook salmon in the Trinity River include the North Fork, New River, Canyon Creek, and Mill Creek. In the South Fork, fall run UKTR Chinook historically spawned in the lower 30 miles up to Hyanpom, and in the lower 2.7 miles of Hayfork Creek (LaFaunce 1967).

Abundance: While it is likely that UKTR spring Chinook were historically the most abundant run in the Klamath and Trinity Rivers (Snyder 1931, LaFaunce 1967), by the time records were being kept seriously, they had been reduced to a minor component of Klamath salmon. Therefore, estimates of Chinook salmon numbers in the two rivers are presumably primarily of fall Chinook. Snyder (1931) provided an early estimate for Klamath River Chinook runs of 141,000, based on the 1912 fishery catch of 1,384,000 pounds of packed salmon. Moffet and Smith (1950) estimated the Klamath River Chinook runs to approximate 200,000 fish annually, from commercial fishery data from between 1915 and 1943. USFWS (1979) combined these statistics to arrive at an annual catch and escapement of approximately 300,000 to 400,000 fish for the Klamath River system during the period 1915-1928. At the Klamathon Racks, a fish counting station close to the location of Iron Gate Dam, an estimated annual average of 12,086 Chinook were counted between 1925-1949, and the number declined to an average of 3,000 between 1956-1969 (USFWS 1979). In 1965, the Klamath River basin was believed to contribute 66% (168,000) of the Chinook salmon spawning in California's coastal basins (CDFG 1965). This production was 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 fish), Scott (8,000 fish), and Salmon (10,000 fish) Rivers. The Shasta River, which Snyder (1931) recorded as the best spawning tributary in the basin, has seen a marked decline in the number of fish returning. Leidy and Leidy (1984b) estimated an annual average abundance of 43,752 Chinook between 1930-1937; 18,266 between 1938-1946; 10,000 between 1950-1969; and 9,328 between 1970-1976. A review of recent escapement into the Shasta River found an annual escapement of 6,032 fish between 1978-1995, and an escapement of 4,889 fish between 1995 and 2006 (CDFG 2006b). In the Scott River, fall Chinook escapement averaged 5,349 fish between 1978-1996 and 6,380 fish between 1996 and 2006.

Coots (1967) estimated the annual run of Klamath River Chinook salmon to be 168,000, half of which ascended the Trinity River. Hallock et al. (1970) estimated 40,000 Chinook salmon entered the Trinity River above South Fork. Burton et al. (1977 in USFWS 1979) estimated 30,500 Chinook 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 and the estimated average declined between 1996 and 2006 to 23,463 fish (CDFG 2007).

More recently in the 1980s, the Klamath River Chinook stocks accounted for up to 30% of the commercial Chinook salmon landings in northern California and Southern Oregon, which averaged about 450,000 Chinook salmon per year (PFMC 1988). Total inriver escapement into the UKTR Chinook ESU ranged from 34,425 to 245,542 fish with an average 5-year geometric mean of 112,317 fish (Figure 1) between 1978 and 2006.

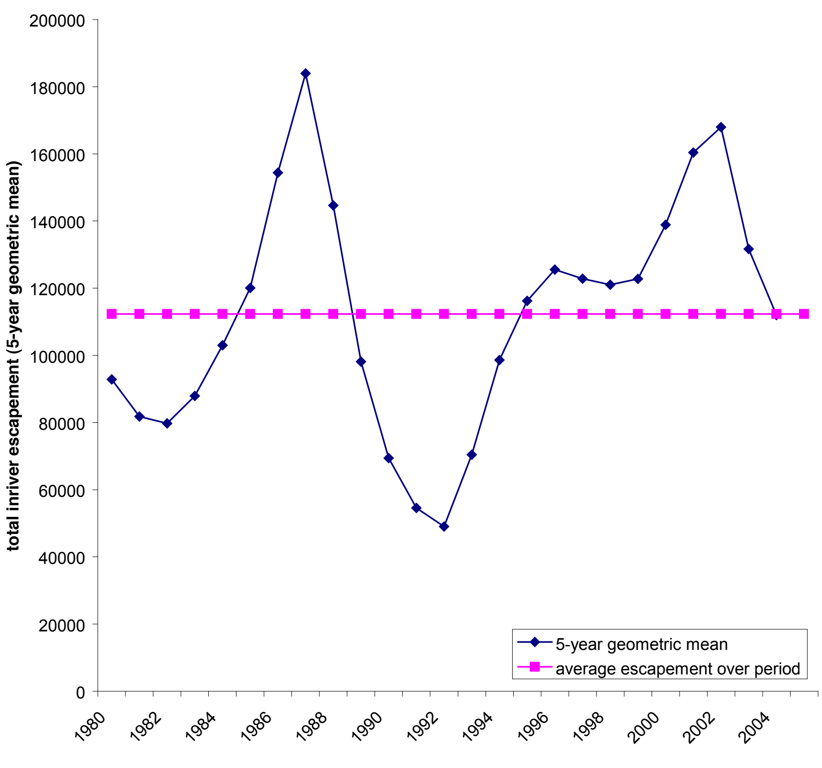


Figure 1: Trend between 1980 and 2005 for 5-year geometric mean of UKTR fall run Chinook salmon.

Hatchery operations have supplemented the abundance of UKTR Chinook salmon since completion of terminal hatcheries on the Klamath and Trinity Rivers in the 1960s. The origins of the hatchery stocks are principally from Klamath stocks and each hatchery relies on returning spawners for egg collection. Approximately 67% of hatchery releases have been fall-run Chinook from Iron Gate and Lewiston hatcheries (Myers et al 1998), with between 7 and 12 million juveniles released annually (NRC 2004). Between 1997 and 2000, an average of 61% of the juveniles captured at the Big Bar outmigrant trap were hatchery origin fish (USFWS 2001). At the Willow Creek emigrant trap on the Trinity River between 1997 and 2000, 53% and 67% of the Chinook captured in the spring and fall were hatchery-origin fish, respectively (USFWS 2001).

Factors affecting status: Numerous factors have influenced the status of UKTR Chinook salmon. These include dams, logging and other land use, fisheries, hatcheries, and disease.

Dams: UKTR fall Chinook are primarily mainstem spawners, so the big dams at Lewiston and Iron Gate have had an impact mainly by changing downstream habitat, and only secondarily by denying access to historic spawning areas (which were mostly below the dams). Iron Gate Dam and the chain of dams above it on the mainstem Klamath are used mainly for hydropower production, so they have had minimal impact on total flows below the dam (although water diversions to support agriculture in the upper Klamath basin do reduce the amount of water available for river flow). However, the dams have eliminated spawning gravel input from upstream and reduced hydrologic variability. The lack of adequate release of water from the dam is a factor blamed for the major fish kill in the lower river in September 2002.

Lewiston Dam and other dams on the Trinity River significantly reduced flows to the river, with all the attendant impacts of creating a smaller river. Starting in 1964, 75-90% of Trinity River flow was diverted to the Central Valley. Decline in naturally spawning fall Chinook populations were one result of this diversion of water. The decline resulted from reduced and degraded spawning and rearing habitats. In 1984, Congress ordered restoration of the river to support salmon at historic levels (see http://www.trrp.net/). Little was actually done for the river until The Trinity River Mainstem Fishery Restoration EIS was completed and the Record of Decision (ROD) was signed on December 19, 2000. The EIS calls for numerous restoration actions as well as a rough doubling of flows of the river, in a natural flow regime pattern. Implementation was delayed due to lawsuits until 2004, but is now underway (http://www.trrp.net/).

Logging and other land use: The majority of spawning and rearing habitat for UKTR Chinook salmon is surrounded by public lands in Klamath-Trinity National Forest, which have been heavily logged, roaded, and mined. As a result, the Klamath River, including spawning areas of fall Chinook, is regarded as impaired because of its sediment loads. In addition, elevated water temperatures have been identified as a factor limiting anadromous salmonids in the Klamath River basin, as the result of multiple land use factors combined with climate change. Water temperature has increased about 0.5°C/decade and has resulted in a loss of about 8.2 km of cool summer water in the mainstem each decade (Bartholow and Hendrikson 2006). Bartholow and Hendrikson (2006) also showed the timing of high temperatures that are potentially stressful to Chinook has also moved forward by about one month. These temperature changes are consistent with measured basin-wide air temperature increases. Resultant loss of rearing habitat, both temporally and spatially may influence the survival of UKTR fall Chinook. See UKTR spring Chinook account for a further description of impacts of logging and other factors.

Harvest: The Pacific Fisheries Management Council (PFMC) has paid close attention to Upper Klamath and Trinity River Chinook salmon in recent years because annual escapement goals have not met the Council's current objective of 35,000 natural adult spawners in most years. In November 2006, the PFMC accepted new fisheries guidelines that are supposed to result in natural spawning escapements of 22,000 -35,000 fish. This was considered a compromise to account for (1) recent critically low spawner abundances in consecutive years, (2) the risk that populations were dropping below critical genetic thresholds, (3) prevailing ocean conditions, and (4) Endangered Species Act considerations (PFMC 2007). Poor ocean conditions can severely impact escapement especially when combined with excessive harvest. Because of the combined conditions of both Central Valley and Klamath River salmon stocks, the ocean fishery (and probably the inland sport fishery as well) is likely to be greatly restricted for an extended period of time, unless a mark-selective fishery is allowed (all hatchery fish are marked).

Hatcheries: Although most tributary spawning stocks are comprised of a majority of wild fish, the spawning stocks in the mainstem Trinity and Klamath Rivers and those around hatcheries are comprised of mixed hatchery and wild Chinook salmon. The operation of hatcheries has likely influenced the age of maturation and spawning distribution of UKTR Chinook salmon. Hatcheries first began operating on the Klamath River for rearing and releasing fall run Chinook in 1914. 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. A significant proportion of mainstem spawning now occurs between Shasta River and Iron Gate Dam. The proportion of hatchery returns to total escapement has increased from 0.18 in 1978-82 to 0.26 in 1991-95 and 0.29 in 2001-2006 (CDFG 2007, Myers et al 1998). In 1999, 73% of redds were between Iron Gate Hatchery and the Shasta River and this proportion has increased through time (Bartholomew and Hendrikson 2006). Similar observations have been made on the Trinity River. More than 50% of outmigrating smolts observed at the Willow Creek outmigrant monitoring traps were hatchery fish between 1999 and 2000. This proportion increased to more than two-thirds during the fall monitoring period (USFWS 2001). This large number of hatchery fish may impact naturally produced Chinook juveniles through competition, predation, and disease transmission. Competition and predation may become a factor when releases of large hatchery juveniles flood shallow water refuge habitats used by naturally spawned juveniles (NRC 2004). This can be exacerbated by disease (next section). See Central Valley fall Chinook account for a further discussion of hatchery effects.

Disease: Chinook salmon in the Klamath and Trinity Basins emigrate as juveniles and return to spawn as adults when water temperatures and minimum flows begin to approach their limits of tolerance, increasing their susceptibility to disease. In September 2002, between 30,000 and 70,000 predominantlyUKTR fall run Chinook adult salmon died in the lower Klamath River. The immediate cause of death of fish was infection by ich disease (caused by the ciliated protozoan *Ichthyopthirus multifilis*) and columnaris disease (caused by the bacteria *Flavobacter columnare*) (Lynch and Riley 2003). Factors that led to the lethal infections are still not entirely clear but the die off appears to be the result of the combination of (1) high water temperatures, (2) crowded conditions, and (3) low flows. In response to high water temperatures and low flows, the fish apparently stopped migrating, concentrating in large numbers in pools. These conditions then allowed for the disease epidemic to sweep through the population of stressed fish. Increased base flows likely reduce pathogen transmission risk during Chinook migration (Strange 2007).

In juvenile Chinook salmon, high water temperatures and minimum flows can increase susceptibility to a number of other diseases. While the myxozosporean parasites common to the

Klamath River- *Ceratomyxa shasta* and *Parvicapsula minibicornis*- are often present, they are neither always abundant nor do they always encounter the conditions necessary for infecting large numbers of Chinook salmon. *C. shasta* appears in the mainstem and Upper Klamath River, Copco reservoir, both Klamath and Agency Lakes, and the lower reaches of the Williamson and Sprague Rivers (Buchanan et al. 1989, Hendrickson et al. 1989) and it is likely that UKTR fall run Chinook were historically infected by these diseases. Although the Shasta, Scott, and Trinity Rivers appear to be free of *C. shasta* (Foott et al. 2004), Trinity River smolts become infected with *C. shasta* while migrating through the Lower Klamath River and a majority of those infected salmon later die of Ceratomyxosis (Foott et al. 2002). We presume that juvenile Chinook from the Scott and Shasta are also not surviving their exposure during emigration and these diseases may therefore favor fall and winter outmigration by UKTR juvenile Chinook. When high densities of infected fish and warm temperatures are present in combination, *C. shasta* infection appears to be accelerated (Foott et al. 2003). *P. minibocornis* appears to be more infectious than *C. shasta* and was detected in 23% of juveniles in the Klamath estuary and 95% of juveniles in the Klamath River (Nichols et al. 2003).

Conservation: There are significant opportunities to adopt aquatic management strategies in the Klamath and Trinity Rivers to benefit UKTR Chinook salmon. The Trinity River Restoration Program provides for maintaining and potentially recovering healthy populations of UKTR Chinook salmon by taking a holistic approach to restoration. This approach involves using flows and restoration activities to focus on the habitat requirements of Chinook and other critical aquatic species in the riverscape. A similar program needs to be part of the Klamath River Restoration Program. Models evaluating limiting factors and habitat availability for UKTR Chinook salmon suggest that crucial steps need to be taken soon to increase UKTR fall Chinook spawners (Bartholow and Henrikson 2005) and restoration objectives that are part of the Trinity River Restoration Program provide feasible targets for ameliorating limiting factors and increasing habitat along the Trinity River. While the Salmon River and some smaller watersheds in the Klamath National Forest remain in relatively good condition, the Shasta and Scott Rivers need continued restoration efforts and improved water allocation to protect the salmon.

Water temperatures may be more important to UKTR Chinook salmon than a restored natural flow regime *per se*, although the two often go together. Bartholow (2005) modeled the changing thermal regime that could eventually eliminate UKTR Chinook spawning in the mainstem and disconnect critical spawning tributaries from the lower mainstem, an important migratory corridor. Both adults migrating upstream and juveniles moving downstream face water temperatures that are bioenergetically unsuitable or even lethal, especially in relation to disease. Protecting and restoring cool water habitats throughout the Klamath and Trinity Rivers and their tributaries will be essential to conserving UKTR Chinook salmon. The behavioral plasticity displayed by these fish indicates the potential biocoupling of UKTR Chinook life history with strategies that increase juvenile survival compared to if only a single juvenile life history was utilized. Along the mainstem, Belchik (1997) demonstrated that UKTR Chinook use cool water areas as refuges along the mainstem corridor, which increases outmigrant survival. These locations should be conserved, monitored, and if possible expanded.

Many of the suggestions for conservation of UKTR spring Chinook also apply to fall Chinook (see account).

Trends:

Short term: UKTR Chinook salmon abundance has experienced a major downward trend in the past 10 years, especially as a result of the 2002 kill in the lower river. While new challenges, including disease outbreaks and fisheries impacts have received attention by managing agencies, the modifications being undertaken may not be sufficient to adequate to restore UKTR Chinook salmon to historic numbers. Current efforts to modify or remove upstream hydroelectric dams, to increase our knowledge of ocean use patterns for UKTR Chinook to reduce fisheries impacts, to improve spawning habitats, and to increase monitoring of the population all offer opportunities for reversing the current decline of UKTR Chinook.

*Long term***:** Historic numbers of wild UKTR fall-run Chinook probably ranged between 125,000 and 250,000 fish per year. While numbers in the past 25 years have often reached into that range, much lower numbers are typical and many of the fish are of hatchery origin. There is little reason to be optimistic about long-term trends in the future without major changes in watershed management. High summer water temperatures are a major driver of UKTR Chinook survival and they are likely to increase under most climate change scenarios. Likewise, changes in ocean conditions may cause decreased survival of fish once they leave the river.

Status: 3. UKTR fall Chinook are not in danger of extinction although their numbers may be slightly declining. However, there is increasingly reliance on hatcheries to maintain fisheries and hatchery production is likely masking a decline of wild production in the Klamath-Trinity basins. The UKTR Chinook salmon ESU was determined to not warrant listing under the Endangered Species Act on March 9, 1998. UKTR fall Chinook are a US Forest Service Sensitive Species. They are managed by CDFG for sport, tribal, and ocean fisheries.

Metric	Score	Justification
Area occupied	3	Widely distributed in Klamath and Trinity basins
Effective pop. size	5	Abundant with several large populations
Dependence on Intervention	3	Presumably they would persist even without much human intervention, albeit in small numbers. Major intervention is required to maintain fisheries.
Tolerance	3	Moderate physiological tolerance, multiple age classes
Genetic risk	4	One genetically diverse population
Climate change	2	Climate change can reduce abundance but their 'ocean' life history strategy makes them least vulnerable of all runs, although warm temperatures in Klamath River threatened this part of population.
Average	3.3	20/6
Certainty	4	Most studied of Klamath River Chinook runs

Table 1. Metrics for determining the status of UKTR fall run Chinook salmon, where 1 is poor value and 5 is excellent.

UPPER KLAMATH-TRINITY RIVERS SPRING CHINOOK
Oncorhynchus tshawytscha

Description: The description in the Upper Klamath-Trinity Rivers (UKTR) fall Chinook salmon account generally applies to UKTR spring Chinook as well. However, UKTR spring Chinook salmon enter natal streams in the upper Klamath and Trinity Rivers as sexually immature adults during the spring season without the breeding colors or elongated kype seen in the fall Chinook salmon (Snyder 1931).

Taxonomic Relationships: The broader taxonomic relationships of this ESU are discussed in the UKTR Fall Chinook salmon account. The UKTR ESU includes both runs and is genetically distinguishable from other California Chinook ESUs (Banks et al. 2000, Waples et al. 2004). Members of this ESU are also genetically distinct from members of the Southern Oregon/Northern California Chinook salmon ESU, which spawn downstream of the confluence of the Klamath and Trinity Rivers.

Within the UKTR Chinook ESU, genetic analyses have demonstrated that stock structure mirrors geographic distribution (Banks et al. 2000). Fall and spring Chinook salmon from the same subbasin appeared more closely related than fall Chinook from adjacent basins. This pattern is distinct from Chinook of different run timings in the Sacramento and Columbia Rivers, where spring Chinook from different basins are more similar to each other than the fall Chinook found in the same basins. Furthermore, fall Chinook salmon populations from both the Klamath and Trinity subbasins appear more similar to the respective spring Chinook populations in the same subbasin than to fall Chinook in Lower Klamath River tributaries. While spring Chinook in the Smith River are placed in the Southern Oregon Northern California Coastal Chinook ESU, they have not been characterized genetically. It is likely that fish in this small run are derived from UKTR spring Chinook (Jim Waldvogel, UC Cooperative Extension, pers. comm. 2007), given the small population size of the run in the Smith River.

Despite the lack of strong genetic differentiation from UKTR fall Chinook, we treat the UKTR spring run as a distinct taxon because it represents a life history strategy (or distinct population segment) that is an essential adaptive component of the ESU and that requires separate management strategies. Historically, these fish were presumably on their own evolutionary trajectories before being derailed by human activities in the basin.

Life History: Adult UKTR spring Chinook salmon enter fresh water before their gonads are fully developed and hold in cold water areas for 2-4 months before spawning. They enter the Klamath estuary during spring and summer, starting 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 apparently are of hatchery origin (Barnhardt 1994; NRC 2004) and Leidy and Leidy (1984) noted that the adult Trinity River spring Chinook migration continued until October. However, given this late timing, it is unclear if these fish are sexually mature and able to spawn with spring Chinook adults already in the system. Because this late spring run is limited to the Trinity River, it is possible these fish represent hybrid spring and fall Chinook created by hatchery practices. The Trinity River hatchery classified Chinook entering between September 3 and October 15 in 2004 as spring Chinook (CDFG 2006). Moffett and Smith (1950) noted spring Chinook migrate quickly through the watershed and over obstacles; more recent work (Strange 2005) has confirmed this rapid migration pattern in the Trinity River. While migration occurred throughout the day and night, there was a peak in movement during the two hours following sunset (Moffett and Smith 1950).

Spawning starts in mid-September in the Salmon River; in the Trinity River basin spawning usually begins in early October. Trinity River spawning typically is 4-6 weeks earlier than that of fall UKTR Chinook (Moffett and Smith 1950). Spring Chinook in the South Fork Trinity River begin spawning in late September with a peak in mid-October (LaFaunce 1967). Overlap between fall and spring Chinook spawning areas historically was minimal. In the South Fork Trinity, the majority of spring Chinook spawning occurred above Hitchcock Creek above Hyampom Valley, while fall Chinook spawned below this point (LaFaunce 1967, Dean 1995). Moffett and Smith (1950) stated spawning of the fall and spring runs overlapped in October on suitable spawning riffles between the East Fork and North Fork, and redd superimposition and hybridization may have occurred. In the Salmon River, an overlap exists between spawning times of fall and spring Chinook, although redds constructed upstream of the confluence of Matthews Creek are predominantly of spring Chinook origin (Olson et al. 1992). Overall, spatial separation between the two runs in the Klamath-Trinity system occurs at approximately 1,700 ft.

UKTR spring Chinook fry emerge from the 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 Dam became the upper limit for migration on 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 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 Chinook populations north of the Klamath River (e.g., Columbia River) UKTR spring Chinook do not consistently display "stream type" juvenile life histories, where juveniles spent at least year in the stream before migrating to the ocean (Olson 1996). Juvenile emigration occurs primarily from February through mid-June (Leidy and Leidy 1984). This may be earlier than UKTR fall Chinook salmon where between 1997 and 2000, natural juvenile Chinook salmon were not observed emigrating past Big Bar (rkm 91) earlier than the beginning of June with a peak in mid-July (USFWS 2001). On the Salmon River, a tributary refuge for spring Chinook along the Klamath River, two peaks of juvenile emigration have been observed: spring/early summer and in the fall. Snyder (1931) examined scales from 35 adult spring Chinook and 83% displayed juvenile "ocean type" growth patterns (see Central Valley spring Chinook account for discussion of Chinook juvenile "types"). In the Salmon River, an otolith study (Sartori, unpublished) identified 31% of fall emigrating juvenile Chinook salmon as having similar growth patterns as Salmon River spring Chinook.

Other aspects of their life history are similar to UKTR fall Chinook and Chinook salmon in general (Moyle 2002),

Habitat Requirements: UKTR spring Chinook enter the Klamath estuary during a period when river water temperatures are at or above optimal holding temperatures (see table of temperature tolerances in the California Central Coast Chinook ESU account). Temperatures in the Lower Klamath River typically rise above 20°C in June and can attain 25°C in August. Spring Chinook use thermal refuges in the estuarine salt wedge and associated nearshore ocean prior to entering fresh water (Strange 2003). Strange (2005) found adult migration changed with different temperature trajectories. When daily water temperatures were increasing, Chinook migrated upstream until temperatures reached 22°C, while when temperatures were decreasing fish continued to migrate upstream at water temperatures of up to 23.5°C. A cool water refuge at the confluence of Blue Creek was used by 38% of spring Chinook for more than 24 hours in 2005 (Strange 2005). Optimal adult holding habitat is characterized by pools or runs greater than one meter 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). Because the Salmon River and its forks regularly warm to summer daytime peaks of 21-22°C, presumably the best holding habitats are deep pools that have cold water sources, such as those at the mouths of tributaries, or are deep enough to be subject to thermal stratification.

For UKTR spring Chinook, a majority of spawning habitat is found on low gradient gravelly riffles and at pool tail outs. Spawning and redd construction appears to be triggered by a change in water temperature, rather than an increase in flows, and redd superimposition may occur when suitable habitat is limited above holding pools. Thus redd superimposition has been noted among spring Chinook spawning in the South Fork Trinity River (Dean 1995). West (1991) noted that spring Chinook survival to emergence ranged from 2-30% on the Salmon River in 1990. Juvenile habitat requirements for spring UKTR Chinook salmon are similar to fall UKTR Chinook salmon.

Distribution: UKTR spring run were once found throughout the Klamath and Trinity basins, using suitable reaches in the larger tributaries (e.g., Salmon River) or, flows permitting, in smaller tributaries for holding and spawning. Historically, they were especially abundant in the major tributary basins of the Klamath and Trinity Rivers, such as the Salmon, Scott, Shasta, South Fork and North Fork Trinity Rivers. Their distribution is now restricted by dams that block access to the upper Klamath and Trinity Rivers. Passage of spring Chinook into Upper Klamath Lake, to attain holding and spawning grounds on the Sprague, Williamson and Wood Rivers, was blocked below Klamath Falls in 1895 by construction of Copco 1 Dam (Hamilton et al. 2005).The construction of Dwinnell Dam on the Shasta River eliminated access to UKTR spring Chinook habitat in that watershed. Today, only the Salmon River and its two forks maintain a viable population in the Klamath River basin. Approximately 177 km of habitat is accessible to spring Chinook in the Salmon River (West 1991) but most of it is underutilized or unsuitable. The South Fork Salmon River holds the majority of the spawning population but smaller tributaries where spring Chinook redds have been found in the Salmon River basin include Nordheimer, Knownothing, and Methodist Creeks In addition, there are dwindling populations of spring Chinook in Elk, Indian, Clear and Wooley Creeks.

In the Trinity River basin, spring Chinook salmon historically spawned in the East Fork, Stuart Fork, Coffee Creek, and the mainstem Upper Trinity River (Campbell and Moyle 1991). In 1964, Lewiston Dam was completed, blocking access to 56 km of spawning and nursery habitat on the mainstem (Moffett and Smith 1950). Currently, Trinity River spring Chinook are present in small numbers in Hayfork and Canyon Creek, as well as in the North Fork Trinity, South Fork Trinity and New Rivers. LaFaunce (1967) found spring 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. The highest density of redds in the South Fork Trinity was between 60.7 and 111.8 rkms in 1964 (LaFaunce 1967) and 1995 (Dean 1995).

Abundance: UKTR spring Chinook populations once likely totaled more than 100,000 fish

(Moyle 2002). The spring run was apparently the main run of Chinook salmon in the Klamath River, but by the end of the 19th century it was depleted as the result of hydraulic mining and commercial fishing (Snyder 1931). In each of four Klamath tributaries alone, historic run sizes were estimated by CDFG (1990) to be at least 5,000: Sprague River (Oregon), Williamson River (Oregon), Shasta River, and Scott River. The runs in the Sprague, Wood, and Williamson Rivers were probably extirpated in 1895 after the construction of Copco 1 Dam. Approximately 500 total fish returned to Iron Gate Hatchery each year during the 1970s (Hiser 1985), but the hatchery was not able to maintain this run without a source of cold summer water. The last spring Chinook returned to the hatchery at in 1978. 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 access to upstream spawning areas by Dwinnell Dam, which was erected in 1926. The smaller Scott River run was extirpated in the early 1970s from a variety of anthropocentric causes that depleted flows and altered habitat (Moyle 2002). Along the middle Klamath, spring Chinook are extirpated from their historic habitat except in the Salmon River and Wooley Creek (NRC 2004). Less than 10 spring run Chinook are annually observed in Elk, Indian, and Clear Creeks (Campbell and Moyle 1991).

In the Salmon River, spring Chinook summer counts show high variability among years but no recent downward trend, although the lowest counts have been in recent years (Figure 1) The 2005 adult count estimate was 90 fish, the lowest on record, but in 2007 the number reached 841. The numbers in Wooley creek ranged from 0 to 81 during 1968-1989, but more recent surveys suggest spring run Chinook are nearly extinct in this watershed. In 2005, only 18 spring run Chinook were observed.

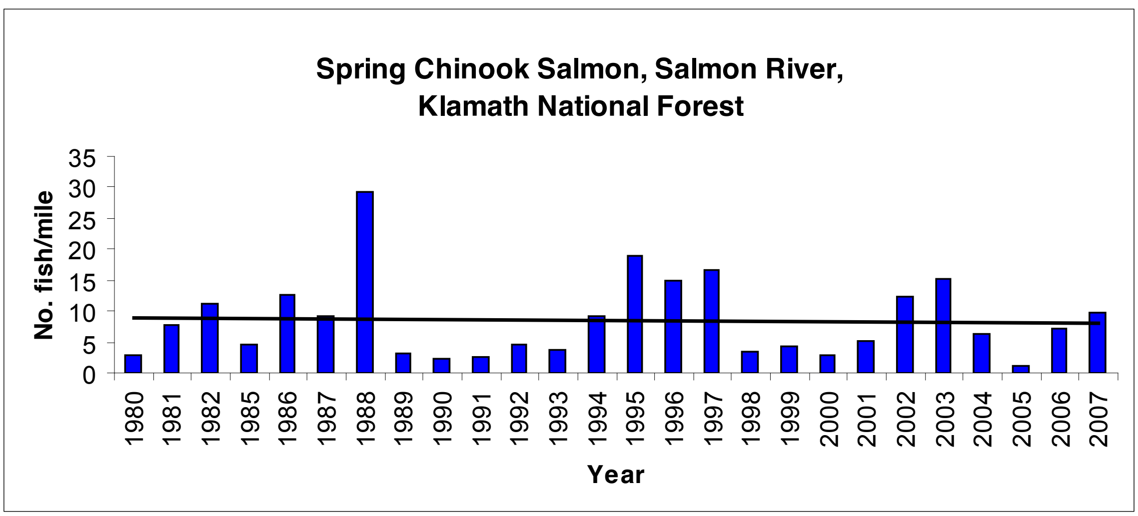


Figure 1. Number of adult spring run Chinook salmon per mile observed in the Salmon River and its forks, 1980-2007. The number (out of about 77 miles of river) of reaches surveyed varied from year to year although the entire river has been surveyed in most recent years. Analysis by Rebecca Quiñones, Klamath National Forest.

In the Trinity River, spring Chinook runs above Lewiston Dam are now extinct, but historically included more than 5,000 adults in the upper Trinity River and 1,000-5,000 fish each in the Stuart Fork Trinity River, East Fork Trinity River and Coffee Creek (CDFG 1990). An average of 263 fish have been counted annually, over about the last thirty years, in the South Fork Trinity River with runs being as low as 59 (1988, 2005) and as high as 1097 (1996). Between 1980 and 1989, an average of 142 spring run Chinook were counted annually in the South Fork Trinity River; 351 fish between 1990 and 1999; and most recently 232 fish between 2000-2005. Historically, 7,000 - 11,000 spring Chinooks entered this stream (LaFaunce 1967) and outnumbered fall run Chinook in the watershed. Between 1980 and 2004 an average of 18,903 spring Chinook returned above Junction City on the mainstem Trinity River. In 2004, 16,147 spring Chinook salmon were estimated to migrate into this area with 6,019 (37%) fish entering Trinity River Hatchery being classified as spring Chinook.

Overall, while spring Chinook salmon are still scattered throughout the lower Klamath and Trinity basins, the only viable wild population appears to be that in the Salmon River. Trinity River fish numbers are presumably largely influenced by fish from the Trinity River hatchery. Even if Trinity River tributary spawners are considered to be wild fish, the total number of spring Chinook in the combined rivers rarely exceeds 1000 fish and may drop to <300 in many years.

Factors affecting status: UKTR spring Chinook have been largely extirpated from their historic range because their life history makes them extremely vulnerable to the combined effects of dams, mining, habitat degradation, and fisheries, as well as multiplicity of smaller factors. Here we discuss mainly factors that most strongly affect spring Chinook; other factors are discussed in the UKTR fall Chinook account.

Dams: A significant portion of the historic UKTR spring run Chinook habitat has been lost behind Lewiston, Iron Gate, and Dwinnell dams. Iron Gate dam blocked access to the largest amount of habitat and there are currently about 970 km of anadromous habitat of varying quality upstream of it (Hamilton et al. 2005). These barriers to adult holding habitat and spawning grounds and juvenile nursery areas have reduced the resilience of spring Chinook populations due to smaller population sizes, loss of available habitat, and reduction in spatial isolation between spring and fall Chinook. This has likely led to significant introgression between fall and spring Chinook in the Trinity River (Myers et al 1998). Dams have also led to the extirpation of spring Chinook in the Klamath and Shasta Rivers due to alteration in water quality and temperature, channel simplification, and disconnection from floodplain.

Logging: Logging and its associated road building are a pervasive negative influence on aquatic habitats in the Klamath and Trinity River basins (NRC 2004). Logging has been altering watersheds in the basins since the 19th century (see SONCC coho account for description of legacy effects of logging) and continues to have an impact. The steep and unstable slopes of the region make them particularly prone to erosion following tree removal, pouring large amounts of sediment into the streams, imbedding spawning areas and filling in pools needed for holding over the summer. Thus, the low numbers of spring Chinook salmon currently using the heavily-logged South Fork Trinity River may be a result of the catastrophic 1964 flood, which triggered landslides that filled in holding pools and covered spawning beds. Other logging effects include elimination of large trees that historically fell into the river and were used for cover by the salmon and loss of shade (especially on tributaries), increasing water temperatures. As discussed in the UKTR fall Chinook account, increasing temperatures are a growing problem for salmonids in the basin. The altered forests have also become more prone to large-scale, damaging fires. For example, over 50% of the Salmon River watershed, the main refuge for UKTR spring Chinook, has been severely burned in the past 100 years (NRC 2004).

Mining: Mining, mainly for gold, has both legacy and ongoing effects. Some of the most damaged habitat is the legacy of hydraulic and dredge mining in the 19th century. Presumably this activity largely wiped out spring Chinook from many areas such as the Scott River and large areas in the Trinity River, followed by some recovery after large-scale mining ceased. But long reaches of ruined river still exist, such as the Scott River in the Scott River Valley, where a depleted river winds through immense piles of dredge tailings. The mining legacy still affects the Salmon River spring Chinook population while the estimated 16 million cubic yards of sediment disturbed between 1870 and 1950 are slowly transported through the basin (J. West, U.S. Forest Service, personal communication, 1995). This activity has disconnected and constricted juvenile salmon habitat, filled in adult holding habitats, and degraded spawning grounds. Pool in-filling is particularly a problem because high stream temperatures reduce survival of both holding adults and rearing juveniles (West 1991, Elder 2002).

Mining continues throughout the basins and is likely increasing as the price of gold increases. Particularly damaging to spring Chinook is instream suction dredge mining. Suction dredging represents a chronic unnatural disturbance (noise and turbidity) of natural habitats that are already stressed by other factors and can therefore have a negative impact on salmon that use areas being dredged. Direct effects include entrainment of invertebrates (food for juveniles) and small fish in the dredges, altering of the habitat that supports the food supply of fishes, and changing channel structure in ways that make it less favorable for fish (usually by making it less stable and complex). Instream mining also decreases water clarity, decreasing efficiency of foraging of juveniles. An area of particular concern in the Klamath, Salmon and Scott Rivers and their tributaries is the creation of piles of dredge tailings that are attractive for the spawning of salmonids but that are so unstable they are likely to scour under high flows, greatly reducing survival of the embryos placed within the gravel. Equally important is that suction dredging (and the constant presence of people in sections of river) can be a continuous disturbance to holding adults and juveniles during summer, increasing stress and probability of premature death. For more details on the effects of suction dredging see Harvey and Lisle (1998).

Rural development: The long history of mining and logging in the Klamath and Trinity basins has left the region honey-combed with roads which provide access to many remote areas. This has resulted in people living throughout the basin, on mining claims, small farms, and communities. This diffuse rural development undoubtedly has an impact on spring Chinook salmon, through the cumulative effects of recreational disturbance (e.g., swimming, fishing), small water diversions, sediment from roads, toxic spills, and other impacts.

Harvest: Both illegal harvest of holding adults, as well as legal harvest of fish in the ocean and river can reduce spawning populations. Holding adults are extremely vulnerable to illegal take, although this is largely undocumented. However, the general absence of spring Chinook from populated areas or areas with easy access suggests this is factor. Because UKTR spring Chinook are not considered by agencies as distinct from fall Chinook, they are taken legally in sport and commercial fisheries. Removal of even a small number from the population by this means presumably has an effect, if not known.

Hatcheries: The only hatchery in the Klamath Basin that still cultures spring Chinook salmon is the Trinity River Hatchery below Lewiston Dam. The impact of the hatchery on spring Chinook salmon in the Trinity Basin is presumably large; it is likely that a majority of the naturally spawning fish, especially in the mainstem, are of hatchery origin (Barnhart 1994). Hatchery spring Chinook are also most likely to hybridize with fall Chinook.

Disease: Disease has risen as a major limiting factor for salmon in the Klamath Basin ever since the major die-off of fall Chinook in September 2002. But other die-offs of juvenile and pre-spawn adult UKTR Chinook have also occurred during the past decade (USFWS 2002). However, the impact of these events on already depressed stocks of spring Chinook is unknown. Further discussion of recent events and the understanding of biologists about diseases are expanded in the UKTR fall Chinook section. *Ceratomyxa shasta* appears in the mainstem and Upper Klamath River, Copco reservoir, both Klamath and Agency Lakes, and the lower reaches of the Williamson and Sprague Rivers (Buchanan et al. 1989; Hendrickson et al. 1989). It is likely that UKTR spring juveniles and adults Chinook were historically infected by these diseases. While UKTR spring Chinook do not show a rigid "stream-type" juvenile emigration strategy, this strategy may show reduced mortality because these fish remain out of the mainstem during warmer temperatures when disease is most likely an issue. Warmer temperatures favor epizootic outbreaks of *Ichthyophthirius multifiliis* and transmission of the bacteria *Columnaris*. Columnaris disease is associated with pre-spawn mortality of spring Chinook that are exposed to above-optimal water temperatures. Increased base flows likely reduce pathogen transmission risk during Chinook migration (Strange 2007).

Conservation: Monitoring of spring Chinook occurs annually across the system. These efforts demonstrate that habitat exists for adult holding and spawning, yet spring Chinook have not increased in distribution or abundance and remain on the verge of extinction. Oversummering behavior and habitat requirements are the most distinctive features of spring run Chinook. The rarity of cool water refuges throughout the UKTR Chinook ESU region is a significant threat to spring Chinook survival and recently even the fall Chinook have been greatly impacted by lack of cool water. Reconnecting historic habitats in the Klamath and Trinity Rivers and their tributaries is necessary for long term persistence of these fish. This effort would increase habitat availability for spring Chinook and remove barriers, which negatively impact water quality and quantity. UKTR spring Chinook are an indicator species due to their sensitivity to water quality, temperature, and presence during some of the most challenging months for riverine inhabitation. The near extirpation of this sentinel species in the Klamath River subbasin indicates potential future problems for other anadromous stocks that rely on freshwater habitats during the juvenile and adult life histories. Some actions that could improve the situation for spring Chinook in the Klamath and Trinity basins include:

- List the UKTR spring Chinook as a threatened species under both state and federal endangered species acts, after declaring it a Distinct Population Segment within the ESU. This would give it the attention it needs for survival.
- Remove dams on the mainstem Klamath to allow access to historic upstream spawning and rearing areas. Spring Chinook are probably the species that would benefit the most from this action.
- Restore the Shasta River as a cold-water refuge for all salmonids in the Klamath Basin by recapturing spring flows in the river and removing Dwinnell Dam.
- Manage the Salmon River as a spring Chinook and summer steelhead refuge, by restricting use of the river in summer (e.g., ban suction dredging).
- Investigate the impact of the Trinity River Hatchery on spring Chinook populations and manage the hatchery accordingly.
- Place a high priority on reducing the impact of roads, logging, and other activities on sediment production in the rivers, especially on public lands.

- Determine the impact of sport, commercial, and traditional fisheries on UKTR spring Chinook to improve fisheries management.

Trends:

Short term: The numbers of spring Chinook in the Klamath and Trinity River have remained at low levels for the past 20 years with no obvious trends, but numbers are so low, especially in the Salmon River, that extirpation is a distinct possibility. Trinity River spring Chinook appear to rely on the Trinity River Hatchery for persistence.

Long term: UKTR spring Chinook have declined from being the most abundant run in the basin, to being a tiny run in danger of extinction. There are multiple possible futures for this distinctive salmon. The two extremes are extinction and restoration to a large segment of its historic range. At the present time it is headed for extinction. Climate changes will lead to increased water temperatures and fluctuations in many portions of the basin. Without drastic management measures, climate change will likely be the final blow to wild spring Chinook in the Klamath Basin. The run will then simply be a remnant hatchery run in the Trinity River for a few decades before it finally becomes so introgressed with the fall run so that loses it genetic and life history distinctiveness. Alternately, there is potential for UKTR spring Chinook salmon to be restored to large portions of the Klamath basin through a few decades of restoration of habitat and habitat access (e.g., Shasta River, upper Klamath Basin). While these regions will continue to warm into the future, there is potential for more precipitation around Mt. Shasta that will replenish cold water sources for the Shasta River.

Status: 2. Given the fluctuating nature and small size of the Salmon River population and its localized distribution in a single watershed, UKTR spring Chinook are vulnerable to extinction in the next 50-100 years (Table 1). Essentially, the only viable wild population today is in the Salmon River. Other populations are either small and intermittent or heavily influenced by hatchery fish, so may not be self-sustaining and are likely to be extirpated in the near future. Spring Chinook are a CDFG Species of Special Concern and qualified to be added to the state and federal lists of threatened or endangered fish (Moyle et al. 1995). They are also considered a Sensitive Species by the Pacific Southwest Region of the US Forest Service.

Metric	Score	Justification
Area occupied	2	Multiple populations exist including hatchery populations but only Salmon River is viable
Effective population. size	2	Although there is a hatchery stock, there are few natural spawners support the population.
Dependence on intervention	3	Hatchery program in Trinity is probably maintaining the Trinity run. The Salmon River wild population is vulnerable to extinction from both local and out-of-basin events. More human intervention necessary to preserve Klamath stock by re-establishing populations.
Tolerance	2	Temperature and other factors in summer holding areas may exceed physiological tolerances.
Genetic risk	2	Hybridization may be occurring in some watersheds with fall-run fish; populations are low enough so genetic problems can develop.
Climate change	1	The Salmon River has temperatures in summer (21-23°C) that approach lethal temperatures. A 1-2°C increase in temperature could greatly reduce the amount of suitable habitat.
Average	2.0	12/6
Certainty	3	Monitoring efforts by USDA Forest Service, CDFG, tribes and local organizations give us reasonable information about status.

Table 1. Metrics for determining the status of Upper Klamath/Trinity River spring Chinook salmon, where 1 is poor value and 5 is excellent.

CALIFORNIA COAST CHINOOK SALMON
Oncorhynchus tshawytscha

Description: 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 Chinook adults are the largest Pacific salmonid, typically 75-80 cm SL, but lengths may exceed 140cm. California Chinook are usually smaller and Puckett (1972) found that the average size of Eel River Chinook was 56 cm FL. The average weight is 9-10 kilograms, although the largest Chinook 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. Juvenile Chinook 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 Evolutionary Significant Unit (ESU) from other Chinook salmon ESUs, so the separation is based on genetic data.

Taxonomic Relationships: The California Coast Chinook salmon (CC Chinook) ESU includes Chinook salmon that spawn in coastal watersheds from Redwood Creek (Humboldt County) in the north to the Russian River in the south, inclusive. Chinook salmon found occasionally in coastal basins south of the Russian River (e.g., Lagunitas Creek, Marin County) are also considered to be in this ESU. Recent genetic analyses with microsatellite loci and reanalysis of older allozyme datasets demonstrate moderate levels of differentiation among populations. Bjorkstedt et al. (2005) concluded that CC Chinook in the Eel River and northern watersheds differ from those on the Mendocino coast and the Russian River. Differentiation among fish from different tributaries to the Eel River is low, suggesting high dispersal among tributaries in the basin by Chinook. Additionally, 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 (Bjorkstedt et al. 2005). With a lack of data on the genetic structure of the ESU's two largest populations (Eel and Russian Rivers), it is difficult to know if they are independent populations that are important elements of the ESU's historic population structure or if the two population are similar due to the derivation of Russian River fish from Eel River fish or earlier hatchery runs (see below).

Life History: California Coast Chinook salmon are fall-run salmon. Historically, this ESU included spring-run Chinook salmon but because runs with this life history strategy have apparently been extirpated our discussion is limited to fall-run fish. There is 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 coastal watersheds. CC Chinook typically return to their natal rivers between September and early November following early large winter storms.

Entrance into fresh water is often delayed in smaller coastal watersheds when sand bars across the mouth and other low flow barriers prevent access until December or even January (M. Sparkman, DFG, pers. comm.). 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. CC Chinook salmon may spawn immediately or may rest in holding pools for considerable time when early storms permit entrance to rivers but do not permit access to preferred spawning habitat upstream. Mature 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, presumably increasing carrying capacity for their own young.

The vast of majority of CC Chinook salmon demonstrate an "ocean-type" juvenile life stage. Fry emerge from the gravel in the late winter or spring and initiate outmigration within a week to months of emergence when they are 30-50 mm FL. Emigration of smaller fish is likely a function of a 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, 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 night, when the fish hide in deep cover to reduce predation and for energy conservation. Small numbers of "stream-type" parr will over summer 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, CDFG, pers. comm.).

Estuaries and transitional habitats between river and ocean are important for Chinook salmon survival to changing environments. CC Chinook may reside in estuaries, lagoons, and bays for a few months, gaining in size, and then exit these habitats gradually over the summer. Historically, estuaries with summer access to the ocean were favorable juvenile habitat and fish had greater flexibility to leave or to remain in the estuaries until fall storms dispersed them into the ocean. The extended occupancy by smoltifying Chinook of these habitats suggests enhanced growth may benefit ocean survival. 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 peaked in early June and none were captured past July 28 (Cook 2005). 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.

Once they enter the ocean, CC Chinook salmon migrate along the California coast, often moving northward. Ocean productivity plays a large role in their survival and growth, so oscillations and shifts in marine productivity influences their abundance. Chinook salmon are predators in the ocean, feeding on small fish and crustaceans. As their size increases, fish increasingly dominates their diet. This piscivorous diet provides for rapid growth, to the order of 0.35-0.57mm/day (Healey 1991). In California, 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 a selective effect of fisheries 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 (Table 1). Likewise, they are sensitive to dissolved oxygen levels, water clarity and other factors that indicate high water quality.

Chinook spawning use the largest substrate of any California salmonid for spawning, a mixture of small cobble and large gravel. Such coarse material has sufficient flows for subsurface infiltration, which provides oxygen for developing embryos and removes their metabolites. 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 spawning habitat is in the upper main stems of rivers and lower reaches of coastal creeks. These habitats, when in proper condition, provide stable substrate and sufficient flows into late winter. Typically, redds are observed at depths from a few centimeters to several meters and at 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 oxygen-containing water (Healey 1991). However, because females dig the redds, 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. 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.

	Sub-Optimal	Optimal	Sub-Optimal	Lethal	Notes
Adult Migration	<10ºC	10-20°C	20-21°C	>21-24°C	Migration usually stops when temp. 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 refugia.
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 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% > 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.
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; lab studies suggest optimal temperatures are 13-17ºC (Marine and Cech 2004) but observations in the wild indicate a greater range.

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). Information largely from McCullough (1999).

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 available, 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 juveniles just before they go out to sea,

In the ocean, habitats for the first few months are poorly documented, but it is assumed that the fish stay in coastal waters where the cold California Current creates rich food supplies, especially small shrimp, by upwelling. During the day, they avoid surface waters. Subadult Chinook salmon swim about in pursuit of anchovies, herring, and other small fish, typically at depths of 20-40 m, moving off shore 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 County) in the north to the Russian River in the south, inclusive. Chinook salmon found occasionally in coastal watersheds south of the Russian River are also considered to be in this ESU. 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 Region*, the Upper Eel and Middle Fork Eel Rivers contain independent CC Chinook stocks while the Lower Eel and Van Duzen Rivers have the potential to support viable populations. Chinook are annually observed in the Middle Fork Eel River, in Black Butte River, and near Williams Creek. They continue to be observed annually in the Outlet Creek drainage and in the smaller tributaries feeding Little Lake valley (Scott Harris, CDFG, pers. comm.). In the *North Coastal Region*, Redwood Creek and the Mad, Lower Eel, South Fork Eel, Bear and Mattole Rivers all contain sufficient habitat for independently viable CC Chinook salmon populations. NMFS also identified Little River and Humboldt Bay tributaries as containing potentially independent populations. In the *North-Central Coastal Region*, numerous watersheds in Mendocino County contain (or contained) small runs of CC Chinook that are dependent for persistence upon self-sustaining stocks in Ten Mile, Noyo, and Big Rivers. Along the *Central Coastal Region*, the Navarro, Garcia and Gualala Rivers historically had independent populations but apparently no longer do. Additionally, the Russian River appears to support a self-sustaining population although the role of hatcheries and straying from the Eel River (by fish attracted to Eel River water which has been diverted into the Russian River) is uncertain (Chase et al. 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 fish (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 and are not included in the ESU.

Abundance:

North Coastal region: CC Chinook that inhabit the northern portion of the ESU, between Redwood Creek and Humboldt Bay, appear to have annual runs of a few hundred spawners annually. The Mad River hatchery raised Chinook salmon until 2003; between 38 and 656 adult salmon returned to the hatchery between 1971 and 1989, but hatchery escapement declined in the 1990s to range between 0 and 62 fish. These returns are a poor indicator of CC Chinook population abundance in the Mad River and a creel survey estimated an average of 631 Chinook were caught-and-released by fishers annually between 1999-2003 (Sparkman 2003). Within Humboldt Bay, the smaller coastal tributaries also likely supported combined runs of several hundred fish. Presumably, CC Chinook runs in many of these steep coastal tributaries such as Freshwater Creek and 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 (Mike Wallace, CDFG, pers comm.), resulting in higher-than-expected numbers of returning adults. Chinook salmon have been observed in declining numbers in Freshwater Creek over the past decade. Chinook salmon 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 Mattole River contains a CC Chinook population that likely contains up to 1000 spawners annually (Campbell Thompson, MSG, pers. comm.).

North Mountain Interior Region: Historic abundance of Chinook salmon in the entire Eel River system was estimated by Steiner Environmental Consulting (1998), based on historic cannery records compiled by Humboldt County (Humboldt Public Works 1991). For the period of record, 1857-1921, SEC (1998) estimated that the average catch was 93,000 fish per year with

Chinook and coho salmon combined, 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 the most abundant and accessible fish in the fishery. If we assume that the catch was 90% Chinook salmon, then an average catch of Chinook would be 85,000 fish per year, with a maximum of 525,000. There are no records of how many fish actually escaped up-river to spawn, but a conservative estimate would be that the annual runs of Chinook in the Eel River (catch + escapement) in this period were on the order of 100,000-600,000 fish per year.

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 the historic escapement given that the Potter Valley Diversion Project was almost forty years old at the time and Chinook were facing challenges from flow alteration, habitat degradation, pollution, 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 had declined to less than 5,000 fish annually and have continued to 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 Reservior (Puckett and Hinton 1974).

In the Upper Eel River, an estimated 367 Chinook salmon entered Tomki Creek, the most productive upper mainstem tributary below Van Arsdale Reservoir, to support this run in 1975-76 (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 Fisheries Station provides an estimate of the Chinook entering only the Upper Eel River and in 2006-07, 700 Chinook passed this location. In all probability, a number of the larger subbasins in the Eel River such as the Van Duzen, South Fork Eel and North Fork Eel Rivers continue to support spawning runs, although monitoring data is extremely limited. 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 (Harry Vaughn, Eel River Salmon Restoration Program, pers. comm.).

North-Central Coastal region: Monitoring data is sparse for coastal Mendocino tributaries such as Ten Mile, Noyo, Big, Navarro, Garcia, and Gualala Rivers, and while CC Chinook are occasionally reported in these watersheds, they likely do not currently support viable populations. Early logging practices likely extirpated 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.

Central Coastal region: CC Chinook in the Russian River are of uncertain genetic origin following close to fifty years of interbasin 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 intrabasin locations to establish a self-sustaining hatchery run. Unfortunately, returns were very low and ranged from 0 to 304. 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 the Agency'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. Spawning takes place primarily in the mainstem between Cloverdale and upstream of Ukiah, but spawning has been observed in Austin, Green Valley, Dry, and Forsythe Creeks (Chase et al. 2007).

Overall: CC coast Chinook salmon are clearly much less abundant in the four regions 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 5,000-20,000 fish annually.

Factors affecting status: The factors affecting CC Chinook salmon fall into five general categories: habitat degradation, estuarine alteration, alteration of flows, urbanization, gravel mining, and alien species. These are also discussed and documented in Moyle (2002).

Logging and road construction: CC Chinook life history requires intact and interacting riparian, freshwater and estuarine ecosystems to support critical growth during the freshwater and estuarine portions of their life cycle. Historic and current land use practices related to logging and 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, Chinook salmon have disappeared from or are imperiled in these watersheds due to alteration of spawning, incubation, and rearing habitats, mainly by 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 floods ripped through the basins. "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 fish was greatly reduced when sections of stream subsequently became too warm and shallow for juveniles during the summer (Moyle 2002, p. 57)."

Recovery after such massive changes would have been difficult in the best of times, but many of the activities that created the problem, especially logging and road building continued with few restrictions. Continued erosion from abandoned logging areas and rural residential roads has 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 entomb embryos. Large-scale sedimentation combined with 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 juveniles. Increased sediment has also been shown to reduce juvenile survival by impacting feeding success through increased turbidity, reducing prey visibility, and irritation of gills. These factors can also create widened, shallow channels, in which temperatures are too high and depths to low to support Chinook salmon juveniles.

Estuarine alteration: Estuaries, bays, and lagoons are increasingly being recognized as critical rearing habitats for salmonids. Numerous lagoons form at the mouth of rivers and creeks in this ESU when summer flows become too low to wash out mouth bars, a factor exacerbated by upstream diversions. These 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 during the end of the spring juvenile outmigration and as well as that of smolts that enter the estuary after it has closed. In addition, once productive estuarine marsh habitats have been drained and diked for pasture, greatly reducing habitat available for rearing of juveniles. Redwood Creek, tributaries to Humboldt Bay, and the Eel River all have lost this estuarine complexity, contributing to the decline of the salmon populations.

Dams: The alteration and withdrawal of water impacts water quality and quantity in rural and urban watersheds inhabited by CC Chinook. The situation in two major rivers in the ESU is double edged, because the main withdrawal of water in the ESU is the interbasin transfer from the upper Eel River into the upper Russian River. This transfer supplies increased flows during fall and spring for vineyard irrigation and municipal uses, which presumably indirectly helps to sustain CC Chinook migration in the mainstem Russian River. The transfer has clearly has contributed to declines in Eel River CC Chinook runs because by reducing flows available for out-migration by juveniles and for upstream spawning migration by adults. The water withdrawals from the Eel River to the Russian River also likely impact water temperature in the upper mainstem Eel by creating thermal barriers earlier in the spring and restricting emigration of juveniles.

Dams on the Mad, Eel and Russian Rivers have also influenced geomorphic regimes and decreased the quality of spawning substrates below them. Ruth Dam is a barrier to Chinook salmon and other anadromous fish 123 km (77 mi) from the ocean on the Mad River and influences flow in this section of the river considerably. It reduces total habitat available for spawning and has altered downstream habitats through reduced flows and gravel recruitment. It is operated in concert with five collector wells in the lower portion of the Mad River operated by the Humboldt Bay Municipal Water District. These wells draw 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 Chinook into the lower portion of the river. A mitigation hatchery was built but Chinook escapement was so low it was abandoned prior to 2000.

Climate Change: Due to CC Chinook's need for small cobble and large gravel for spawning, they most frequently spawn in the main stems of rivers. In the majority of watersheds in this ESU, flows are not controlled by dams and interbasin transfers so natural flows are still the major on influence embryo and juvenile survival. Without sufficient early fall storms, Chinook often will spawn in the lower portion of a river's mainstem and their redds can be lost due to bedload movement if large storms follow the spawning period. Thus, the relationship between flows and spawner timing is critical, and large storms following insufficient rains can lead to significant loss of spawning productivity. This is believed to have occurred in the Mattole River, Freshwater Creek, and Redwood Creek drainages in recent years. In these locations, low counts of outmigrating juveniles despite high spawner abundance estimates have followed dry fall seasons, when flows needed for adults to reach more stable reaches in the drainages were inadequate. Increasing climatic variability may threaten the viability of some coastal populations of CC Chinook when runoff is intensified quickly through area where Chinook may build redds. Logging, urbanization, agriculture, and other factors may also increase the amount and magnitude of run-off from rain storms, increasing their potential for negative effects on Chinook redds and juveniles.

Urbanization and agriculture: Urbanization and agriculture present multiple problems for CC Chinook in many parts of the ESU, especially in the lower portions of watersheds where Chinook are most likely to spawn. Water quality is often degraded by urban pollution and agricultural runoff. The use of land around creeks for towns and farms has led to channelization, construction of revetments, removal of instream habitat, and channel erosion. Increasing urbanization, vineyard planting, and other development through the southern portion of the ESU is straining the capacity for water agencies to meet municipal needs; this is likely to further increase water withdrawals and negatively impact CC Chinook. Likewise, many tributaries are facing increasingly frequent water withdrawals to irrigate vineyards and other crops and to provide frost protection for grape vines.

Gravel mining: Gravel mining continues in the Mad, Eel, Van Duzen, Russian Rivers and Redwood Creek. These operations have been increasingly regulated to minimize impacts in the main stems 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 will not provide necessary flows for incubation, and decrease water quality from pollution and sedimentation. Gravel mining also creates seasonal barriers during critical migratory periods and cause stranding of adult Chinook trying to enter tributaries.

Alien species: Alien fish species, primarily predators, are significant problems mainly in the Eel and Russian River drainages. In the Eel River, Sacramento pikeminnow were introduced illegally in 1979 and they quickly spread throughout the much of the watershed (Brown and Moyle 1997). They are now one of the most abundant fish in the river and it is highly likely that they are suppressing Chinook salmon populations through predation on emigrating juveniles. This effect on Chinook juveniles is likely compounded by stress associated with other factors discussed above. Pikeminnow are native to the Russian River and are not as abundant as they are in the Eel River, but the salmon also face predation from alien predators, such as smallmouth bass (which are abundant). The effect of these predators on Chinook salmon populations in the Russian River is not known, but almost certainly negative.

Hatcheries: The declining state of the ESU has long been recognized by local groups, which operate small scale wild broodstock 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), although these propagation efforts have been curtailed under the Endangered Species Act. In addition, artificial propagation of CC Chinook by CDFG at the Van Arsdale Fisheries Station on the Eel River and at the Mad River Hatchery have been stopped due to the potential negative impacts of these programs on wild fish and, presumably, low returns. It appears that such hatcheries have done little to bolster returns of CC Chinook adults and may increase risks of extirpation in those watersheds where Chinook are being reared and planted (Good et al. 2005; NMFS 2007). While the operation of small scale hatcheries can have beneficial effects if they are in concordance with management of wild spawners, the necessary monitoring for this coordinated effort has not been done, so effects of hatchery fish on wild populations are not well understood.

Conservation: The virtual disappearance of commercial and sport fisheries for Chinook salmon in this ESU and along California's North Coast demonstrates the need for strong conservation measures in CC Chinook salmon, which were first listed as threatened in 1999 (see status, below). A recovery outline was recently released for the CC Chinook ESU assessing the biology, threats, and conservation considerations that will be part of a recovery strategy for the ESU (NMFS 2007). Included in this outline for a recovery plan are an estimated 2630 km (1,634 mi) of stream habitat and 65 square km (25 square mi) of estuarine habitats, which were designated as critical habitat on September 2, 2005. However, the designation has not improved returns of adult CC Chinook. Considerable effort to preserve and restore spawning and rearing habitat have been made over the past two decades but much more needs to be done, especially at a landscape level.

Pressing water quantity and quality issues need to be resolved in most of the ESU's basins to protect and restore habitat required by CC Chinook. Resolution of issues surrounding the balance of water between the Russian and Eel Rivers will greatly influence the persistence of CC Chinook in these basins. While it appears that Chinook are able to exist within the historic and current hydrograph of the Russian River (Chase et al. 2007), 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 viable populations of CC Chinook, but ecological changes in the Eel's mainstem now seem to favor warmer-water species such as the non-native Sacramento pikeminnow. Until water transfers out of the Eel River basin are reduced to provide necessary spring and fall flows for juvenile and adult Chinook, recovery of these multiple populations is unlikely.

Elements of a conservation strategy for CC Chinook salmon should include:

1. Develop a strategic land acquisition program to protect spawning habitats. This should focus holistically on watersheds, and not wetted channels, because sedimentation can only be ameliorated through watershed-wide reduction.
2. Restore estuarine marshes and floodplains and improve lower river riparian corridors to increase juvenile-to-smolt survival. This action is particularly important on the Eel River, Redwood Creek, and other rivers with historically extensive tidal and lagoon habitats.
3. Establish a managed flow regime, similar to the historic hydrograph in volume and timing, for the Eel River below Scott and Van Arsdale dams to provide necessary migration of Chinook into upper portions of spawning habitat and for juveniles to successfully migrate out to sea. The entire operation of the water system that diverts Eel River water into the Russian River (Potter Valley Project) needs to be carefully reevaluated to develop conservation strategies for CC Chinook salmon in both rivers. Recovery alternatives need to consider the overall benefits to the ESU of having multiple viable populations on the Eel River in the center of the ESU and a single viable population in the Russian River at the edge of the ESU.
4. Increase amounts of water allocated from Mendocino and Sonoma reservoirs for fish in the Russian River, in conjunction with reducing flows from the Potter Valley Project.
5. 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. Clear-cutting practices should be stopped and alternate approaches used to adequately address historic and cumulative affects. 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., Navarro).

6. Conduct annual monitoring of spawner abundance and juvenile and smolt abundance for all major, remnant populations within the ESU.
7. Promote municipal, industrial, agricultural outreach programs that conserve water, reduce pollution, and create greater awareness about CC Chinook as an indicator of healthy waters.
8. Evaluate the artificial propagation programs for this ESU to determine their effectiveness and impact on naturally spawning salmon. If their importance in maintaining the populations in this ESU is high, then ways should be found to improve operations until the watersheds can naturally support equivalent numbers of spawners.

Trends:
Short term: The ESU is greatly reduced from historic abundance and is probably still declining (perhaps at a reduced rate) despite some efforts at artificial propagation. However, monitoring is inadequate to determine trends in most ESU rivers especially in the *North Coastal* and *North Mountain Interior* regions. A significant portion of the spawners in the Russian River apparently rely upon a contested interbasin transfer of water from the Eel River, a highly impaired basin with multiple stocks of CC Chinook. The loss of persistent spawning populations along the Mendocino coast represents irreplaceable loss of diversity within the ESU.
Long term: Multiple factors will influence the long-term persistence of CC Chinook, including climate change. Regardless, without major shifts in water allocations to fish in streams within the ESU and without large-scale improvements of logging practices and reductions in harvest rates to reduce erosion and temperature alterations, habitat eventually will not be available for spawning and rearing of CC Chinook. The likely additional negative effects of climate change will be to make major intervention on behalf of CC Chinook necessary to prevent extinction.

Status: 2. Vulnerable to extinction in the next 100 years (or less). The California Coast 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. This ESU has no official status with the California Department of Fish and Game, though it deserves to be officially recognized as Threatened under the California Endangered Species Act by the Fish and Game Commission.

Metric	Score	Justification
Area occupied	3	ESU occupies multiple watersheds.
Effective pop. Size	3	All populations are under 1000 spawners in most years but some mixing among populations
Intervention dependence	2	Severe declines indicate strong intervention needed, especially in Russian and Eel Rivers
Tolerance	2	Resilient life history but warm water puts embryos at risk
Genetic risk	3	Major watersheds may have distinct populations, all threatened by small size and similar genetic issues
Climate change	2	Likely to accelerate declines, especially where flows are reduced and altered channels increase temperatures.
Average	2.5	15/6
Certainty (1-4)	3	NMFS has analyzed much of the existing information in reports.

Table 1. Metrics for determining the status of CC Chinook salmon, where 1 is poor value and 5 is excellent.

CENTRAL VALLEY FALL CHINOOK SALMON
Oncorhynchus tshawytscha

Description: Members of Central Valley fall Chinook salmon Evolutionary Significant Unit (ESU) are morphologically similar to other Chinook salmon (see California coast Chinook salmon for description).

Taxonomic Relationships: The four runs of Chinook salmon in the Central Valley are differentiated by their life history characteristics including maturity of fish entering fresh water, time of spawning migrations, spawning areas, incubation times, and migration timing of juveniles (Moyle 2002). Central Valley fall Chinook mainly migrate upstream in September through November as mature fish, although they have been recorded from June through December. Fall Chinook are part of the Central Valley Chinook genetic complex; all populations within the Central Valley are more closely related to each other than they are to populations outside the valley. Fall Chinook salmon are considered by NMFS to be a distinct ESU that includes late fall Chinook salmon as well, which we regard as distinct. However, all Central Valley fall Chinook salmon (except the late fall Chinook) from throughout their range are genetically extremely similar, an artifact of the constant mixing of hatchery and wild fish and trucking of hatchery juveniles for release into the lower San Francisco Estuary (e.g., Benicia). The movement of juveniles apparently results in many adult salmon with limited imprinting of 'directions' to their natal hatchery rivers and therefore a high degree of straying to non-natal streams.

Life History: Fall Chinook are reasonably well studied because they are the most abundant run in the Central Valley, persisting in large numbers in rivers below dams, and are the principal run raised in hatcheries (Moyle 2002, Williams 2006). They have the classic "ocean type" life history in that adults enter rivers as mature individuals, migrate to spawning grounds and usually spawn in 1-2 months after entry (see Central Valley spring Chinook account for full discussion of life history patterns). Peak spawning time is typically in October-November but can continue through December. Juveniles mostly emerge in December through March and rear in natal streams for 1-7 months, usually moving downstream into the main rivers within a few weeks after emerging. They enter the San Francisco Estuary as both fry and smolts. Despite long-term monitoring, causes of apparent high mortality rates of fish as they pass through the estuary are poorly understood. Two general observations suggest that rearing conditions in the estuary are often poor: survival rates seem to be higher in the rivers than in the estuary and highest survival occurs during wet years, when passage through the estuary is likely to be most rapid (Brandes and McClain 2001; Baker and Mohrhardt 2001). Hatchery fry are mostly trucked to be planted below the Delta, on the assumption that their survival is poor when they pass through it naturally. Flooding in wet years also increases rearing habitat in the Delta and Yolo Bypass, which may have a positive effect.

From the estuary, juvenile salmon move through the Golden Gate into the Gulf of the Farallons, which is a region typically extremely food-rich because of upwelling associated with the California current. Immature fish spend 2-5 years at sea before returning as adults, where they feed on fish and shrimp. Most of the fish remain off the California coast during this period, between Point Sur and Point Arena, but many move into coastal waters of Oregon as well. Their movements in the ocean during the rearing period are poorly known but both inshore-offshore movements and movements along shore are likely through the rearing period, in response to changing temperatures and upwelling strength.

Naturally, there are many exceptions to this general life cycle, including fry that spend as much as one year in fresh water. Overall, this life history strategy reflects adaptations that allow these fish to use the productive lower reaches of Central Valley rivers for spawning and rearing, with the fry moving out as water temperatures become increasingly warm in spring and summer. Historically, it is likely that many of the fry reared for several months (or more) in the somewhat cooler Delta and lower estuary after leaving the river. The present-day lower Sacramento River generally has temperatures suitable for rearing all year around in its upstream reaches, thanks to cool-water releases from reservoirs, although levees and diversions have reduced rearing habitat, especially in dry years.

The attributes of fall Chinook salmon that have made them so well adapted to low elevation rivers have also made them ideal for use in hatcheries, because they can be spawned as they arrive and because the fry only have to be reared for a relatively short time before being released. Other aspects of their life history are similar to other Chinook ESUs which are covered in more detail in the Central Coast Chinook ESU account and also in detail in Moyle (2002) and Williams (2006).

Habitat Requirements: The general habitat requirements are similar to that of other Chinook salmon that minimize their time in fresh water. See the Central Coast Chinook salmon account for details on temperature and other requirements. For a more specific summary of Central Valley Chinook salmon requirements see Stillwater Sciences (2006). The habitat use that may differ most from Chinook salmon elsewhere in California is the use of off-channel habitats by fry, including floodplains, where they grow faster because of warmer temperatures and abundant food (Sommer et al. 2001; Limm and Marchetti 2006; Jeffres et al. 2008). Historically, this habitat was extremely abundant along the valley reaches of the rivers and was probably a major reason for the large numbers of salmon produced by Central Valley rivers. Off-channel habitat (e.g., tidal marshes) may also have been important at one time in the San Francisco Estuary, but it is largely unavailable at the present time.

Distribution: Central Valley fall run Chinook historically spawned in all major rivers of the Central Valley, migrating as far as the Kings River in the south and the Upper Sacramento, McCloud, and Pit Rivers to the north. There were also small, presumably intermittent runs, in smaller streams such as Putah and Cache Creeks. Today they spawn upstream as far as the first impassible dam (e.g., Keswick Dam on the Sacramento River), although on the San Joaquin side of the Central Valley they are only allowed as high up as the Merced River because Friant Dam has cut off all natural flows to the lower San Joaquin River. Further upstream movement today is blocked by the CDFG-operated weir at Hills Ferry. Overall, about 70% of Chinook salmon spawning habitat has been cut off by dams (less for fall run by itself), although cold-water releases from some dams may allow some spawning where it did not formally exist before, such as in lower Putah Creek (Yoshiyama et al. 1998).

Abundance: The historic abundance of fall Chinook is hard to ascertain because they were heavily fished in the 19th century, hydraulic mining debris buried major spawning and rearing areas, and estimates are inaccurate due to poor record keeping. It is likely that they were the most abundant of the four Central Valley runs or tied for that honor with spring Chinook, at about a million spawners per year, plus or minus a couple of hundred thousand fish (Yoshiyama et al. 1998). In the 1960s-90s, average production (the total of in-river escapement plus catch in the fisheries) was about 374,000 fish per year (Figure 1), although the number of spawners usually varied somewhere between 200,000 and 300,000 fish, occasionally dropping to 100,000 or so. In 1992-2005, production averaged about 450,000 fish per year, although it dropped to less than 200,000 fish in 2006 and to about 90,000 spawners in 2007, despite virtual cessation of fisheries. These numbers include fish of both wild and hatchery origin, with hatchery fish making up to 90% of the total, depending on river, year, and who is counting (Barnett-Johnson et al. 2007). Escapements vary tremendously among rivers in the Central Valley as well, with perhaps the greatest variation in the Stanislaus, Tuolumne, and Merced Rivers, tributaries to the lower San Joaquin River (Figure 2). The exact cause of the variation in abundance in these three rivers is not well understood but largest returns follow years with high outflows and high smolt survival.

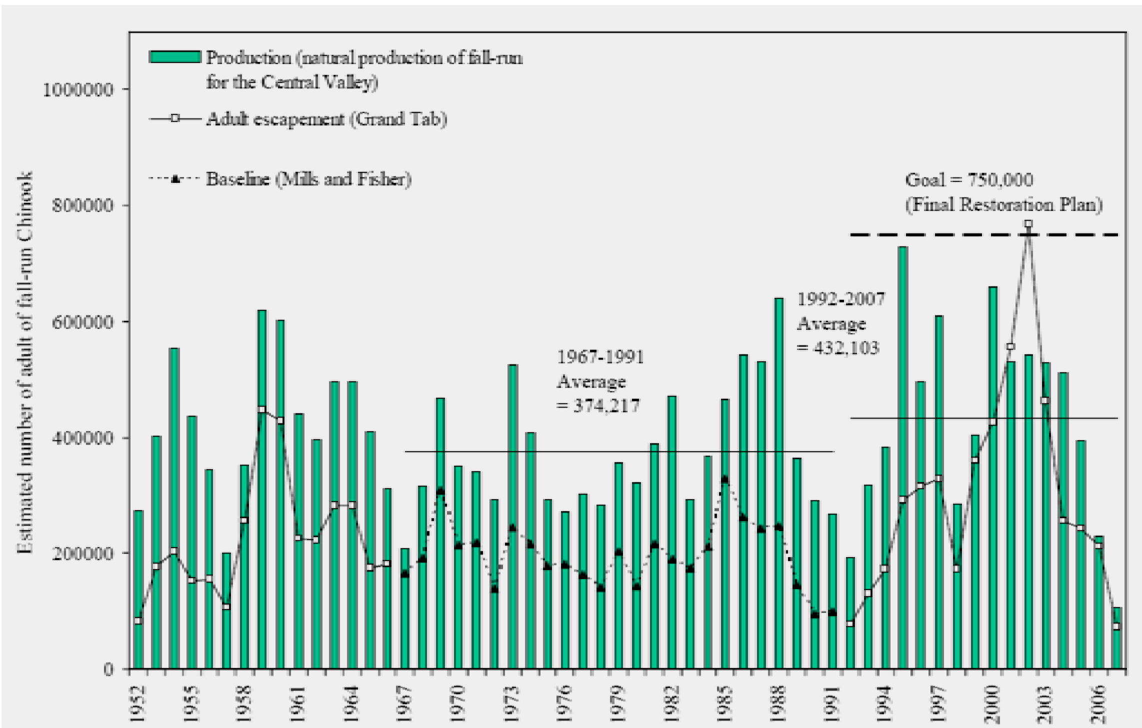
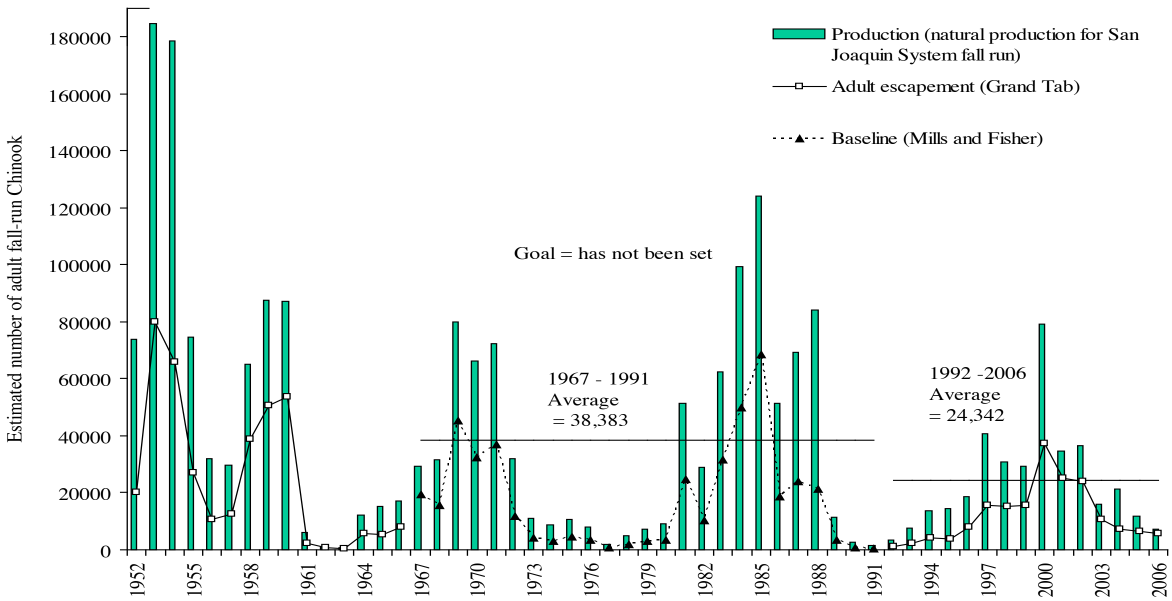


Figure 1. Estimated total production (escapement + catch in fisheries) and escapement of fall Chinook salmon in the Central Valley. Source: http://www.delta.dfg.ca.gov/afrp



Production (natural production for San Joaquin System fall run)
Adult escapement (Grand Tab)
Baseline (Mills and Fisher)
Goal = has not been set
1967 - 1991 Average = 38,383
1992 -2006 Average = 24,342
Estimated number of adult fall-run Chinook
180000
160000
140000
120000
100000
80000
60000
40000
20000
0
1952
1955
1958
1961
1964
1967
1970
1973
1976
1979
1982
1985
1988
1991
1994
1997
2000
2003
2006

diversions, even cumulatively, probably do not kill many salmon, unless they are on small tributaries. In general, the higher percentage of flow taken by a diversion, the more likely the diversion is to have a negative impact on local salmon populations through entrainment of juveniles.

The largest diversions in the Central Valley are those of the State Water Project (SWP) and the federal Central Valley Project in the south Delta. They entrain large numbers of fall Chinook salmon (as well as salmon of other runs) but especially from the San Joaquin River tributaries. The diversions are screened and salmon are 'salvaged' from the projects by capturing, trucking, and then releasing them downstream in the Delta. However, mortality is likely high, both directly and indirectly. Kimmerer (2008) calculated about a maximum 10% loss of juvenile Chinook to direct entrainment, recognizing the high degree of uncertainty associated with any such estimate. Direct mortality is caused by high predation rates in Clifton Court Forebay from which the SWP pumps its water (prior to running it through the salvage facility), by the stress of salvage, and by predation after they are released, disoriented, into predator-rich areas. Indirect mortality is likely considerably higher than direct mortality and is caused by changes in Delta hydrology due to project operations, created by both the pumping itself and by the dam releases (or lack thereof) to provide water for the water project pumps. The salmon essentially can be diverted into unfavorable parts of the Delta in which they are much more likely to die of environmental stress or predation. In general, when flows are higher and salmon avoid the pumps, survival of outmigrants tends to be higher, although there is no simple relationship between the amount of water being diverted *per se* and salmon survival (Brandes and McClain 2001). However, San Joaquin fall Chinook salmon are likely affected by South Delta Pumping, especially when their populations are at low ebb, because they are most vulnerable to the pumps from sheer proximity.

Habitat loss: Loss of adult habitat has been discussed under dams, but loss of juvenile habitat in the rivers is equally a problem, especially the shallow riverine and estuarine habitats needed for feeding and protection from predators during migration. Construction of levees to contain rivers has had multiple effects, including simplifying bank structure through use of rip-rap and removal of trees, reduction in shade, and reduced access to floodplains. This whole process of bank hardening has been made much easier by the reduction of peak flows by dams. Today, restoration of floodplain habitat is regarded as especially important for juvenile salmon growth and survival (Sommer et al. 2001, Jeffres et al. 2008). Loss of shallow water habitat in the San Francisco Estuary may also have had a negative impact on juvenile Chinook salmon although restoration of this habitat is problematic in its positive effects because of the presence of so many alien predators and competitors in the habitat, especially in fresh water.

Fisheries: The effects of harvest on Central Valley salmon in general is discussed at length by Williams (2006). Chinook salmon are harvested in both ocean and in-river fisheries. Hatchery fish can sustain higher harvest rates than wild fish, but fisheries do not discriminate between them. The fisheries are presumably taking wild and hatchery fish in proportion to their abundance and a harvest rate that is sustainable for hatchery fish may be unsustainable for wild fish. This can lead to hatchery fish *replacing* wild fish in the fishery rather than just supplementing them (as they were supposed to do).

Commercial fisheries also may have affected Chinook populations indirectly through continual removal of larger and older individuals. This selectivity results in spawning runs made up mainly of three-year-old fish, which are smaller and therefore produce fewer eggs per female. The removal of older fish also removes much of the natural "cushion" salmon populations have against natural disasters, such as severe drought, which may wipe out a run in one year. Under natural conditions, the four- and five-year-old fish still in the ocean help to keep the runs balanced and can make up for the fish lost during an occasional catastrophe. In order to protect declining stocks of Chinook salmon, marine salmon fisheries were greatly restricted in 2006 and 2007 by the National Marine Fisheries Service and the Pacific Fisheries Management Council (Congressional Record, 50 CFR Part 660); they were banned completely in 2008. This has resulted in the return of a higher proportion of larger and older fish than in previous years, although numbers of fish were nevertheless exceptionally low.

Hatcheries: After an exhaustive review of the literature on hatchery practices in California, Williams (2006) concluded that hatcheries almost certainly have deleterious effect on wild populations of salmon, which may run contrary to recovery goals for wild fish. The effects can stem from competition by hatchery fish with wild juveniles when hatcheries flood the environment with juveniles that are bigger and more numerous than wild fish. This behavior can displace wild fish, making them more vulnerable to predation and reducing growth rates. Such competition potentially can exist at all phases of the life cycle, including during ocean feeding and on the spawning grounds. As indicted above, the presence of large numbers of hatchery fish also resulted in unsustainable harvest rates of wild Chinook salmon, further reducing the viability of wild populations.

In addition, studies on other salmonids, especially steelhead, have shown that fitness (ability to produce young that survive to reproduce) decreases rapidly when fish are raised in hatcheries. Araki et al. (2007) estimate that fitness of steelhead decreases almost 40% per generation of hatchery culture. The loss of fitness may be less severe for Chinook, but it is almost certainly serious, given that the fish spawning in Central Valley rivers are increasingly of hatchery origin. This may also result in fish that are less well adapted to persisting through adverse conditions in both fresh and salt water (e.g., physiologically less capable of surviving on less food, less sensitive to changing ocean conditions, less able to avoid predation). It is also possible that the fairly uniform nature of Central Valley fall Chinook has reduced variability in response to environmental conditions, making them more vulnerable to mortality under variable ocean conditions.

Pollution: Juvenile salmon are continuously exposed to toxic materials discharged in to rivers from both urban and agricultural sources. The latter are particularly likely to affect juvenile San Joaquin fall Chinook salmon and a potentially major source of mortality is the toxic, anoxic water associated with the Stockton Deepwater Ship Channel which results from pollutants from agricultural wastewater, discharges from the Stockton sewage treatment and storm drains, and other sources. A new threat is the use of pyrethroid pesticides which are particularly toxic to fish. The effects of these diverse pollutants on wild juvenile salmon abundance is largely unknown but mortality is periodically recorded. In the Sacramento River, a potential major problem is water laden with toxic heavy metals from the Iron Mountain mine site, if the Spring Creek retention reservoir spills or bursts. These highly toxic wastes could wipe out either migrating adults or, more likely, juveniles foraging in the river. Even if pollutants are not directly lethal, they (or poor water quality in general) can stress both adult and juvenile fish so the fish become more susceptible to diseases that are always present in the environment.

Alien species: For the past 150 years, Chinook salmon have been faced with an onslaught of potentially deleterious alien species yet have managed to persist despite them. Probably most significant are fish that are predators include striped bass, largemouth bass, smallmouth bass, and spotted bass. Striped bass have not been implicated directly in any salmon declines, perhaps because they arrived early enough on the scene so they mainly replaced native predators. They can consume large numbers of juvenile salmon, however, below diversions such as Red Bluff Diversion Dam, or where hatcheries release large numbers of naïve fish. The three centrarchid (black) basses can also be locally important as predators, especially when they inhabit in-channel gravel pits and other obstacles the juvenile salmon have to pass through on their way downstream. Fortunately, their metabolic processes are relatively slow, due to low temperatures, when peak juvenile salmon out-migrations occur, which reduces predation. One of the reasons CDFG made such a huge effort to eradicate northern pike (*Esox lucius*) from Davis Reservoir on a tributary to the Feather River is that pike are cool-water predators, so are likely to be much more effective predators on juvenile salmon than existing alien predators. Their potential invasion of the Sacramento River system could be disastrous for salmon runs.

Climate change: Naturally spawning fall Chinook are largely dependent on fall releases from dams to stimulate migration of adults and juveniles. Under most climate change scenarios, much of the Sierra Nevada snow pack will be lost, so precipitation will fall largely as rain, running off quickly. This means less water will be stored in the system and potentially less water will be available for salmon downstream of dams, especially in spring and summer. What water is available is also likely to be warmer, perhaps even stressful to salmon by late spring. For fall Chinook salmon, this means adults may have to ascend streams later in the season and juveniles leave earlier, narrowing the window of time for successful spawning of wild fish. Williams (2006) regards climate change as one of the biggest future threats to the ability of wild salmon in the Central Valley to sustain populations and provides more detailed discussion on the subject.

Ocean conditions: One of the least understood effects of climate change is the impact on ocean conditions. However, the implications of melting polar icecaps and glaciers, as well as changes in wind patterns, ocean currents, and upwellings, indicate major impacts on salmon populations. It is already obvious that existing natural variations in ocean currents and temperatures, related to ENSO (El Nino-Southern Oscillation) and PDO (Pacific Decadal Oscillation), have dramatic effects on salmon populations, although Central Valley Chinook populations do not show much response to the PDO, compared to more northern populations. Central Valley Chinook salmon tend to stay close to the California, Oregon, and Washington coasts, therefore, when these effects cause a decrease in upwelling along the California coast, local ocean productivity declines and Chinook salmon starve (or at least have lower growth and survival rates). However, probably the single most important region for their survival is the Gulf of the Farrallons, which is not only the first place juvenile Central Valley Chinook go but it is also one of the most productive areas in the region, at least during high upwelling years. In recent years (2005-2008), short-term anomalies in ocean conditions, resulting in decreased upwelling during critical times of year, may have been responsible for low ocean survival of Central Valley chinook salmon (Barth et al. 2007). See Williams (2006) and following discussion for more details on how ocean conditions impact California Chinook salmon.

Factors affecting status-an integrated view: Ever since the Gold Rush, Central Valley Chinook salmon populations have been in decline. Historic populations probably averaged 1.5-2.0 million (or more) adult fish per year (Yoshiyama et al. 1998). The high numbers resulted from four distinct runs of Chinook salmon (fall, late-fall, winter, and spring runs) taking advantage of the diverse and productive freshwater habitats available, created by the cold rivers flowing from the Sierra Nevada. When the juveniles moved seaward, they found abundant food and good growing conditions in the wide valley floodplains and complex San Francisco Estuary, including the Delta. The salmon smolts then reached the ocean, where the southward flowing, cold, California Current and coastal upwelling together created one of the richest marine ecosystems in the world, full of the small shrimp and fish that salmon require to grow rapidly to large size. In the past, salmon populations no doubt varied as droughts reduced stream habitats and as the ocean varied in its productivity, but it is highly unlikely the numbers ever even approached the low numbers we are seeing now.

Unregulated fisheries, hydraulic mining, logging, levees, dams, and other factors discussed above caused precipitous population declines in the 19th century, to the point where the salmon canneries were forced to shut down (all were gone by 1919). Minimal regulation of fisheries and the end of hydraulic mining allowed some recovery to occur in the early 20th century but the numbers of harvested salmon steadily declined through the 1930s. There was a brief resurgence in the 1940s but then the effects of the large rim dams on major tributaries began to be severely felt (Yoshiyama et al. 1998). The dams cut off access to 70% or more of historic spawning areas and basically drove the spring and winter runs to near-extinction, making the fall run the principal support of fisheries. In the late 20th century, thanks to hatcheries, special flow releases from dams, and other improvements, salmon numbers (mainly fall-run Chinook) averaged nearly 500,000 fish per year, with wide fluctuations from year to year, around 10-25% of historic abundance (Figure 1). In 2006, numbers of spawners dropped to about 200,000, despite closure of the fishery (to protect Klamath River Chinook runs). In 2007, the number of spawners fell further to about 90,000 fish, among the lowest numbers experienced in the past 60 years, with expectations of even lower numbers in fall 2008 (probably <64,000 fish). The evidence suggests that these runs are largely supported by hatchery production, so numbers of fish from natural spawning are much lower.

So, what caused this apparently precipitous decline in salmon? Unfortunately, the causes are historic, multiple and interacting. The first thing to recognize is that Chinook salmon are adapted to living in a region where conditions in both fresh water and salt water can alternate between being highly favorable for growth and survival and being comparatively unfavorable. Usually, conditions in both environments are not overwhelmingly bad together, so when survival of juveniles in fresh water is low, those that make it to salt water do exceptionally well, and vice versa. This ability of the two environments to compensate for one another's failings, combined with the ability of adult salmon to swim long distances to find suitable ocean habitat, historically meant salmon populations fluctuated around some high number. Unfortunately, when conditions are bad in both environments, populations crash, especially when the heavy hand of humans is involved.

The recent precipitous decline has been blamed largely on "ocean conditions." Generally what this means is that the upwelling of cold, nutrient-rich water has slowed or ceased, so less food is available, causing the salmon to starve or move away. Upwelling is the result of strong steady alongshore winds which cause surface waters to move off shore, allowing cold, nutrient-rich, deep waters to rise to the surface. The winds rise and fall in response to movements of the Jet Stream and other factors, with both seasonal and longer-term variation. El Nino events can affect local productivity as well, as can other 'anomalies' in weather patterns. And Chinook salmon populations fluctuate accordingly.

The 2006 and 2007 year classes of returning salmon mostly entered the ocean in the spring of 2004 and 2005, respectively (most spawn at age 3). Although upwelling should have been steady in this period, conditions unexpectedly changed and ocean upwelling declined in the spring months, so there were fewer shrimp and small fish for salmon to feed on. According to an analysis by Barth et al. (2007), conditions were particularly bad for a few weeks in spring of 2005 in the ocean off Central California, resulting in abnormally warm water and low concentrations of zooplankton, which form the basis for the food webs which include salmon. All this *could* have caused wide scale starvation of the salmon. While the negative impact of ocean anomalies on salmon is likely, monitoring programs in ocean are too limited to make direct links between salmon and local ocean conditions.

"Ocean conditions" can also refer to other factors which can be directly affected by human actions, especially fisheries. For example, fisheries for rockfish and anchovies can directly or indirectly affect salmon food supplies (salmon eat small fish). Likewise, fisheries for sharks and large predators may have allowed Humboldt squid (which grow to 1-2m long) to become extremely abundant and move north into cool water, where they *could* conceivably prey on salmon. These kinds of effects, however, are largely unstudied.

Meanwhile, what has been going on in the Sacramento and San Joaquin Rivers? On the plus side, dozens of stream and flow improvement projects have increased habitat for spawning and rearing salmon. Removal of small dams on Butte Creek and Clear Creek, for example, increased upstream run sizes dramatically. Salmon hatcheries also continue to produce millions of fry and smolts to go to the ocean. On the contrary side:

- The giant pumps in the South Delta have diverted increasingly large amounts of water in the past decades, altering hydraulic and temperature patterns in the Delta as well as capturing fish directly.
- The Delta continues to be an unfavorable habitat for salmon, especially on the San Joaquin side where the inflowing river water is warm and polluted with salt and toxic materials.
- Hatchery fry and smolts are released in large numbers but their survivorship is poor, compared to wild fish, although they contribute significantly to the fishery. Nevertheless, they may be competitors with wild produced fish under conditions of low supply in the ocean. Most of the hatchery fish are planted below the Delta, to avoid the heavy mortality there. Unfortunately, the fitness of naturally produced salmon versus hatchery produced salmon is not understood; it is possible that the influence of hatchery-reared fish is so strong today that the progeny of natural and hatchery spawners have similar survival rates in the wild.
- Numbers of salmon produced by tributaries to the San Joaquin River (Merced, Tuolumne, Stanislaus) continue to be exceptionally low, in the hundreds, and the promised restoration of the San Joaquin River will take a long time to be effective.

Thus reduced survival of naturally spawned fish in fresh water, especially in the Delta, combined with the naturally low survival rates of hatchery fish, could make for plummeting numbers of adult spawners. This is especially likely to happen if young salmon also hit adverse conditions in the ocean, as they enter the Gulf of the Farrallons. The growing salmon can also hit other periods when food is scarce in the ocean, along with abundant predators and stressful temperatures, at any time in the ocean phase of their life cycle. Once again, our ignorance of how the salmon survive in the ocean is profound. For example, much could be learned about how ocean food supplies are affecting salmon growth and survival by tracking the growth and condition of juveniles once they have moved out to sea.

The overall message here is that indeed "ocean conditions" have had a lot to do with the recent steep decline of salmon populations in the Central Valley in recent years. However, they are superimposed on a population that has been declining in the long run (with some apparent stabilization in recent decades, presumably due to hatchery production). The salmon still face severe problems before they reach the ocean, especially in the Delta. Overall, blaming "ocean conditions" for salmon declines is a lot like blaming Hurricane Katrina for flooding New Orleans, while ignoring the many human errors that made the disaster inevitable, such as poor construction of levees or destruction of protective salt marshes. Managers have optimistically thought that salmon populations were well managed, needing only occasional policy modifications such as hatcheries or removal of small dams, to continue to go upward. The listings of the winter and spring runs of Central Valley Chinook as endangered species were warnings of likely declines on an even larger scale.

On a final somewhat more optimistic note, there is a reasonably good chance that Chinook salmon populations will once again return to higher levels as they have in the past. However, the lower the population goes and the more the environment changes in unfavorable ways, the more difficult recovery becomes.

Conservation: Before Central Valley winter and spring Chinook salmon were listed, virtually all salmon conservation actions were focused on fall Chinook, because it was the abundant run that supported fisheries. Prior to the passage of the Central Valley Project Improvement Act (CVPIA) by Congress in 1992, which established the Anadromous Fish Restoration Program (AFRP), actions to protect fall run salmon were either focused on improving hatchery production or initiating defensive actions to prevent further declines. Thus minimum flow releases were established as dams were relicensed, the largest diversions were screened, efforts were made to rescue salmon entrained at the large pumping plants in the South Delta, barriers to passage were removed in some streams, and minimal monitoring continued. The AFRP and its associated agencies began to take additional actions to enhance wild salmon populations, including limiting the ocean fishery, improving management of diversions (such as Red Bluff Diversion Dam), investigating ways to improve passage through the Delta, and other measures. The AFRP has pledged to use "all reasonable efforts to at least double natural production of anadromous fish in California's Central Valley streams on a long-term, sustainable basis" http://www.delta.dfg.ca.gov/afrp). The final goal is to average 990,000 fish for all four runs combined, but predominately fall Chinook.

The listing of winter Chinook as threatened in 1990 (endangered in 1994) and spring Chinook salmon as threatened in 1998, increased the urgency of salmon restoration efforts, and actions to benefit these two runs have benefited fall Chinook as well, at least in the Sacramento River. Funding for much of the recent restoration efforts, especially the more innovative projects (such as rehabilitating Clear Creek and Battle Creek), has largely come through CALFED, established in 1994, which coordinates the actions of 25 state and federal agencies. The increase in fall Chinook numbers up to 2005 (Figure 1) was attributed in part to CALFED actions in the Sacramento River drainage, although generally favorable water years (no major drought) and good ocean conditions may have been more important overall, as the rapid decline in populations in 2006-2008 suggests.

In the San Joaquin tributaries, considerable effort has been made to improve conditions for fall Chinook salmon, including flow regimes, better habitat management, reducing impacts of instream gravel pits, and other actions. However, these actions and the presence of a hatchery on the Merced River have still not prevented recent declines in fall Chinook numbers (Figure 2), presumably as the result of factors outside the San Joaquin basin, especially in the southern Delta. Better management of New Melones reservoir for increasing San Joaquin fall Chinook smolt survival could include increasing releases during wet years to better mimic natural spring releases; this would also benefit downstream water quality needs.

One step towards improving management of Central Valley Chinook salmon stocks in general is a better marking program for hatchery fish. While improvements in constant fraction marking programs at Central Valley hatcheries have been made, commercial and sport fishing management for Central Valley fall run Chinook should move towards a fishery in which only marked (hatchery) fish can be kept. A mark-selective fishery could provide a flexible and cost-effective management tool for guaranteeing sport and commercial fisheries for fall Chinook in the face of increased regulation for mixed ocean stocks and will accelerate recovery of ESA-listed Central Valley Chinook stocks. However, high mortality rates of released fish in the ocean, due to stress and marine mammal predation, may make a mark-selective commercial fishery problematical.

Overall, in the short run, there are only a few 'levers' we can pull to improve conditions for salmon which include shutting down the commercial and recreational fisheries, reducing the impact of the big pumps in the South Delta, and perhaps changing the operation of dams (e.g., increasing outflows at critical times), regulating hatchery output, and reducing other ocean fisheries. In the longer run (10-20 years) we need to be engaged in improving the Delta and the rest of the San Francisco Estuary as habitat for salmon, reducing inputs to the estuary of toxic materials, continuing with improvements of upstream habitats, managing floodplain areas such as the Yolo Bypass for salmon, restoring the San Joaquin River, and generally addressing the multiplicity of factors that affect salmon populations. There is also a huge need to improve monitoring of salmon in the ocean as well as the coastal ocean ecosystem itself off California. Right now, our understanding of how ocean conditions affect salmon is largely educated guesswork with guesses made long (sometimes years) after an event affecting the fish has happened. An investment in better knowledge should have large pay-offs for better salmon management.

Trends:

Short term: For about 10 years (1994-2004), fall Chinook salmon numbers fluctuated widely but overall appeared to be about 20% higher on average than numbers in the 1960s - 1980s, apparently in response to conservation actions (but see above). It is clear that ocean conditions and factors outside the watershed can impact survival, however, resulting in lower than expected returns in 2005-2008.

Long term: Following the changes caused to rivers by the Gold Rush and overexploitation by early fisheries, fall Chinook salmon numbers declined to perhaps 10% of their original numbers. Following the construction of large dams in the 1940s-60s, numbers would have declined even further if hatcheries had not been built for mitigation, more or less as an afterthought, to mollify concerns of commercial fishermen. The hatcheries maintained populations at around 375,000 fish (escapement + catch) but the impact of hatchery fish on wild salmon populations presumably replaced further natural reproduction, resulting in the genetically uniform population of fall Chinook salmon than now exists. Recent conservation efforts appear to have boosted salmon production in recent years but these improvements may be reversed by the effects of climate change on both rivers and ocean. On the San Joaquin side, there is now a court order to restore a self-sustaining population of fall run Chinook salmon to the river below Friant Dam, which, if successful, will help maintain salmon numbers overall.

Status: 4. No immediate extinction risk, but the reliance of this ESU on hatchery production and the recent severe decline of the population suggests that more effort needs to be made to maintain self-sustaining wild populations, particularly if we want to maintain commercial fisheries. The CV fall Chinook is listed as a species of special concern by NMFS. A status review by NMFS concluded for fall Chinook 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)". According to Williams (2006, p 304) "This uncertainty remains." However, by the criteria of Lindley et al. (2007) fall Chinook could be listed as threatened because the heavy hatchery influence is associated with a decline of wild populations. There is the distinct possibility that this run could be reduced in the future, even with hatchery production, to such a small size that it could no longer support a commercial fishery of any size.

Metric	Score	Justification
Area occupied	2	Multiple apparent populations in the Central Valley although only one population genetically
Effective population size	5	This is the most abundant salmon stock in California
Dependence on intervention	4	Presumably this ESU would persist even without much human intervention, albeit in small numbers. Major intervention is required to maintain fisheries.
Tolerance	3	Moderate physiological tolerance, multiple age classes
Genetic risk	5	One genetically diverse population
Climate change	3	Climate change can reduce abundance and survival but their 'ocean' life history strategy makes them the least vulnerable of all runs to extirpation, but not severe population decline.
Average	3.7	22/6
Certainty	4	Well studied although high uncertainty about ocean stage

Table 1. Metrics for determining the status of Central Valley fall Chinook salmon, where 1 is poor value and 5 is excellent.

CENTRAL VALLEY LATE FALL CHINOOK SALMON
Oncorhynchus tshawytscha

Description: Central Valley late fall Chinook salmon are morphologically similar to other Chinook salmon (see California Coast Chinook ESU). They tend to be larger than other Central Valley Chinook salmon, reaching 75-100 cm TL and weighing up to 9-10 kg or more.

Taxonomic Relationships: The four runs of Chinook salmon in the Central Valley are differentiated by their life history characteristics including maturity of fish entering fresh water, time of spawning migrations, spawning areas, incubation times, and migration timing of juveniles (Moyle 2002). The late fall run population is part of the Central Valley Chinook genetic complex; all populations within the Central Valley are more closely related to each other than they are to populations outside the valley. Late fall Chinook, however, were only fully recognized as a distinct run in 1966 after the construction of Red Bluff Diversion Dam (see abundance section). Using modern genetic techniques, late fall Chinook can be distinguished from the other runs (Williams 2006) although NMFS manages them as part of the Central Valley fall run ESU because of their close relationship to it. We follow Yoshiyama et al. (1998), Moyle (2002), Williams (2006) and others in recognizing it as a genetically-distinct life history type within the Central Valley Chinook salmon complex.

Life History: The basic life history of late fall Chinook is similar to that of other Chinook salmon runs (see Central Coast Chinook salmon account, Moyle 2002, Williams 2006), although it is much less well known in its details because of its comparatively recent recognition and its tendency to ascend and spawn at times when the Sacramento River is most likely to be high, cold, and turbid, making the fish hard to study. In the past, these migrating fish were a mixture of age classes, ranging from two to five years old. At the present time a majority of the fish are probably three-year olds. Late fall Chinook mostly migrate upstream in December and January as mature fish, although they have been recorded from November through April (Williams 2006). Spawning occurs mainly in late December and January, shortly after the fish arrive on the spawning grounds, although it may extend into April in some years (Williams 2006). Emergence from the gravel starts in April and all fry have usually emerged by early June. The juveniles may hold in the river for 7-13 months before moving out to sea. Peak migration of smolts appears to be in October. However, there is evidence that many migrate out at younger ages and smaller sizes. Williams (2006) indicates that if DFG size criteria are used, downstream migrating late fall Chinook can be found in most months of the year.

Habitat Requirements: The specific habitat requirements of late fall Chinook have not been determined, but they are presumably similar to other Chinook salmon runs and optimal conditions fall within the range of physical and chemical characteristics of the unimpaired Sacramento River above Shasta Dam. See Central Coast Chinook salmon account for details on temperature and other requirements. For a more specific summary of Central Valley Chinook salmon requirements see Stillwater Sciences (2006).

Distribution: Currently, Central Valley late fall Chinook are found mainly in the Sacramento River, where most spawning and rearing of juveniles takes place in the reach between Red Bluff Diversion Dam (RBDD) and Redding (Keswick Dam). However, varying percentages of the total run spawn downstream of RBDD in some years. In 2003, for example 3% of the fish spawned below the dam, while in 2004 no fish spawned below the dam (Kano 2006a, b). R. Painter (DFG, pers. comm., 1995) indicated that apparent late fall Chinook have been observed spawning in Battle Creek, Cottonwood Creek, Clear Creek, Mill Creek, Yuba River and Feather River, but these are presumably at best a small fraction of the total population. The Battle Creek spawners are likely derived from fish that originated from the Coleman National Fish Hatchery. The historic distribution of late fall Chinook is not well documented, but they most likely spawned mostly in the upper Sacramento and McCloud rivers in reaches now blocked by Shasta Dam, as well as in sections of major tributaries where there was adequate cold water in summer. There is also some evidence they once spawned in the San Joaquin River in the Friant region and in other large San Joaquin tributaries (Yoshiyama et al. 1998).

Abundance: The historic abundance of late fall Chinook is not known because it was recognized as distinct from fall Chinook only after Red Bluff Diversion Dam was constructed in 1966. In order to get past the dam, salmon migrating up the Sacramento River had to ascend a fish ladder in which they could be counted with some accuracy for the first time. The four Chinook salmon runs present in the river (fall, late fall, winter, spring) were revealed as peaks in the counts, although salmon passed over the dam during every month of the year. In the first 10 years of counting (1967-1976) the run averaged about 22,000 fish; in the next 10 years (1982-1991) the run averaged about 9,700 fish (Yoshiyama et al. 1998). Since 1991, estimates of abundance are less accurate but in 1992-2007, total numbers were estimated to have averaged 20,777 fish, with a wide range in annual numbers, including a 1998 production total of over 80,000 fish. The less accurate counts were the result of opening the gates at Red Bluff for free passage of the listed winter Chinook salmon from September 15 to May 15 starting in 1992. This made estimation of late fall Chinook spawner numbers more difficult because most of the fish could not be counted while ascending the fish ladders as they had been previously. In 1992-1996, estimates were made by extrapolating from counts of only part of the run. These numbers are extremely low and unreliable (Figure 1). In 1998, DFG initiated surveys based on carcass and redd counts from airplanes and estimated that over 35,000 late fall Chinook had spawned above Red Bluff Diversion Dam. Subsequent surveys have resulted in lower estimates (e.g. 5,000 in 2003) but with variability from year to year. The numbers seem to indicate that measures taken to benefit winter Chinook salmon have probably also benefited late fall run. It is possible that fish from Coleman National Fish Hatchery on Battle Creek are contributing to the spawning population in the main stem Sacramento River (Figure 2).

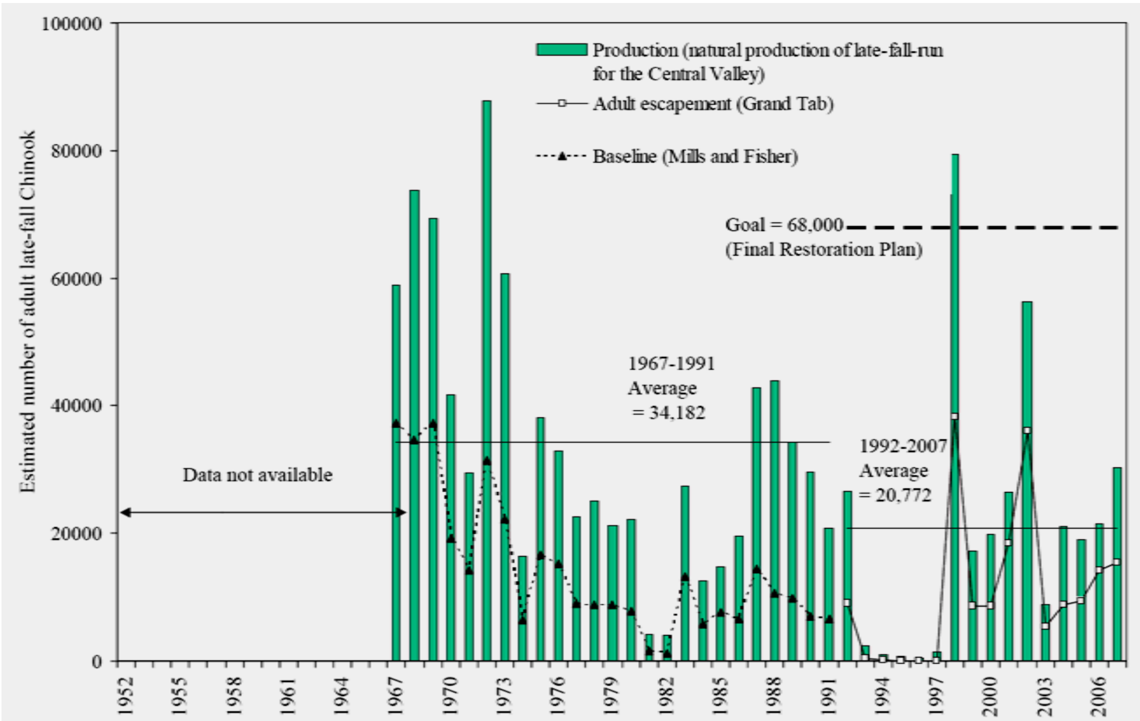
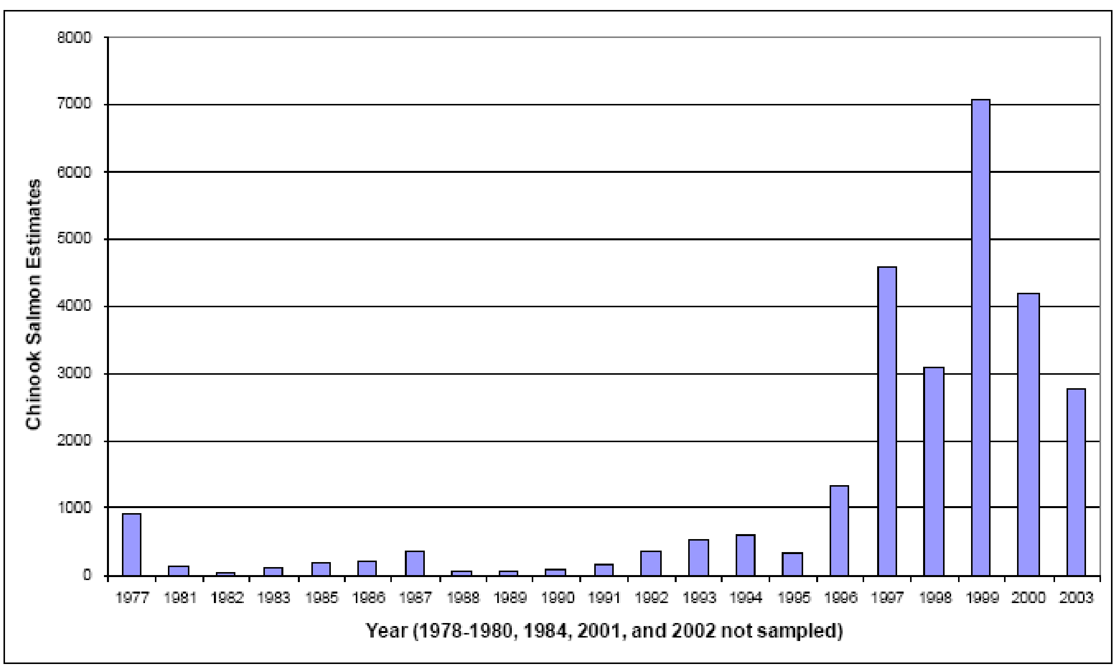


Figure 1.Estimates of late fall run Chinook salmon spawners 1967-2007, between Red Bluff Diversion Dam and Keswick Dam. http://www.delta.dfg.ca.gov/afrp/documents/Doubling_goal_graphs_031308.pdf



Note: GrandTab, DFG, Red Bluff Office, contact Colleen Harvey Arrison, 2004

Figure 2. Numbers of late fall run Chinook salmon in Battle Creek, where the Coleman Fish Hatchery is located. From DWR 2005.

Factors affecting status: For late fall Chinook salmon, the causes of its population decline from pre-dam numbers are poorly documented, but likely are similar to those of the other three runs, in whose accounts more general factors affecting status are discussed. Some of principal factors more specifically affecting late fall Chinook salmon status, past and present, seem to be (1) dams, (2) loss of habitat, (3) fisheries, (4) outmigrant mortality, and (5) hatcheries.

Dams: When Shasta and Keswick Dams were built in the 1940s, they denied late fall Chinook access to upstream spawning areas where spring water originating from Mt. Shasta, as well as extended snow-melt, kept water temperatures cool enough for successful spawning, egg incubation and survival of juvenile salmon all year around. The effects of RBDD were more subtle and not recognized until fairly recently. This dam apparently delayed passage to upstream spawning areas and also concentrated predators, increasing mortality on out-migrating smolts. Kope and Botsford (1990) documented that the overall decline of Sacramento River salmon was closely tied to the construction of RBDD. Raising the dam's gates for much of the year to allow salmon passage has apparently alleviated much of this problem.

Habitat loss or deterioration: Large dams on the Sacramento River and its tributaries have not only denied salmon access to historic spawning grounds, but they have reduced or eliminated recruitment of spawning gravels into the river beds below the dams and altered temperature regimes. Loss of spawning gravels in the Sacramento River below Keswick Dam is regarded as a serious problem and large quantities of gravel are now trucked to the river and dumped in, mainly to provide spawning sites for winter Chinook. However, it is likely that late fall salmon also use these gravel deposits. Warm water temperatures are potentially a problem in this reach, during drought years when the cold water pool in Shasta Reservoir is reduced. The installation of means to provide cooler water in summer for winter Chinook has presumably also benefited late fall Chinook.

Outmigrant mortality: Outmigrant mortality of both fry and smolts is undoubtedly a factor affecting late fall Chinook abundance as it is for all runs of salmon in the Sacramento-San Joaquin drainage. Small numbers of outmigrants are presumably entrained at the larger irrigation diversions along the Sacramento River that are operating during the migration period. At the same time, extensive bank alteration, especially rip-rapping, had reduced the amount of cover available to protect outmigrants from striped bass, terns, herons, and other predators. Once the fish reach the Delta region, there is a complex series of factors that affect their survival (Brandes and McClain 2001). Basically when outflows are high enough so pumping at the SWP and CVP pumping plants does not affect seaward movement, survival is high. At lower river flows and higher exports, juvenile Chinook can be entrained in large numbers, are consumed by predators in Clifton Court Forebay and other off-channel areas, and or are otherwise diverted from their downstream migration into unfavorable habitat. Regulations are in place to protect outmigrating salmon from diversions but their effectiveness varies.

Fisheries: The effects of harvest on Central Valley salmon in general is discussed at length by Williams (2006). The actual harvest rates of late fall Chinook are not known, but it is highly likely that they are harvested at the same rates as fall Chinook, the principal remaining run in the Sacramento River. Although hatcheries exist to sustain fisheries and hatchery fish can sustain higher harvest rates than wild fish, fisheries do not discriminate between them. The fisheries are presumably therefore taking a disproportionate number of wild late fall Chinook. Other effects are discussed in the fall run Chinook account.

Hatcheries: Late fall Chinook are reared in large numbers (ca. 1 million smolts released each year) in Coleman National Fish Hatchery on Battle Creek. This has been taking place since the 1950s, even though the run was not formally recognized until 1973 (Williams 2006). Hatchery brood stock selection for late fall Chinook includes both fish naturally returning to Battle Creek and those trapped below Keswick Dam. The production goal is 100 million smolts per year, which are released into Battle Creek in November through January (Williams 2006). Large numbers are needed because survival rates are low (0.78% at Coleman). Williams (2006) after an exhaustive review of the literature and hatchery practices in California concludes that hatcheries almost certainly have deleterious effects on wild populations of salmon, a finding which may make it more difficult to achieve recovery goals for naturally-spawning late fall Chinook salmon.

Conservation: At present, less management is done to benefit directly late fall Chinook salmon than for any other run in the Sacramento River, mostly because the least is known about it and because it is considered a segment of the fall Chinook population by NMFS. Fortunately, this run should benefit considerably from measures being made to enhance winter and fall Chinook populations in the river. However, studies should be undertaken to better understand the environmental requirements of this run because the population needs protection at all stages of its life cycle. The Anadromous Fish Restoration Program (AFRP) 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 fish (http://www.delta.dfg.ca.gov/afrp/). Whether or not existing habitat is enough to sustain populations at either level is problematic.

Restoration will require: (1) continuing to provide passage of adults to holding and spawning areas, through Red Bluff Diversion Dam, (2) protecting adults in spawning areas, (3) establishing additional spawning areas (e.g., Battle Creek, San Joaquin River), (4) providing passage flows for out-migrating juveniles to get through the Delta as rapidly as possible, (5) maintaining and expanding rearing habitat for juvenile fish, including the mainstem and floodplains, (6) regulating the fisheries to minimize impact, and (7) reducing the effects of hatchery fish on wild populations. Most of these require continuous, creative management, as well as greatly improved monitoring programs for both hatchery and wild fish (Williams 2006).

An aspect of their conservation that needs to be carefully evaluated is the practice of rearing large numbers in Coleman Hatchery, because they appear to an increasingly large proportion of the total population (Williams 2006). While the hatchery fish serve as a back up population for fish in the river, they can also have an adverse effect on wild populations.

Trends:

Short term: In the past 10 years, numbers have fluctuated but appear to be comparable to numbers in the 1970s and 1980s. According to NMFS, late fall Chinook "continue to have low, but perhaps stable, numbers."
(http://www.nmfs.noaa.gov/pr/pdfs/species/chinooksalmon_highlights.pdf).

Long term: The historic run sizes of late fall Chinook are not known although they were undoubtedly an order of magnitude higher than they are today. In the 1970s their numbers appeared to be tracking the downward trajectory of winter run Chinook salmon, albeit the numbers did not drop to critical levels in the 1980s. The low numbers recorded in 1993-1996 are presumably the result of poor sampling rather than actual decline to near-extinction. Because actions to protect endangered winter Chinook salmon seem to benefit the late fall Chinook as well, it is reasonable to expect this run to persist as long as winter Chinook actions are successful, barring major disasters.

Status: 2. Late fall Chinook salmon are vulnerable to extinction within the next 100 years or less because of their relatively small population size. The limited area for spawning and rearing would seem to make the single population exceptionally vulnerable to changes in water quality and flow in the Sacramento River, such as might be created by an extended drought or a major spill of toxic materials from Iron Mountain Mine. Its persistence depends entirely on operation of water projects (Shasta Dam) and hatchery operations, which can easily be changed. The late fall Chinook is considered to be a species of special concern by the California Department of Fish and Game and the National Marine Fisheries Service, although the latter agency lumps them with the fall ESU in this category.

Metric	Score	Justification
Area occupied	1	Only one population present in Sacramento River.
Effective population size	4	If average population is 10,000 spawners and the effective population size for salmon is 20% of the actual population (Lindley et al. 2007), then the effective population size is around 2000 fish.
Dependence on intervention	3	Requires periodic actions as for winter run Chinook salmon; importance of hatchery production not well understood
Tolerance	3	Moderate physiological tolerance, multiple age classes
Genetic risk	2	Risk of hybridization with other runs and hatchery fish is high although consequences are poorly known.
Climate change	1	Just one population, in the Sacramento River, which requires cold water from Shasta Reservoir, so vulnerable to extended drought.
Average	2.3	14/6
Certainty	3	Least studied of Sacramento River Chinook runs

Table 1. Metrics for determining the status of Central Valley late fall Chinook salmon where 1 is poor value and 5 is excellent.

SACRAMENTO WINTER CHINOOK SALMON
Oncorhynchus tshawytscha

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

Taxonomic Relationships: For a more complete discussion of taxonomic relationships among Central Valley Chinook salmon, see the Central Valley spring Chinook salmon account. Sacramento Winter Chinook salmon are genetically distinct from all other runs. Historically, there were four presumably distinct populations of winter-run Chinook, in the upper Sacramento, McCloud, and Pit rivers, and in Battle Creek, which have been reduced to a single population that spawns in the Sacramento River below Keswick Dam (NMFS 1997). Winter Chinook possess a life-history strategy, in which they incubate their embryos in the hottest months of the summer. This is unique among *all* populations of Chinook salmon and indicates the unusual geographical and hydrological conditions in which the winter run evolved, where cold-water springs maintain summer temperatures amenable to egg incubation and juvenile survival.

Life History: The basic life history of Chinook salmon is discussed in the North Central Coast Chinook account. Winter Chinook have a life history that differs considerably in its timing from the other three Central Valley runs. Their spawning migration ranges from January to May with runs peaking in mid-March. They enter fresh water as sexually immature adults and migrate upriver to the reaches below Keswick Dam. They hold there for several months until spawning in April through early August (Williams 2006). The timing of winter Chinook spawning puts embryo incubation, which is the most temperature-sensitive life history stage, in the hottest part of the year when water temperatures in California rivers can exceed the lethal range for embryos. Therefore, winter Chinook only existed in areas that had a continuous supply of cold water such as the spring-fed streams of the basalt and porous lava region of the northeastern part of the state; this habitat was lost to them with the erection of Shasta Dam on the Sacramento River in the 1940s. Also, unlike the other runs of Central Valley salmon, winter Chinook tend to spawn at depths of 1-7 meters whereas the others predominantly spawn between 25 and 100 cm (Moyle 2002).

Fry emerge from the gravel from July through mid-October (Yoshiyama et al. 1998, Williams 2006). The duration of the rearing period for winter Chinook is intermediate between the "ocean" type of the fall and the "stream" type of the spring Chinook runs (see Box 1 in Central Valley spring Chinook account), so winter Chinook juveniles rear for approximately 5-10 months before moving down-river (Yoshiyama et al. 1998). Winter Chinook juveniles are similarly intermediate in their size, in that winter-run smolts between January and April average 118 mm FL (Stillwater 2006). The larger size of the smolts results in higher smolt survival during migration and ocean rearing as compared to the fall Chinook, presumably due to diminished vulnerability to predation (Stillwater 2006). Thus, winter Chinook have an advantage over the spring and late-fall runs from longer rearing times in the stream, without juveniles having to over-summer, a tradeoff for spawning in summer (Stillwater 2006). Peak movement for juveniles of all of the runs tends to be at night, thus reducing the risk of predation. 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 they reach the Delta in early winter, another distinctive life history trait that puts them somewhere between stream-type and ocean-type life history. Juvenile entry into the Sacramento-San Joaquin Delta occurs from January to April where winter Chinook complete smoltification and migrate out the Golden Gate to the open ocean to mature (Stillwater 2006).

Habitat Requirements: For general Chinook salmon habitat requirements see North Coast Chinook salmon account. Winter Chinook occur only in the Sacramento River because of their unique life history in which water temperatures must be cold enough in summer to enable successful embryo incubation, but be warm enough in winter to support juvenile rearing (Stillwater 2006). Winter chinook historically migrated high into the watersheds of the McCloud, Pit, and upper Sacramento Rivers to spawn, thereby necessitating an early migration when flows were high enough to allow them passage to the highest areas in the watershed. Winter Chinook will attempt to migrate to the highest upstream spawning location available to them (Stillwater 2006). Once winter Chinook reach their spawning grounds they hold for several months in deep pools with good cover until they are ready to spawn. Of the four runs of Central Valley Chinook, winter Chinook appear to spawn in the deepest water, generally from 0.9-5 meters (USFWS 2003), but have been observed spawning in water as deep as 7 meters (Moyle 2002). Optimal temperatures for holding range from 10-16° C (see temperature chart in North Coast Chinook account) and optimal velocities for winter Chinook range from 0.47-1.25 m/s, significantly higher than selected by the other runs (Table 1,USFWS 2003). Juveniles emerge from the gravel in mid-summer and are restricted in their rearing habitat to those reaches that maintain cool summer temperatures (generally upstream of the mouth of Deer Creek at River Mile 220) between July and September (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.

Run	Range of Suitable Values					
	Velocity		Depth		Substrate	
	ft/s	m/s	ft	m	in	cm
Fall	0.93–2.66	0.28–0.81	1–14	0.3–4	1–3 to 3–5	3–8 to 8–13
Late-fall	0.90–2.82	0.27–0.86	1–14	0.3–4	1–3 to 4–5	3–8 to 10–13
Winter	1.54–4.10	0.47–1.25	3–16	0.9–5	1–3 to 3–5	3–8 to 8–13

Table 1. Ranges of suitable values of velocity, depth, and substrate size for the fall, late-fall, and winter runs of Central Valley Chinook salmon (USFWS 2003).

Because of their distinctive emergence time, winter Chinook fry generally have little competition from other juvenile salmonids during the first few months of their lives, but as they move lower into the rivers, they must share rearing habitat with spring 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 Chinook juveniles (Williams 2006, Stillwater 2006). While this may result in a competitive advantage for spring Chinook, there is some indication that the two runs use habitat differently based on their sizes and thus do not directly compete (Stillwater 2006). Winter Chinook juveniles historically benefitted from the typical winter flooding that took place in the Sacramento River basin and the floodplain habitat that they were able to access for rearing. Sommer et al. (2001) indicate significantly higher growth rates for juvenile Chinook rearing in the floodplain as opposed to those rearing in riverine habitats. Floodplain production and temperatures are considerably higher, thus providing conditions for rapid growth. Rapid growth results in larger out-migrants which presumably have higher survival rates in the ocean. However, there are very few floodplains now available on the Sacramento River, which may have a profound negative impact on winter Chinook recruitment in addition to the loss of spawning habitat upstream of Shasta Dam. Little is known about current juvenile usage of the San Francisco Estuary, but a recent study by the U.S. Army Corps of Engineers indicates that residence time is limited and outmigration through this region is swift.

Distribution: All four of the historic winter Chinook populations are now extirpated from their historic spawning areas in the Upper Sacramento, Pit, and McCloud Rivers and Battle Creek (Lindley et al. 2007). The closing of Shasta Dam halted migration into the Upper Sacramento, Pit and McCloud River drainages. The Battle Creek population of winter Chinook was extirpated by hydropower dam operations that created unsuitable conditions for holding and spawning, particularly during dry years (NMFS 1997, Lindley et al. 2007). Additionally, the weir at Coleman National Fish hatchery was a barrier to upstream migration until recently (NMFS 1997, Lindley et al. 2007). The current single population now holds and spawns at the base of Keswick Dam, where cold-water releases from Shasta Reservoir, combined with artificial gravel additions, have created suitable habitat (NOAA 2005, Lindley et al. 2007). In addition, fish are spawned and reared at the Livingston Stone National Fish Hatchery at the base of Shasta Dam. Juvenile emigration and rearing takes place along the Sacramento River, in various tributary streams, and in the Delta itself (Figure 1) (CalFed 2005).

Abundance: Historical abundance of winter Chinook is thought to have been approximately 200,000 spawners per year (NOAA 2005). Since 1992, numbers have averaged about 10,000 fish, but in 2004-2006, numbers averaged 26,870 +/- 2280 individuals (Lindley et al. 2007). There has been extreme variation in adult escapement (Figure 2), but since listing under the ESA, the population has steadily risen. Accurate abundance data has been difficult to collect and there have been numerous instances (illustrated in Williams et al. 2006) in which putative winter Chinook 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 estimated 18% in 2005, a percentage and overall trend Williams (2006) finds alarming.

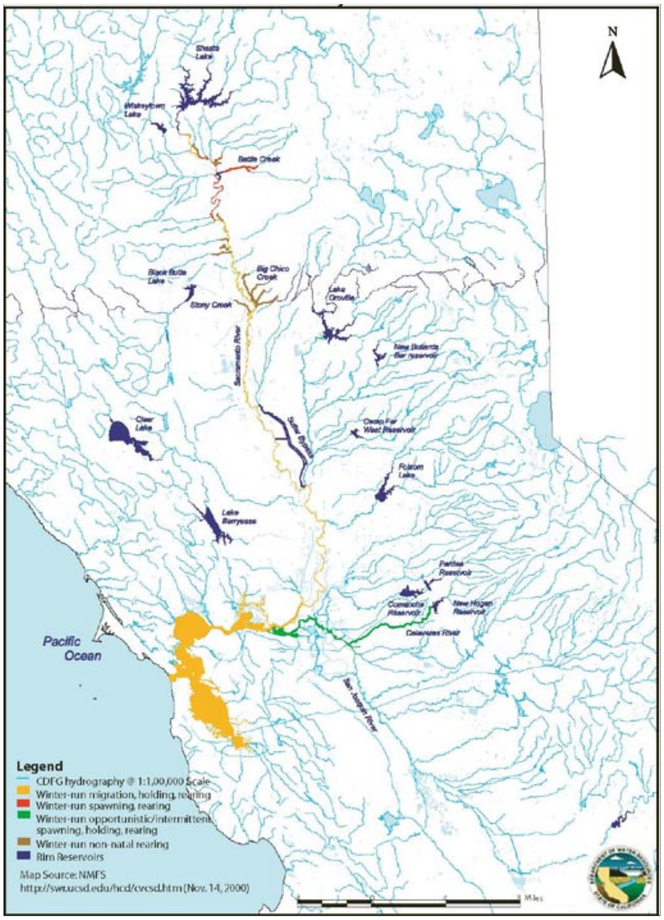


Figure 1. Critical habitat, migration, holding, spawning, and rearing for Central Valley winter-run Chinook salmon. From CalFed Bulletin 250: Fish Passage Improvement (2005).

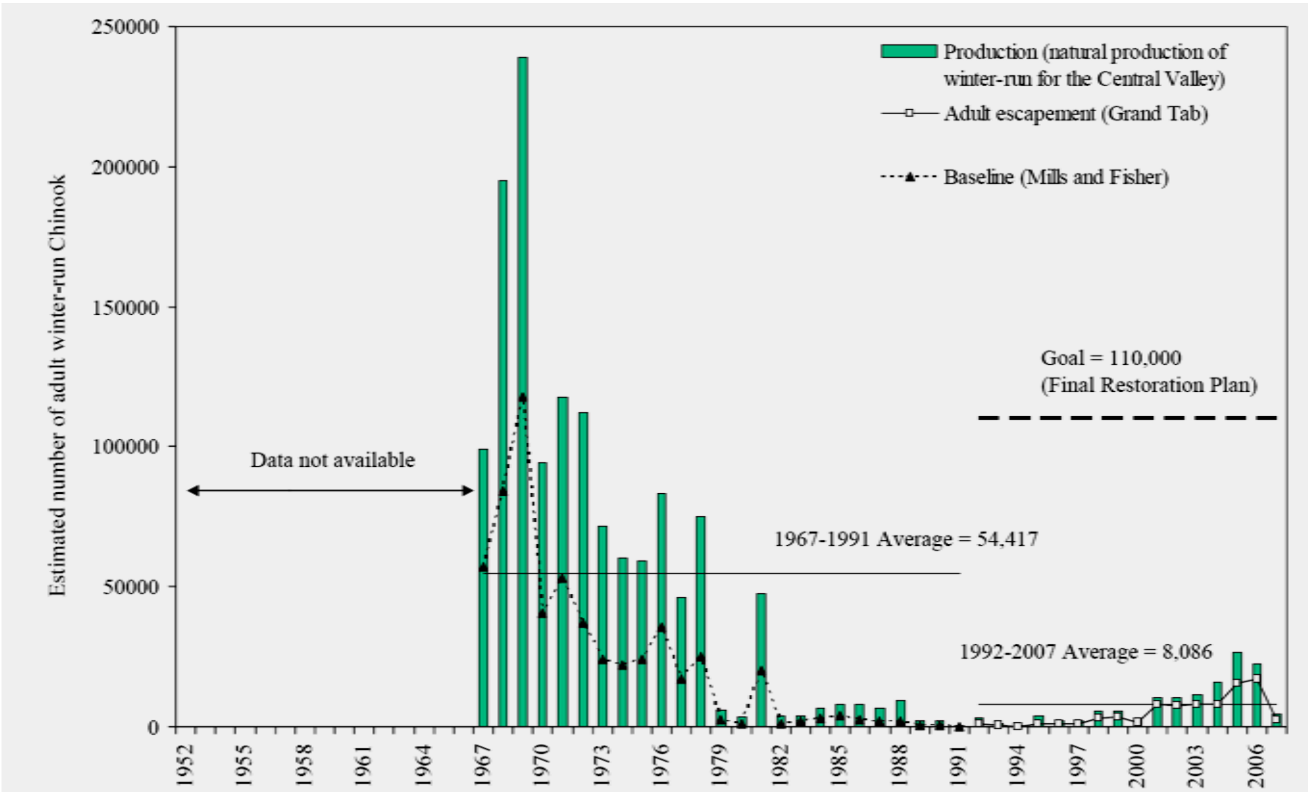


Figure 2. Estimated adult production (escapement plus catch in fisheries) for Central Valley winter Chinook, 1967-2007. Graph from http://www.delta.dfg.ca.gov/afrp/

Factors affecting status: For an overview of factors affecting salmon numbers in the Central Valley, see the discussion in the Central Valley fall and spring Chinook accounts.

The biggest single cause of decline of winter Chinook salmon was the blocking of access to spawning areas by Shasta and Keswick dams in the 1940s. The subsequent steep decline of winter Chinook 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), and commercial and recreational fisheries, which do not discriminate between hatchery fall run Chinook and wild fish of any run. NMFS (1997) has also expressed concern over climatic events that exacerbate the habitat-based problems through extended droughts, low flows and higher temperatures. Additionally, 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 available at sea (NMFS 1997). For a more in depth look at the factors impacting salmon declines on the west coast, see http://www.nwr.noaa.gov/ESA-Salmon-Listings/Salmon-Populations/Reports-and-Publications/upload/chnk-ffd.pdf and the NMFS Recovery Outline (NMFS 2007).

Dams: Shasta and Keswick Dams effectively prevented all upstream migration for winter Chinook, denying access to key spawning and 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 Chinook would survive after Shasta Dam was built (Moffet 1949), but the cold water releases allowed spawning to occur in a previously unsuitable reach of river below

Keswick Dam (NMFS 1997). Keswick Dam, located 14 km below Shasta Dam, regulates the releases from Shasta Dam, as well as flows diverted from the Trinity River. Initially water temperatures were cold enough below Keswick dam for annual spawning of winter Chinook. However, drought years and high levels of water removal rendered water temperatures in the new habitat unsuitable (up to 27°C) with enough frequency so 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).

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 for the fish.

Barriers to Migration: Red Bluff Diversion Dam is widely credited with causing 30 years of significant passage impairment to both upstream migrating adults and outmigrating juveniles due to inadequate fish passage (i.e., poorly designed fish ladders). In addition, predatory fish gathered at the base of the dam, devouring many outmigrating juveniles with the assistance of the RBDD's lighting system which made the juveniles visible at night. This has since been changed. The NMFS Biological Opinion required that the dam gates be raised for six to nine months of the year to allow unimpaired passage and this has significantly increased survivorship and migration success (CDFG 2004).

Water exports and entrainment: Diversions along the Sacramento River presumably have some impact on outmigrating juvenile winter Chinook salmon (but see Moyle and Israel 2006), but more important is probable is direct and indirect mortality at the Central Valley Project and State Water Project pumps in the southern Delta. A color-coded system of red, yellow, and green "lights" (stages) is designed to protect migrating juveniles from too high a level of entrainment mortality. A yellow light goes on with the entrainment of 1% of the estimated number of juveniles entering the Delta; 2% entrainment brings on the red light stage in which results in a mandatory consultation with NMFS under the ESA (CDFG 2004). 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 in order to reduce pumping at other times of year (for protection of Delta smelt and other species) may further increase entrainment mortality rates (Kimmerer 2008).

Iron Mountain Mine: Iron Mountain Mine 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 down the creek, resulting in low levels of dissolved heavy metals in Sacramento River water. However, the solutions must be regarded as temporary, given the potential for dam failure and other factors causing massive pollution of the river. EPA has provided a trust fund of $11 million to be used for salmon restoration in the upper Sacramento to mitigate for the years of damage done by Iron Mountain Mine operations (CDFG 2004).

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 Chinook salmon on their behavior and genetics because hatchery fish are increasingly dominating the population (Williams 2006) and (2) the effects of competition from large numbers of hatchery fall Chinook when ocean productivity is low (Levin et al. 2001). The concerns boil down to the likelihood of hatcheries in the long run accelerating the decline of naturally-spawning Chinook of all runs.

Fisheries: Myers et al. (1998) examined harvest impacts and found that freshwater harvest was negligible, but that ocean harvest had considerable impacts, because fishermen cannot distinguish between hatchery fall run Chinook and endangered winter (and spring) Chinook. In 1994, the ratio of ocean harvest to ocean harvest plus escapement was 0.54, which is a significant impact to an endangered species (Myers et al. 1998). However, harvest rates were estimated to be 0.26, 0.23, and 0.24 for 1988, 1999, 2000 cohorts, respectively (CDFG 2004). This is presumably due to an increase in overall population size and changes to fishing regulations that delayed the season opener to the benefit of winter Chinook and established size restrictions (CDFG 2004, NOAA 2005). Even these harvest rates may be excessive, when combined with other mortality factors (Kimmerer 2008). The Chinook fishery was halted in 2007 because of the rapid decline of the fall Chinook population but the winter Chinook population did not rebound (Figure 2).

Conservation: Since the ESA listing in 1990, there have been a great number of conservation measures instituted for winter Chinook salmon, including opening the gates at Red Bluff Diversion Dam to allow free passage of adults and juveniles, construction of a temperature control device at Shasta Dam, hatchery rearing, habitat improvements, and screening of diversions (NMFS 1997, NMFS 2007).

Red Bluff Diversion Dam blocked free passage up and down river for prolonged periods, so raising the gates on the dam to provide free fish passage was a key action taken to protect winter Chinook salmon, which all spawn above the dam (Stillwater 2006, NMFS 2007). It allowed adults to find their way 'home' easily, with no delays below the dam and it allowed juveniles to pass through the dam with minimal predation (NOAA 2005).

Another important action was the installation of a temperature control device (TCD) 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 benefited the spring and fall runs as well. An additional advantage of the TCD is to regain power generation capability that was lost when cold water releases required by-passing hydroelectric generators (CDFG 2004).

Improving habitats for spawning and rearing of Chinook salmon is an on-going process in the Sacramento River. Probably the most important action in the winter Chinook spawning reach has been addition of gravel on a regular basis to provide more spawning habitat (NMFS 2007). Other habitat improvements, such as riparian and floodplain restoration are discussed in the fall Chinook account.

Hatchery rearing of winter Chinook began as a desperation effort to save a species that seemed headed for extinction. Prior to 1997, numbers were so low that some winter Chinook were reared through their entire life cycle at Bodega Marine Laboratory, the Steinhart Aquarium in San Francisco, and Livingston Stone Hatchery below Shasta Dam (NMFS 2007). As populations started to recover, this program was halted although the Livingston Stone Hatchery still produces 200,000 smolts per year. The percentage of hatchery fish appearing in the spawning population is high enough so that Williams (2006) characterizes it as "worrying."

There has been a concerted effort at improving diversions through screening and a number of large projects such as the Anderson-Cottonwood Irrigation District, Glenn Colusa Irrigation District, Reclamation District 108, and Reclamation District 1004 (CDFG 2004). More importantly, a large CALFED-sponsored restoration project on Battle Creek should 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 was slated to begin in 2005 with a budget of $70 million and affects nine PG&E hydropower installations (CDFG 2004). Since that time, the environmental analyses have been completed and current contracts are being finalized for distribution of monies between agencies. It is hoped that the final contracts will be ready by 2008, at which point work on the actual project can begin (Mary Marshall, USBOR pers. comm. 10/2007). This interagency project is touted as being a model situation in which both habitat and power production efficiency are improved (Mary Marshall, Bureau of Reclamation. pers. comm.).

The problem with the fishery lies in the lack of marking of all hatchery Chinook, which are mostly fall Chinook and make up the bulk of the fish caught. However, fishers cannot distinguish between hatchery and wild fish. A mark-selective fishery would conceivably increase survival of adult fish if and when the fishery is restored.

Trends:

Short term: Lindley et al. (2007) performed a population viability analysis and risk assessment on winter Chinook. They determined that the population is trending upwards with an estimated growth rate of 28% per year and an average of 8,140 spawners in a given year. The dramatic upswing in the population since the extreme lows of the 1990s indicates a positive trend for the species and Lindley et al. (2007) scored winter Chinook as having a low likelihood of extinction in their risk assessment. Nevertheless, 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 no geographic redundancy in the species at this time. Furthermore, Lindley et al. (2007) cited anthropogenic incidents such as toxic spills and other pollutants as negatively impacting populations. A particularly severe problem would be failure of the dirt dam holding back toxic waste from Iron Gate Mine, which could wipe out fish in a long reach of river. In addition to catastrophic events, Lindley et al. (2007) showed that proportion of hatchery-produced fish spawning in the wild was on the rise. There has been >5% hatchery-origin spawners since 2001, and in 2005, hatchery-origin spawners made up 18% of natural spawning. If spawning contributions from hatchery fish exceed 15% in the 2006-2007 season, then winter Chinook will be reclassified as "moderate" risk due to problems associated with hatchery influence on fitness and survival, and lower levels could still have adverse impacts on wild fish (Lindley et al. 2007). Unfortunately, in 2007, less than 2,500 winter Chinook returned to spawn (Figure 2) indicating that the same factors affecting fall Chinook were also affecting winter Chinook, making it likely that risk of extinction has increased.

Long term: Winter-run Chinook are 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. They have declined from having perhaps 200,000 fish divided among four populations, to having a few thousand (once a few hundred) in just one population. Because of their limited distribution (spawning only downstream of Keswick Dam), a population viability analysis gave them a moderate chance of becoming extinct within one hundred years, even assuming no major disasters happen (Lindley et al. 2007). Continued improvements to habitat and the maintenance of tolerable water temperature and flows are needed to ensure their continued persistence. However, they have no population redundancy and are therefore vulnerable to catastrophic events and prolonged drought. Additionally, current numbers are only 3% of their post-1967 peak mean (NOAA 2005) and they seem to be affected by the same factors causing the crash of fall Chinook populations in recent years. Continued efforts towards improving habitat and restoring access to historical spawning areas (e.g., getting past Keswick and Shasta Dams) and the restoration project on Battle Creek will further increase their viability.

Status: 2, possibly 1. Winter Chinook salmon have a high likelihood of extinction within the next 50 years, as reflected in their listing as an endangered species by both state and federal governments. In 1985, the California-Nevada Chapter of the American Fisheries Society (AFS) petitioned the National Marine Fisheries Service (NMFS) to list Sacramento River winter Chinook salmon as a threatened species under the Endangered Species Act (ESA) (NMFS 1997). In 1987, NMFS concluded that, while the winter Chinook salmon 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 Chinook were listed as threatened in 1990. They were subsequently reclassified as endangered in 1994 (NMFS 1997) a status that was reconfirmed in 2005, and were listed as endangered by the State of California as well.

Winter Chinook salmon's continued persistence shows their remarkable resiliency and adaptability but their fluctuations also indicate their fragility. Their current status is still endangered under both the state and federal endangered species acts and while the population shows positive growth, run numbers are still a shadow of historical levels. Winter Chinook remain extremely vulnerable to loss or alteration of their adopted habitat and it is critical that continued habitat improvement and protection take place. Climate change is likely to make protecting the single wild population increasingly difficult. The restoration project at Battle Creek may help this situation, but it will not be completed for a number of years. Lindley et al. (2007) provide thermal suitability maps based on several warming scenarios and without passage around key artificial barriers to cooler headwater areas, the impact to all the runs of Chinook may be severe. Continued monitoring is critical, as well as developing adaptive management strategies should warming in their current habitat approach or exceed their thermal tolerances.

Metric	Score	Justification
Area occupied	1	A single population in a reach below dams; extirpated from their historical range.
Effective population size	4	The recent assessments indicate an average of 10,000 returning spawners, therefore an effective population size of 2000. In 2007, however, EPS was around 500 fish.
Intervention dependence	1	The population depends entirely on releases from Shasta Dam and secondarily on rearing in Livingston Stone Fish Hatchery.
Tolerance	1	Winter Chinook spawn in the most thermally challenging times of the year and are particularly at risk from drought or climate change.
Genetic risk	2	Considerable genetic drift has probably occurred with the consolidation of the winter Chinook populations into a single population with limited habitat.
Climate change	1	Extremely vulnerable because of reliance on releases from Shasta Reservoir.
Average	1.7	10/6
Certainty (1-4)	4	Well studied populations

Table 1. Metrics for determining status of winter Chinook salmon, in which 1 is a poor value and 5 is excellent.

CENTRAL VALLEY SPRING CHINOOK SALMON
Oncorhynchus tshawytscha

Description: All California Chinook salmon are similar in morphology and other characteristics (see Central Coast Chinook salmon for a full description). Various ESUs and runs are distinguished mainly by genetics and life history traits (e.g., run timing) although there are often statistical differences in size.

Taxonomic Relationships: Within the genus *Oncorhynchus*, Chinook are most closely related to coho salmon, with which they occasionally hybridize (Moyle 2002). There are many distinct populations within the species that are generally referred to as "runs" or "stocks." Chinook runs are named after the season in which they begin their fresh water spawning migrations and populations are delineated genetically and geographically. In California's Central Valley, there are four distinct runs: fall, late fall, winter, and spring. Each is distinct in timing of spawning and migration, as well as location of spawning areas. They can be distinguished using molecular genetic techniques. Genetically, there are two distinct populations of spring Chinook in the Central Valley, those that spawn in Deer and Mill Creeks (Tehama Co.), and those that spawn in Butte Creek (Tehama Co.). In addition, there is a putative population of spring Chinook in the Feather River which is nearly identical genetically to fall run Chinook salmon (Williams 2006).

Life history: The basic life history of spring Chinook salmon is to migrate upstream in spring, hold through the summer in deep pools, and then spawn in early fall, with juveniles emigrating after either a few months or a year in fresh water. Central Valley spring Chinook salmon (CVS Chinook), however, have considerable flexibility in their life history strategies and consequently do not fit well into the life history categories for most other Chinook salmon populations (Box 1).

These salmon begin their spawning migration from February to early July with the migration peaking in mid-April in Butte Creek and in mid-May in Deer and Mill Creeks (Williams 2006). They migrate as silvery, immature fish that mature after they reach their summer holding areas, which are generally higher in the watershed than those of other runs. They travel high into watersheds in order to find deep pools with cool summer temperatures. Spring Chinook often do not stay in the same pool for the duration of the summer, but move from pool to pool, generally moving upstream. They often spawn in the tail waters of their final holding pool (Moyle 2002). Spawning behavior is similar to that of coho salmon. Each female digs a redd in the appropriate substrate and generally a large male fights off other males in order to spawn with her. The gametes of the dominant males are often "supplemented", however, by one or more jacks (two-year-old males) that spawn by sneaking into the nest with the mating pair and releasing their milt as the female releases her eggs (Moyle 2002, Williams 2006).

CVS Chinook maintain a large degree of plasticity in their age at spawning. A significant proportion of the run can be made up of jacks that return to the rivers to spawn after only a single year in the ocean. Age at spawning for spring Chinook salmon varies from age 2 to age 4; approximately 69% of the spawners returning to Butte Creek in 2003 were estimated to be age 4 (McReynolds et al. 2006). There have been observations of sexually mature 1-year old male parr that never go to sea. They spawn in much the same way as jacks. 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. This variability in the male reproductive strategy ensures that around 90% of eggs are fertilized and that a high proportion of the available genes in the population are passed on (Moyle 2002).

Box 1. Chinook salmon life history strategies

Chinook salmon have a tremendous variety of life history adaptations that allow them to persevere through variable and diverse environmental conditions. Chinook are often divided into two life-history strategies, 1) stream-type and 2) ocean-type. Initially, these types simply distinguished salmon that did or did not spend a winter in fresh water before migrating to sea, as revealed by growth patterns in their scales (Gilbert 1913). Later, other characteristics were associated with these types. Generally speaking, ocean-type refers to a population in which juveniles begin their migration to the sea soon after emerging from the redd, spend less than a year in fresh water, and as returning adults, spawn soon after reentering the river. Alternately, stream-type Chinook stay in the stream for longer than a year before initiating seaward migration and reenter fresh water in spring as sexually immature fish. They then mature in the stream over the summer months before spawning in early fall. While in the sea, ocean-type Chinook tend to forage close to the coast, whereas stream-type Chinook venture farther out and forage in the open ocean; stream-type Chinook displaying these characteristics predominate north of 55º latitude, while ocean-type Chinook predominate south of 55° latitude (Healey 1991). Healey (1991) postulated that steam-type represent an Asian or Beringian lineage that had been separated from a Cascadian ocean-type lineage during the last glaciation.

However, Williams (2006) noted that the more southerly spring-run Chinook populations, especially south of the Columbia River, may 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). In summary, Central Valley spring Chinook salmon generally exhibit both ocean-type and stream-type life history patterns in the freshwater juvenile stage but both types apparently forage in coastal waters. 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).

The upper limit of temperature tolerance for adult Chinook appears to be between 21 and 24º C. Evidence from Butte Creek indicates that more than a few consecutive days with daily mean temperatures >= 21 º C increases mortality. Eggs and juveniles are less tolerant and thus adults wait until stream temperatures drop in the fall before spawning, which begins after water temperatures reach around 13-15° C (Williams 2006). Preferred spawning habitat seems to be at depths of 25-100 cm and at water velocities of 30-80 cm/sec, though they 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 are constructed over 2-10 m^2, where the loosened gravels permit steady access of oxygen-saturated water. Embryos are the most sensitive life history stage and have a narrow range of temperature tolerance with considerable mortality occurring at temperatures above 14-16° C (See Central Coast Chinook salmon account for a full description of temperature tolerances). 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 venture forth 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, possibly as a resulted of limited rearing habitat available in the upper watersheds (Stillwater 2006). Some begin their emigration as fry mere hours after emerging from the gravel. Most begin smoltification after a few months of stream rearing and outmigrate as sub-yearlings. A third type remains in the stream for a year, oversummering in their natal stream before beginning their downstream emigration (Hill and Webber 1999, Stillwater 2006). As they move downstream, young CV Chinook of all runs use the lower reaches of non-natal tributaries and the shallow edges of the mainstem to obtain respite from high flows, feed on plentiful aquatic invertebrates and larval fish, and hide 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 floodplains that allow for rapid growth. Sommer et al. (2001) and Jeffres et al. (2008) indicate significantly higher growth rates for juvenile Chinook rearing in the floodplain as opposed to those rearing in riverine habitats. Floodplain production and temperatures are considerably higher and thus provide an important resource for outmigrating juveniles. The extensive levee building that has taken place along the Sacramento River has prevented Chinook juveniles from accessing those 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.

CVS Chinook apparently rear on available floodplains and tidal marsh habitat of the Sacramento-San Joaquin Delta, though they may also utilize the shallow habitats of San Pablo Bay (Williams 2006). Smolt usage of tidal marshes, mudflats and bays of the San Francisco Estuary is not well understood and is understudied (Williams 2006). There is considerable inter- and intra-annual variation in habitat use that varies in part with size of fish. Ocean-type spring Chinook enter the estuary at a smaller size than do the stream-type that spend more time growing in the upper watershed; these small fish therefore presumably have greater necessity to use resources available in the estuary and spend more time there before making the final migration out into the ocean. Food type and availability varies with habitat but in general, aquatic and drift insects, amphipods, copepods, and small crustaceans are available throughout the brackish regions of the estuary. Studies from the early part of the 20th century indicate that young Chinook frequently appeared in trawls and beach seines at locations throughout the lower estuary (Scofield 1913, Williams 2006). We can infer from studies of other estuaries, such as the Columbia River, that estuaries can play an exceedingly important role in smolt growth and survival; size of smolt upon ocean entry appears to be a strong determinant of survival in the first year at sea (Williams 2006). Juvenile spring Chinook that rear on the Sutter Bypass floodplain will likely emerge from that habitat at sizes larger than 70 mm FL and can then proceed to the estuary quickly without utilizing much of the downstream habitat available in the Delta (Hill and Webber 1999). 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 reflect the immense changes and habitat alteration that have taken place in those areas over the last century (e.g., MacFarlane and Norton 2002) The bulk of tidal marsh and creek habitats have been leveed, channelized, and dredged for navigation while water transfers at the Delta pumps have drastically altered hydrology, salinity, and turbidity in the lower Delta. Additionally, numerous non-native fishes and invertebrates have invaded the San Francisco Estuary and it is possible that predation from introduced predators such as striped bass have affected Chinook survival and behavior.

Once smolts arrive in the ocean in the late spring and summer months, they feed on a variety of crustaceans, euphausids, and prey fishes (MacFarlane and Norton 2002, MacFarlane et al. 2005). The condition after the first summer of feeding in the ocean is thought to be a good predictor of smolt survival over their first winter (Williams 2006). It appears that a certain threshold of food abundance must be reached, although in the warmer regions surrounding San Francisco Bay, this threshold may not be as absolute as in the more northerly regions. The California Current creates an area of upwelling along the California coast and it is probably for this reason that newly arrived emigrant salmon mostly forage in this area, rather than farther out in the open ocean. The majority of lifetime growth and weight gain takes place in the ocean. As the young Chinook grow larger, their diet shifts from crustaceans to predominantly fish (such as herring, anchovies, juvenile rockfish, and sardines), and growth becomes very rapid (Moyle 2002).

Size at entry to the ocean differs between stream-type and ocean-type fish, with stream-type fish generally being larger than their ocean-type counterparts. Once in the ocean, growth rates are similar, but the sizes at entry can determine lengths of adults returning to spawn at a given age (Moyle 2002). An additional selector for size is commercial and sport fisheries that take the larger and older fish, which results in smaller (and younger) adults returning to spawn. 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 remarkable number of habitats during their lives and nearly every life history stage requires different habitat (see Central Coast Chinook account for general details). In general, water temperature determines their presence in a particular stream segment. Maximum weekly average temperatures usually do not exceed 21° C, although there is some evidence that spring Chinook in some areas may be able to tolerate slightly higher temperatures. Adult spring Chinook returning to spawn require deep pools with good cover to hold in over the summer. Most spawners reach the summer holding areas by July and select deep (>2m) pools in which to hold. These pools typically have bedrock bottoms and moderate velocities (15-18 cm/sec) and should contain abundant hiding places such as rock ledges, bubble curtains, and woody debris to provide cover (Moyle 2002). Spawning begins once water temperatures decrease to tolerable levels, around 15°. Spawning gravel varies in size, but the most important aspect is good hyporheic flow that provides oxygen-saturated water to the embryos buried in the gravel (Moyle 2002). Ocean-type fry spend longer in the lower reaches of the river and in the Delta, foraging in the shallows at the river's edge, rearing on the floodplains of the Central Valley before smoltifying (Williams 2006). Juveniles that emigrate as yearlings are more likely to become smolts on the downstream migration and not spend much time in the Estuary. In the ocean, both stream-type and ocean-type fish from the Central Valley stay close to the coastal shelf, foraging on the considerable food sources resulting from upwelling of nutrients in the California current (Williams 2006).

Distribution: Spring Chinook salmon historically ranged throughout the Sacramento and San Joaquin watersheds. Lindley et al. (2004) indicate that historically there were 18 independent populations ranging from the Pit River to the southern reaches of the upper San Joaquin (Figure 1). According to Lindley et al. (2007), these populations inhabited five distinct geologic/hydrologic regions: 1) Basalt and porous lava, 2) Northern Sierra Nevada, 3) Northwestern California, 4) Southern Sierra Nevada, 5) Central Valley domain. Within these regions, CVS Chinook distribution is determined by both accessibility of habitat, and water temperature. Many of the streams in the basalt and porous lava region are fed by springs coming through volcanic rocks with precipitation falling 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 spring Chinook and decreased variability in natural instream flow. In contrast, the bulk of the precipitation in the four Sierra Nevada regions, particularly in the southern region, 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 peaks, then tapers off over the summer months into the early fall when temperatures cool enough for spawning. The historical timing of high flow in the spring and early summer provided enough flow for CVS Chinook to reach their summer holding areas. Upstream migration for CVS Chinook was generally truncated by impassible barriers such as waterfalls that limit their access to higher, cooler reaches.

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 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 are extirpated.

In the Sacramento drainage, CVS Chinook once ranged upstream into the Fall, Pit, McCloud, and upper Sacramento Rivers, from which they have been excluded since the 1940s by Shasta Dam. Today, some CVS Chinook can be found in Battle Creek and in the Sacramento River below Keswick Dam, but current distribution of viable spring Chinook populations is limited to just a handful of streams in the northern Sierra Nevada Region. This includes naturally reproducing populations on Mill, Deer, and Butte Creeks. CVS Chinook also occur on a regular basis in some of the smaller tributaries, such as Big Chico, Little Chico, Begum, and Clear Creeks but these populations are presumably not self-sustaining (Lindley et al 2007). The Feather River Hatchery releases about 2 million “apparent” CVS Chinook smolts per year. However, Feather River CVS Chinook have hybridized with hatchery fall-run Chinook and are genetically more closely related to them than to wild CVS Chinook populations. Potential runs in the Yuba River watershed are too data deficient for conclusive analysis. An alternative hypothesis is that Feather River spring-run are a recent divergence from fall-run chinook that recolonized the Feather River after hydraulic mining ended (J. Williams, pers. comm. 2008).

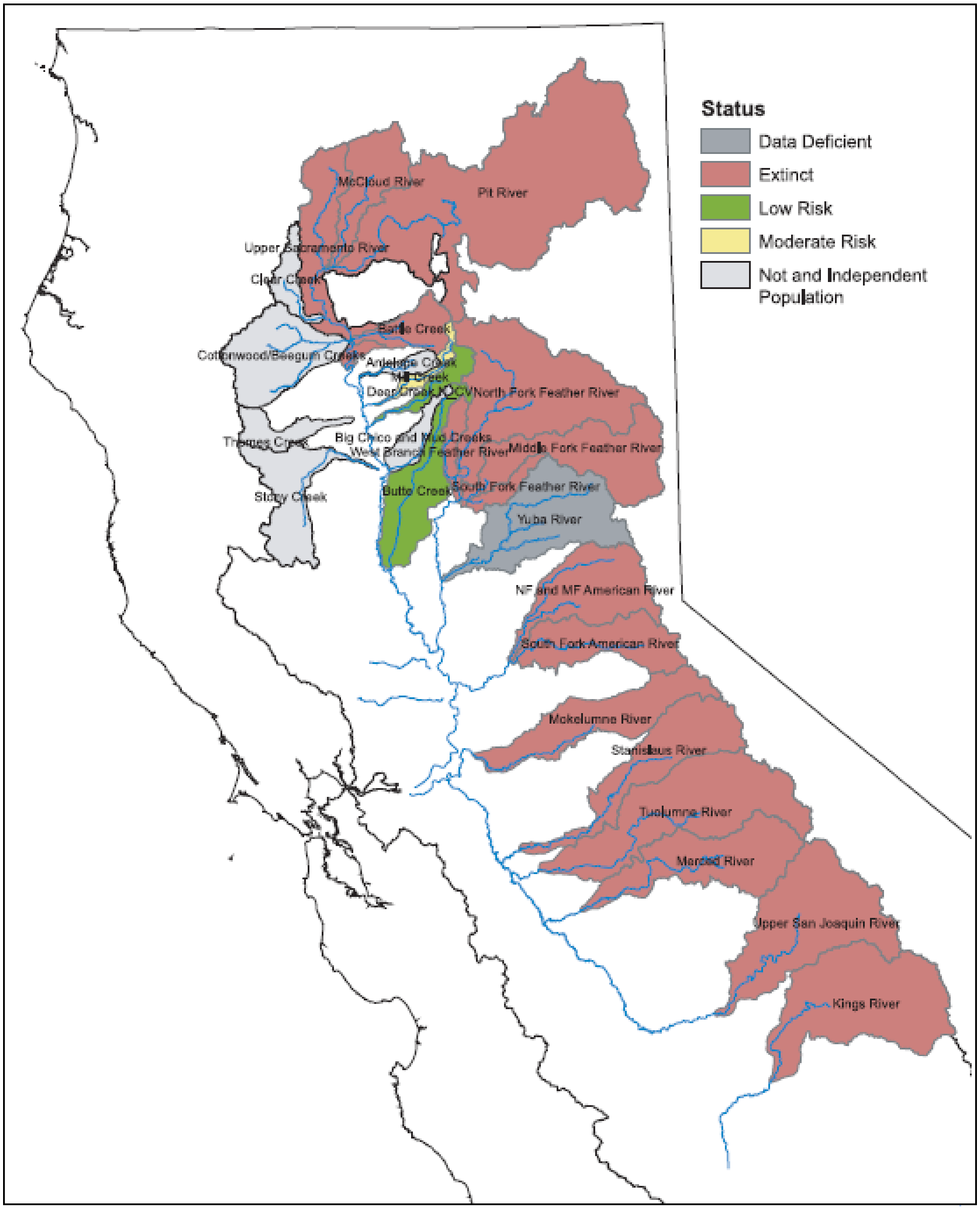


Figure 1. Status of historical spring Chinook populations in the Central Valley of California from Lindley et al. 2007.

Abundance: CVS Chinook have been extirpated from the vast majority of their historic range. 19th century combined run sizes were probably in the range of 1 million fish per year +\- 500,000 (Yoshiyama et al. 1998). Not counting Feather River salmon, total production (escapement plus catch in fisheries) has averaged about 16,000 fish since 1992, although escapement has been less than 1000 fish in some years (Figure 2). Lindley et al. (2007) performed a population viability analysis (PVA) on existing stocks and found that Butte Creek had the largest adult escapement at 22,630 individuals, Mill Creek had 3,360 individuals, and Deer Creek had 6,320 individuals as the mean of $\geq$ 3 years of spawning run estimates. Effective population size was estimated to be approximately 20% of the actual population (Lindley et al. 2007). The fluctuations in abundance of CVS Chinook in Butte Creek and the factors influencing the numbers are presented in Table 1.

Years	Ten year total	Ten year average	Significant Events/Management Activities
Pre-1966	--	--	DeSabla Dam erected in 1903, probably existed in smaller form since Gold Rush days. Project expanded by PG&E to include 3 reservoirs, 3 powerhouses, 14 diversion and feeder dams, 5 canals, and associated equipment and transmission facilities. The installed capacity of the three powerhouses is 26.6 megawatt (MW). (From the public website for the relicensing of the DeSabla-Centerville Project, FERC #803, http://www.eurekasw.com/DC/relicensing/default.aspx)
1966-1975	3375	336	Low spring-run Chinook returns. Lowest recorded number: 80 in 1966. Highest recorded number: 1000 in 1965.
1976-1985	1621	162	Lowest recorded number: 10 in 1979. Highest recorded number: 535 in 1982. Four years of returns <100. PG&E ordered to increase flows to 20 cfs. 1970 CDFG plants adults from Sacramento River to try to increase spawning.1984 CDFG plants 200,000 juvenile broodstock from the Feather River.
1986-1995	13539	1354	Lowest recorded number: 14 in 1987, probably the result of back to back low runs in 1983 and 1984. Highest recorded number: 7,500 in 1995. PG&E ordered to increase flows to 40 cfs in 1992, run immediately increases from 100 to 750. CDFG closes all fishing 1994. In 1995, first return of successful 1992 spawners after flow increase.
1996-2005	94538	9454	Lowest recorded number: 625 in 1997. Highest recorded number: 20,212. In 2002, 3500-7000 fish die in pre-spawning mortality, 11,000+ die the following year, presumably due to high water temperatures. Low pre-spawning mortality in 2004 due to improved water temperature management. All but 1 year class returns >1000, 4 years of >10,000 returning spawners.

Table 2. Decadal record of population averages from 1966 to the present for Butte Creek, Tehama County, with notes on significant events and management activities.

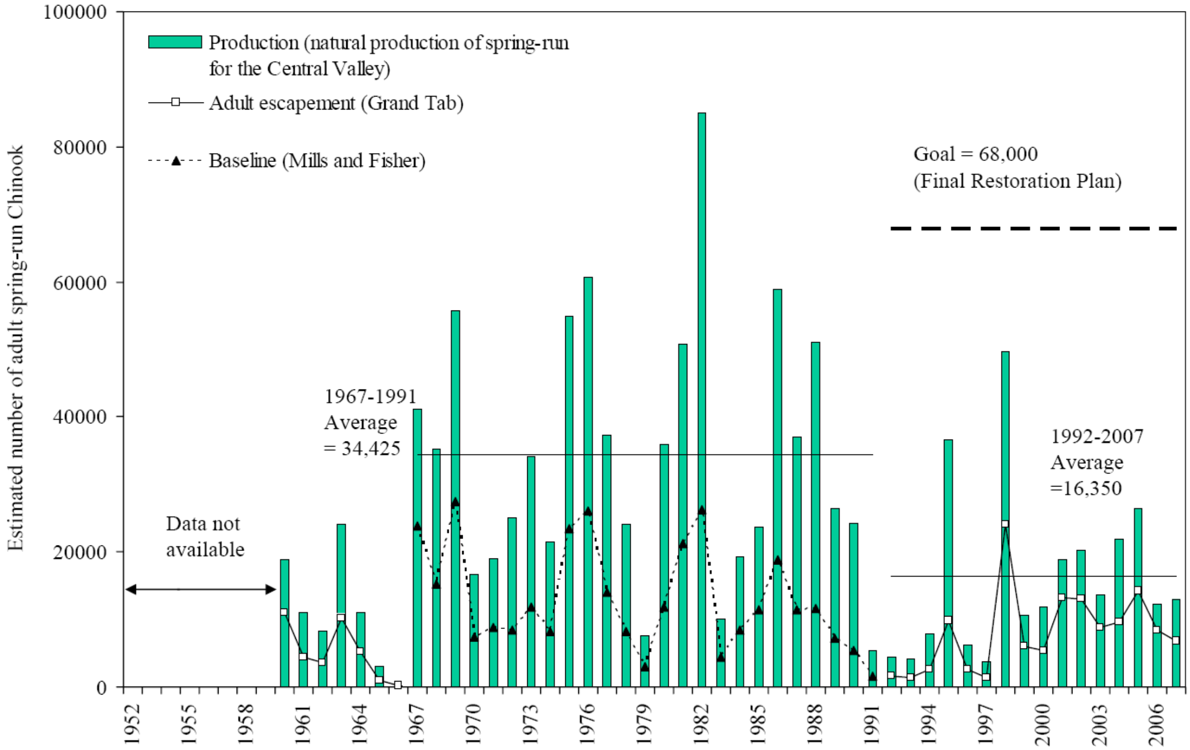


Figure 2. Estimated total production (escapement + catch in fisheries) and escapement of spring run Chinook salmon in the Central Valley. These figures do not include salmon from the Feather River. Source: http://www.delta.dfg.ca.gov/afrp

Factors Affecting Status: Major factors affecting, or potentially affecting, the status of spring Chinook include 1) dams, 2) diversions, 3) urbanization and rural development, 4) logging, 5) grazing, 6) agriculture, 7) mining, 8) estuarine alteration, 9) fisheries, 10) hatcheries, and 11) 'natural' factors. For a discussion of other and more general factors see the account for Central Valley fall Chinook salmon.

Dams: The major cause of the widespread extirpation of spring-run Chinook salmon has been dams that block access to over 90% of their historic spawning and summer holding areas, including all of the San Joaquin drainage, the entire northern Sacramento basin, and the central Sierra Nevada streams such as the Yuba, Feather, and American Rivers (Yoshiyama et al. 1998). All but three historic spawning areas are either behind impassable dams or are strongly impacted by dams and do not support viable populations at present. The remaining independent spawning populations (Mill, Butte, and Deer Creeks) have been negatively affected over the last century by water diversions, small dams, and between-basin transfers.

The large dams also change the flow patterns of the rivers they regulate, reducing the ability of remaining floodplains to flood and reducing the length of spring outflow events that push juvenile salmon downstream and move adults upstream. The cold water releases from the dams attract spring run to the river reaches below dams, where they can easily hybridize with fall run because of the loss of the historic spatial separation of the two runs. This may be a major reason why fall and spring run in the Feather River are not very distinct genetically.

Diversions: There are numerous diversions along the Sacramento River which can potentially entrain spring run Chinook fry and smolts. The larger diversions are all screened and presumably offer some degree of protection from entrainment, while smaller diversions by and large do not need to be screened if they are located on the main river (Moyle and Israel 2006). The intakes in the main river also tend to be deeper than most salmon occur. The large pumps in the south Delta, from the State Water Project and the federal Central Valley project may also have an impact on spring Chinook salmon populations but the impact is uncertain enough so that a federal court decision in April 2008 has forced the National Marine Fisheries Service to revise its somewhat optimistic biological opinion on the impacts of pumping. See Central Valley fall Chinook for further discussion of this factor.

Urbanization and rural development: The towns, suburbs, and ranchettes located along the Sacramento River and its tributaries presumably have an impact on spring Chinook rearing in the main river and its tributaries through polluted run-off, sedimentation, loss of riparian habitat, small diversions, and the dozens of other human actions that disturb aquatic habitats. The effects so such actions, however, are poorly documented. Increased urbanization in the greater Chico area also puts pressure on spring Chinook, because more people spend recreational time on the streams (e.g., rafting and swimming in Butte Creek can disturb holding adults).

Logging: Logging has been, and continues to be, an important economic activity in the watersheds surrounding the current spring-run streams. While forestry practices have generally been fairly benign in the Deer and Mill creek watersheds and have improved in recent years, there have been historic impacts to streams from logging and its associated road-building, resulting in erosion, landslides, and loss of riparian vegetation.

Grazing: Cattle grazing occurs throughout most of the extant spring-run watersheds and there remain basin-wide impacts from grazing which 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 reduce abundance and quality of macroinvertebrates used 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 by landowners.

Agriculture: Historically, the biggest impact of agriculture on spring Chinook salmon was construction of the massive levee system in the Central Valley in the 19th and early 20th centuries in order to prevent flooding of agricultural fields and towns (Kelley 1989). The levees also caused the lower river to down-cut, in part to flush out sediments from hydraulic mining (which had raised river levels and exacerbated problems with flooding).The result was loss of floodplain and backwater habitat important for rearing juvenile Chinook salmon. Historically, juvenile spring Chinook salmon would have left their natal streams in spring to find abundant rearing habitat in river backwaters, edges, and floodplains. Recent studies suggest that floodplains were extremely important rearing areas for juvenile salmon in general (Sommer et al. 2001), allowing them to grow faster and achieve larger sizes before going out to sea. Today this habitat is largely absent along the channelized Sacramento River and diked Delta. A few backwater areas still exist along reaches of the river in the Chico region but patches of 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 spring Chinook take advantage of the favorable rearing conditions found there. The impact of loss of this historic rearing habitat on spring 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 survival of out-migrants.

On the Feather River, water is diverted from Oroville Dam and warmed in a shallow reservoir (Thermolito) for rice farming, with excess warm water returned to the river. This influx of warm water can potentially raise instream temperatures to lethal levels for over-summering spring Chinook. Agricultural return water also contains pesticides and other contaminants which may affect juvenile Chinook health and survival.

On the San Joaquin River, early diversion dams, levees, and similar projects eliminated much of the rearing habitat for juvenile salmon that persisted through the summer as the result of cold-water flows from the high Sierra and from artesian ground water. The rearing habitat was found in the braided channels still faintly visible on aerial photographs and in floodplains.

Mining: Presumably an initial factor in spring Chinook decline was hydraulic gold mining in the late 19th century, which radically altered holding areas in much of the Sierra (Merced to Feather Rivers). Historic mining during the California gold rush resulted in the destruction of many of the streams used by all runs of Chinook, but especially spring Chinook which require high quality habitat and cold water all year around. Hydraulic mining washed millions of tons of sediment into streams, covering spawning gravel and destroying habitat. 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 and high sediment loads in rivers after winter storms are a continuing legacy. Historic records indicate that runs in the rivers subjected to hydraulic mining were extirpated for some time until conditions improved and the salmon were able to recolonize areas not blocked by dams (Williams 2006).

Toxic mining wastes, mainly from abandoned mines, are another legacy affect on spring Chinook. The principal threat today is the potential for a major spill of highly toxic waste from Iron Mountain Mine, if the check dam on Spring Creek should fail. A failure could potentially send a massive plume of toxic water down the Sacramento River, with lethal consequences to any fish residing there.

Estuarine alteration: The San Francisco Estuary is a very different ecosystem today than the one in which Central Valley Chinook salmon evolved. While there have been few studies of juvenile spring 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 is probably indicative of the immense changes and habitat alteration that have taken place in those areas over the last century (MacFarlane and Norton 2002). Historically, juvenile spring Chinook would have arrived in an estuary that was a complex of tidal marshes, with many shallow channels, rich in small crustaceans and aquatic insects. In this system, they could physiologically adjust to changing salinities, 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 is high. Thus, it ‘pays’ for juvenile salmon to move through estuary as rapidly as possible, at considerable cost in energy and vulnerability to predation (and the pumps in the South Delta).

Fisheries: In the nineteenth century, commercial fisheries decimated spring Chinook populations. 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 until the completion of the major rim dams around the valley eliminated most spawning and rearing habitat for spring Chinook. The impacts of commercial and sport fisheries in recent years (prior to recent closures to protect fall run) have been through incidental take in the ocean fisheries that are largely supported by hatchery fish. It is likely that such take has been a significant source of mortality for the diminished populations of spring Chinook, but its impact is not well understood because the lack of a program that marks all hatchery fish, which would enable wild fish (such as spring run Chinook) to be distinguished from hatchery fish. Fisheries also select for younger, smaller, and less fecund fish as spawners, reducing resiliency of the populations.

Hatcheries: There is little obvious hatchery influence on Mill Creek, Butte Creek, and Deer Creek populations, but Battle Creek and the Feather River are strongly influenced by the activities of Coleman National Fish Hatchery, which releases an estimated 2 million "apparent" spring Chinook smolts per year and has received criticism for mingling spring and fall-run stock in the past (Williams 2006). While Butte Creek and Feather River spring Chinook appear genetically distinct, in 1986 200,000 juvenile Feather River spring Chinook were planted into Butte Creek, in response to extremely low numbers of returning fish. However, there is little evidence that this plant had any effect on Butte Creek populations. See Central Valley fall Chinook salmon account for a more extensive discussion of hatcheries.

'Natural' factors: Forest fires, volcanic activity, drought, and climate change all have exceptionally large potential to affect spring Chinook because three major populations are located closely together in the 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 forest land 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 30 km width could simultaneously burn the headwaters of all three populations, leading to heavy potential impacts on spring Chinook. Such a fire has a 10% chance of occurring in any given year in California (Lindley et al. 2007). Likewise, all spring Chinook populations are vulnerable to volcanic eruptions from Mt. 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 volcanic eruption. The USGS has classified Mt. Lassen as "highly dangerous" (Lindley et al. 2007).
Prolonged drought could also easily render most existing spring Chinook habitat unusable, either through temperature increases or lack of adequate flows, even though the streams are partially spring-fed. The potential effects of climate change could have much the same result.

Conservation: Until fairly recently few people gave much thought to protecting CV spring Chinook salmon as a distinct entity because hatchery-raised fall Chinook seemed to satisfy commercial and recreational desires for salmon. Hatchery fish also satisfied whatever legal obligations water agencies acquired from destroying the spring runs in most tributaries through dam construction. Thus, the U.S. Bureau of Reclamation could construct Friant Dam and literally dry up the San Joaquin River and let spring Chinook salmon in the San Joaquin basin go extinct. The fact that spring Chinook managed to persist in three small watersheds in Tehama County is mostly a matter of luck: the streams were too small for economically feasible dams. The lead in protecting this distinctive run was taken by agencies and landowners in the basin, the latter organized as the Deer Creek and Mill Creek conservancies, as well as the Friends of Butte Creek. For Deer and Mill Creeks, cooperative agreements were worked out that allowed ranchers and lumber companies to continue to do business in a fish-friendly manner, while state and federal agencies similarly managed their own lands and waters. Protecting the Butte Creek spring Chinook has been perhaps even more contentious because the flows of the creek are partly used for hydropower production, with some flow augmentation from the Feather River (a plus for the salmon). The upcoming relicensing of DeSabla-Centerville Dam provides a timely opportunity to make critical changes to flow and temperature regimes on Butte Creek. Fortunately, the three key populations of CVS Chinook continue to be of great conservation interest to all parties involved in managing the watersheds and new protective actions continue to be taken.

In 1999, Pacific Gas and Electric Company, National Marine Fisheries Service, U.S. Fish and Wildlife Service, U.S. Bureau of Reclamation, and California Department of Fish and Game reached an agreement to restore salmon and steelhead in Battle Creek. The parties have each signed the detailed Memorandum of Understanding focused on restoring the winter Chinook, spring Chinook, and Central Valley steelhead although fall and late-fall Chinook salmon will also benefit. The restoration proposal includes: 1) increasing the minimum instream flows from the present amount of 3-5 cfs year round to approximately 35-88 cfs adjusted seasonally; 2) decommissioning five diversion dams and transferring their associated water rights to instream uses (Wildcat, Coleman, South, Lower Ripley Creek, and Soap Creek diversion dams); 3) screening and enlarging ladders at three diversion dams (Inskip, Eagle Canyon, and N. Battle Creek Feeder diversion dams); and 4) constructing new infrastructure (tailrace connectors) that will eliminate mixing of North and South Fork waters and significantly reduce redundant screening requirements. This project would open up an additional 42 miles of prime habitat for spring run Chinook that have been closed off by hydropower operations since the early 20th century (CDFG 2007). Funding on this project has been slow in coming, but in March of 2007, CDFG announced $67 million had been appropriated for the project. The project has been delayed somewhat by the immense process of transferring monies and conducting environmental review. However, at this time, the process is nearly complete and the restoration phase can begin (Mary Marshall, USBR, pers. comm., 2007). Projects like this should be developed for Butte Creek and the other remnant wild spring Chinook populations, given their importance for conserving life history and genetic diversity.

In 1948, virtually all water behind Friant Dam on the San Joaquin River was sent down the Friant-Kern and Madera canals, with a small release for riparian landowners immediately below the dam. CDFG officials attempted to rescue the 1948 run by trucking some 1,915 spring 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 and died. The spring Chinook of 1949 and 1950 met a similar fate and thus the run was extirpated, as was the companion run in the Kings River (Moyle 2002). In recent years there has been a considerable push to allow the San Joaquin River to once again support runs of spring Chinook salmon, and secondarily fall run Chinook. In September 2006, the U.S. Bureau of Reclamation, the Friant Water Users Authority, and the Natural Resources Defense Council reached a settlement to end 18 years of litigation over the dewatering and alteration of some 150 miles of the San Joaquin River from the base of Friant Dam downstream to its confluence with the Merced River. The settlement followed a court ruling that dewatering the river and driving the spring Chinook to extinction was an illegal action on the part of the state and must be remedied.

The settlement agreement for the San Joaquin River will provide minimum instream flows, enough to recreate a permanent flow of water all year round, plus additional water for migration, spawning, and rearing of Chinook salmon. In addition, there will be extensive habitat restoration, necessary after 50+ years of complete neglect and abuse of the channel. Restoring continuous flows to the approximately 150 miles of often dry and heavily altered river channel will take place in a series of phases. Planning, design work, and environmental reviews are slated to begin immediately, and interim flows for experimental purposes will start in 2009, with the goal of establishing a self-sustaining population of spring Chinook by 2025. According to the U.S. Bureau of Reclamation, the flows will be increased gradually over the next several years, with salmon being re-introduced by December 31, 2012 (Bureau of Reclamation 2006).

These actions are essential for keeping spring Chinook salmon from going extinct in the next 50 years. Climate change represents the next major conservation challenge for spring Chinook. Lindley et al. (2007) indicate that climate change models show a likely elimination of suitable habitat in much of the extant range. This means the Chinook will need to get higher in the watersheds than current infrastructure (dams) allows. Barrier removal or some kind of trap and truck operation will thus likely be a major part of spring Chinook conservation in the next century. Restoration of former habitat is critical to maintaining long-term population stability, particularly in the face of more challenging future climatic conditions. This makes enhancement of the Battle Creek population and restoration of the San Joaquin River population very important aspects of spring Chinook conservation because both have good sources of cold water and the San Joaquin in particular is distant from other populations.

Trends:

Short term: Recent CVS Chinook populations have been generally stable or increasing (with some interannual variability) and Lindley et al. (2007) indicated 11% growth for spring Chinook populations on Butte Creek, 18% population growth on Mill Creek, and 8% population growth on Deer Creek, although this growth took place in years of favorable freshwater and ocean conditions. Lower numbers were seen in 2006 and 2007. However, there have been several years of very poor survival of holding adults after daily mean water temperatures on Butte Creek in exceeded 21°C for more than 10 days in July, 2002 and 2003. In 2002, there was 20-30% adult mortality, and in 2003, 65% of the over-summering adults died, mostly due to columnaris, a bacterial infection often associated with poor water quality, high temperature, and other stresses (Lindley et al. 2007). In 2006 and 2007, numbers were low again, reflecting the general decline of Central Valley Chinook salmon.

Long term: CVS Chinook declined from once being as abundant as fall run Chinook salmon to a few hundred fish, which have barely been able to hold on. Thus, the trends indicate that their most likely long-term future in California is extinction. Climate change models seem to validate this view. Additionally, the present limited current distribution of spring Chinook makes them vulnerable to localized stochastic events (fire, volcanic eruption) in which the entire run can be jeopardized by a single incident. The seeming inevitability of extinction can be reversed if major conservation efforts are successful, starting with restoration of runs to the San Joaquin River and Battle Creek.

Status: 2. There is high likelihood of CVS Chinook going extinct in next 50-100 years (Table 2). Recent management efforts and protection have somewhat reduced their vulnerability to extinction but the probability of populations plummeting in the future are high. The analysis of Lindley et al. (2007) suggested CVS Chinook in Butte and Deer Creeks were at low risk of extinction in the short term, having recovered from record lows in the 1970s and 1980s. Mill Creek was determined to be at moderate risk of extinction due to its smaller population (Lindley et al. 2007). Lindley et al. (2007) indicated some uncertainty as to whether Mill and Deer Creek constituted a single population with intergenerational straying or were indeed two distinct populations as analyzed. If they are considered two stocks of the same population then their combined risk of extinction in the short run was categorized as low. However, (1) all three populations are in adjacent streams subject to natural and human-caused disasters; (2) populations have been extremely small in the recent past; and (3) all three streams are small and could become marginal for salmon with a few degrees rise in temperature due to climate change. These factors indicate strongly that rating CVS Chinook as vulnerable to extinction in their native range is appropriate. They are currently listed by both state and federal governments as Threatened.

Metric	Score	Justification
Area occupied	2	Found mainly in just three adjacent creeks.
Effective pop. Size	4	Populations in the three streams in recent years have had effective population sizes of 600- 6000, lower in other years.
Intervention dependence	3	Require continuous protection, monitoring etc to maintain populations.
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. There is always risk of inbreeding etc when populations decline during poor years. The Feather River population has hybridized with fall Chinook.
6 Climate change	1	Extremely vulnerable given small population sizes and range, as well as already high temperatures of streams.
Average	2.3	14/6
Certainty (1-4)	4	Well studied.

Table 2. Metrics for determining status of Central Valley spring Chinook salmon, where 1 is poor value and 5 is excellent.

SOUTHERN OREGON-NORTHERN CALIFORNIA COAST COHO SALMON
Oncorhynchus kisutch

Description: Spawning adult coho salmon are 55-80 cm FL (35-45 cm FL for jacks) and weigh 3-6 kg (Moyle 2002). 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 (Moyle 2002). 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 the Central California Coast (CCC) coho ESU 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 and presumably have adapted to the extreme conditions (for coho salmon) of many coastal streams. As discussed in Moyle (2002), 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 genetics of coho salmon in California are fairly well studied (CDFG 2004). The most recent, detailed genetic study of California coho salmon populations, using microsatellite DNA markers, is that of Bucklin et al. (2007) 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 has 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 the coho salmon in California was first documented in the classic studies on Waddell Creek by Shapavalov 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) 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 around basis (Table 1)

	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			xx	XX	XX	XX	xx					
Estuary rearing			xx	XX	XX	xx						

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

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," may 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 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).

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 general, in smaller coastal streams (such as Redwood Creek or the Mattole River) the timing of coho runs is determined by the first rain event which increases flow sufficiently to break bars at the mouth of estuaries, permitting access to the stream. Coho salmon migrate up and spawn mainly in streams that flow directly into the ocean or in tributaries of large rivers.

Females choose redd sites where the 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 may take about a week to complete and a female deposits 1,400-7,000 eggs, with bigger females producing more eggs. Both males and females die after spawning, although the female may guard a 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 can be as low as 10 percent; under adverse conditions such as high scouring flows or heavy siltation, mortality may be 100 percent. Upon emerging, the fry (30-35 mm TL) seek out shallow water, usually along the stream margins. In the Klamath River watershed, emergence of fry starts in mid-February and peaks in March and early April, although apparently fry have been found into July (CDFG, unpublished data). 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, the fish 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 the winter period, refuge from high, turbid flows are required for survival. Typically, these refuges are side channels, complex masses of large woody debris, and small, clear tributaries. Towards the end of March and the beginning of April, juvenile coho begin to migrate downstream and into the ocean. 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 related to rising or falling stream flows, size of fish, day length, water temperature, food densities, and dissolved oxygen levels. At this point, the outmigrants are about one year old and are 10-13 cm FL. The occasional larger fish (ca 20 cm FL) has usually spent two years rearing in the stream. In Prairie Creek (Humboldt Co.), over 20% of emigrating juvenile SONCC coho are 2 year olds (Bell and Duffy 2007). The fish emigrate 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.

After entering the ocean, immature salmon initially remain in inshore waters close to the parent stream. They gradually move northward, staying over the continental shelf. Coho salmon can range widely in the north Pacific, but the 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, pers. comm. 1994). Oceanic coho tend to school together. Although it is not known if the schools are of mixed origin, consisting of fish from a number of different streams, fish from different regions are found 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. The key to understanding why they choose a particular combination of habitat characteristics and how habitat affects growth and survival is bioenergetics (Box 1).

Adult coho salmon move upstream in response to the change in stream flows caused by fall storms, especially in small streams when water temperatures <16EC. 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. High turbidity may delay migration even if other conditions are right.

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 favored by 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 4.4-13.3EC, although eggs and alevins can be found in 4.4-21.0EC 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 the pools or runs. Optimal habitat seems to be pools containing rootwads and boulders in heavily shaded sections of stream, although warmer, more open conditions may be used if food is abundant. In winter, refuge habitat is needed to protect juveniles from being washed away by high flow events.

Juveniles require water temperatures not exceeding 22-25EC for extended periods of time and oxygen and food (invertebrates) levels that remain high. Preferred temperatures are 12-14EC, although juveniles have been found living at temperatures of 18-29°C (Bisson et al. 1988; Moyle 2002). Preferred water velocities for juveniles are .09-.46 m sec^{-1}, depending on habitat. High turbidity is detrimental to emergence, feeding and growth of young coho. Young and adult coho salmon are found over a wide range of substrates, from silt to bedrock.

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 never so constant and juvenile coho are often found at higher temperatures. In tributaries to the Mattole River, juvenile SONCC coho are absent from streams where the mean weekly maximum temperature exceeds 18°C for one week (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 possible because (1) the 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, a juvenile coho will grow if it ingests more energy than it consumes through activities such as searching for food or avoiding predators. It eventually dies if it ingests less energy than it uses for daily activities. Part of that energetic cost can be increased metabolic rates and stress caused by temperatures higher than the optimum. In the studies by Bisson et al. (1988), conditions were so good from a bioenergetic perspective that the coho were able to survive temperatures only slightly below the absolute lethal temperature and grow at temperatures normally considered to be too high. In Mattole River tributaries, where food is not abundant and predators and competitors are common, even moderately high temperatures become lethal if experienced on a regular basis. The energetic costs

Distribution: Coho salmon are widely distributed in the 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 the Soviet Union. 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). NMFS (Williams et al 2006) divided these California populations into five diversity strata, which each represented 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 (William 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.

CDFG (2002) updated the distribution and abundance analysis for California coho salmon by Brown and Moyle (1991) and Brown et al. (1994), from which this information comes.

Oregon: 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.

Smith River and Del Norte County streams: In the Smith River and smaller streams in the region, coho apparently still occupy only part of their historic range in small numbers. CDFG (2002) found them only in Mill and Rowdy Creeks.

Klamath River: Historically, coho were found throughout most of the ~4000 km^2 watershed, spawning and rearing primarily in cold-water tributaries. In the mainstem Klamath, they presumably were present roughly up to the mouth of Jenny Creek, about 335 rkm upstream and used all permanent tributaries for which their 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, and have been recorded in about two-thirds of the 32+ tributaries from which they were once known. A similar proportion is apparently true for the Salmon and Scott watersheds as well. In the Shasta River, upstream access is blocked by Dwinnell Dam and coho are absent from major tributaries. Below the dam, the principal cold-water tributary suitable for coho in summer is Big Springs Creek.

Trinity River: In the Trinity River and its forks, coho were presumably distributed well upstream of the present location of Lewiston Dam. Below the dam (about 175 km upstream from the mouth on the Klamath River), they were present in at least 30 tributary streams. There are recent records from all but six of the streams, as well as in the mainstem up to the Trinity Fish Hatchery. However, upwards of 90% of the coho in Trinity River are of hatchery origin, so the significance of their present distribution is questionable (Spence et al. 2005).

Redwood Creek: Redwood creek and its major tributary Prairie Creek were historically important coho streams, as were their 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 up stream (Madej et al. 2005).

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). They seem to be present in small numbers in about 70% of the historic smaller coastal streams, although Freshwater Creek and Elk River, and their tributaries, still support somewhat larger runs.

Eel River: In the 9500 km^2 Eel River system, coho formerly ascended the mainstem Eel and its forks, South Fork, Middle Fork, North Fork and Van Duzen and 69 tributaries of the

South Fork Eel, the lower mainstem Eel River, and the Van Duzen River. They are currently absent from the Middle and North fork drainages and from about 40% of the tributaries in which they once existed.

Mattole River: The Mattole River (watershed area, 787 km^2) and its 21 larger tributaries presumably all once supported coho salmon but today they are found in 9 of the 21 tributaries and largely absent from the main stem (Welsh et al. 2001).

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. Of 392 coastal and tributary streams that historically held SONCC coho salmon, coho have been detected in 57-61% in recent years depending on the year and who is doing the calculation (CDFG 2002). The percentage increases a bit when analysis is done using three-year increments (the coho brood year cycle). 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.

Abundance: Historical figures 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 and year to year variation in these estimates is uncertain, and so they must be viewed only as "order-of magnitude" approximations. 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, without exception, 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).

The Klamath and Trinity River populations presently are largely maintained by hatchery production. About 80% of the fish returning to Iron Gate hatchery 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). In the Trinity River 89-97% of returning coho are of hatchery origin, which means there is very little, if any, natural spawning. In both rivers, hatchery returns and wild populations fluctuate more or less in synchrony. Hatchery returns are highly variable among years (Figure 1). At Iron Gate Hatchery, for example, only 322 coho returned in 2006-2007, although returns of over 2500 adults have occurred in the past (average about 1500 fish) (Chesney 2007). 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.

NMFS (2007) regards Klamath Basin coho populations to be "depressed but stable."(p. 7), although largely dependent on fish of hatchery origin.

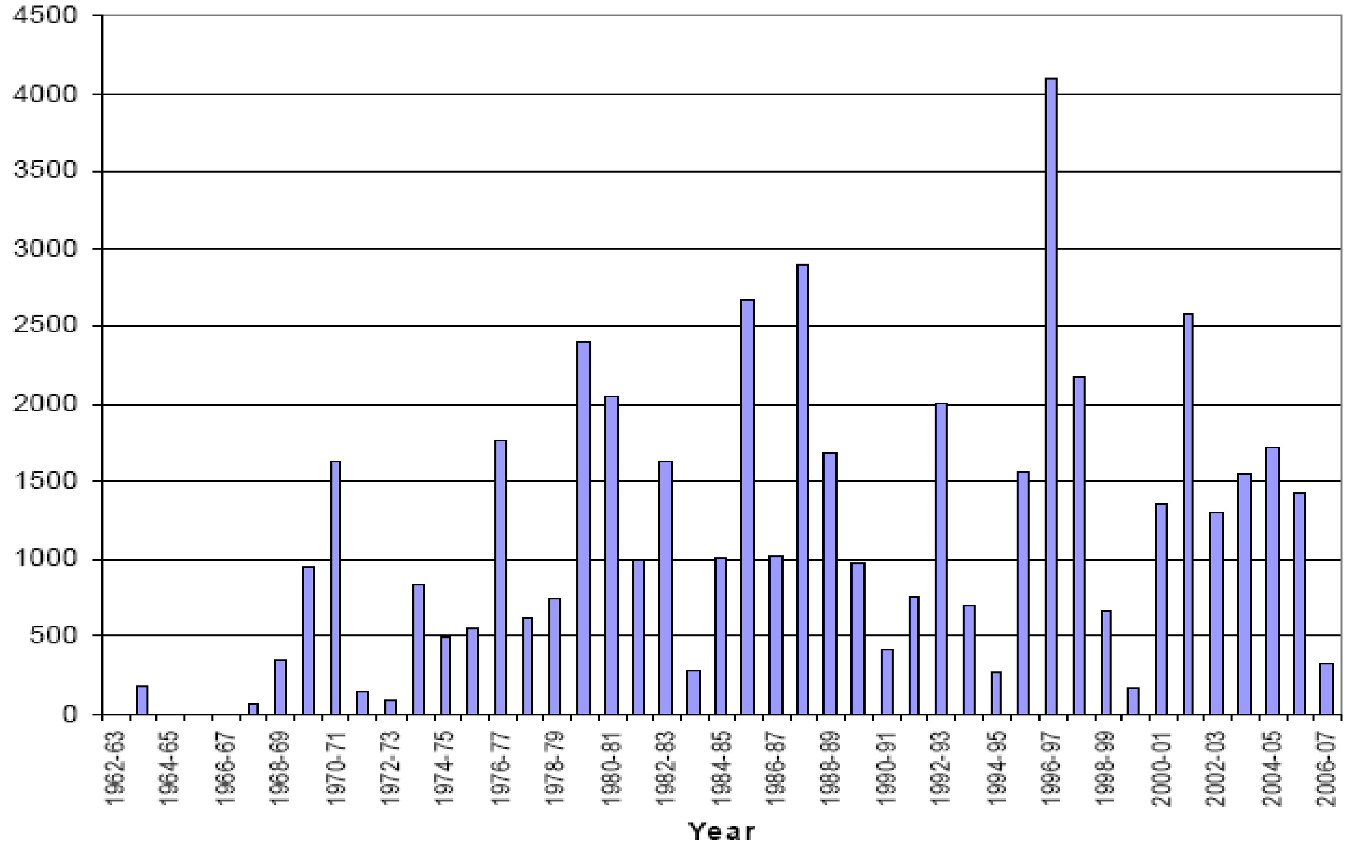


Figure 1. Returns of coho salmon to Iron Gate Hatchery, 1962-2007. From Chesney (2007).

The Shasta River is a Klamath Tributary that presumably once supported runs of several thousand fish each year, based on the presence of high-quality coldwater habitat in upstream areas, some now blocked by Dwinnell Dam, while other habitat has been made unsuitable by agricultural operations. In 2001, CDFG started seriously 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). Few juvenile coho can rear through the summer in much of the Shasta Valley reach, because of high temperatures (the result of irrigation), so it is likely that survival rates of wild-spawned juveniles are low, although the highly degraded Big Springs apparently still supports a few over-summering juveniles (C. Jeffres, pers. comm. 2007, 2008).

Probably the largest concentration of wild fish (with little or no hatchery influence) is in the South Fork of the Eel River, which has been estimated to have runs of about 1,300 fish. A 1990 survey, however, indicated a population one-half to one-third that size, with a downward trend. Numbers today are undoubtedly much smaller but surveys are lacking (Brown et al. 1994).

Brown et al. (1994) considered 5,000-7,000 fish to be a realistic assessment of the total number of naturally spawned adults returning to California streams each year in1987-1991 (80% SONCC coho). Presently, there are probably 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 reaffirming the threatened status of this ESU in 2005, NMFS indicated that the number of streams containing SONCC coho had stayed fairly steady since the estimates of Brown et al. (1994) (Spence et al. 2005) , suggesting that the number of returning fish on average was also about the same. Many of these fish are in populations of less than 100 individuals. 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. 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).

There is every reason to think that SONCC coho populations are not secure, even though hard data on numbers, especially in recent years, are surprisingly hard to come by. What evidence there is makes it seems likely that in most years, total SONCC adult coho spawners in California are somewhere between 3000 and 30,000 fish (not including 10,000 or so Rogue River fish of non-hatchery origin), probably on the lower end of the scale. The actual numbers are imprecise but that does not matter: what information exists indicates that SONCC coho salmon are at a tiny fraction of historical numbers, which are likely going down, and are highly vulnerable to continued environmental change. To make matters worse, these fish are mostly in about 250 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 threats to a species' survival may be categorized, according to the Endangered Species Act, as follows: "(A) the present, or threatened, destruction, modification, or curtailment of its habitat or range, (B) over-utilization for commercial, recreational, or educational purposes, (C) disease or predation, (D) inadequacy of existing regulatory mechanisms, or (E) other natural or manmade factors affecting its continued existence." For coho salmon all of these factors seem to apply. The general reasons for the decline of coho salmon in California are many and well known (Brown et al. 1994); 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. NMFS identified 16 factors limiting SONCC coho populations, which covers virtually every means by which humans damage streams and fish populations (see: http://swr.nmfs.noaa.gov/recovery/Coho_SONCCC.htm). CDFG (2002, 2004) provided extensive discussion of these factors and how they affect coho populations. Here we briefly discuss: (1) dams, (2) diversions, (3) logging, (4) grazing and agriculture, (5) mining, (6) estuarine alteration, (7) pollution, (8)alien species, (9) harvest, and (10) hatcheries, followed by a discussion on integrated effects.

Dams: Dams have two major general impacts on coho salmon: (1) they deny or reduce access to upstream areas and (2) 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 of coho 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 cold-water habitat upstream and the reservoir prevents cold water from reaches 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, imbedded 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.

Diversions: There are literally hundreds of small diversions on SONCC coho streams, which cumulatively can reduce flows and increase temperatures. If the diverted water is used for flood irrigation of pasture, much of it comes back into the river at high temperatures and polluted with animal waste and other nutrients or toxicants (e.g., the outflow of Big Springs on the Shasta River). 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. 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 stream flows drop and water quality becomes unsuitable for juvenile coho salmon (NMFS 2007). However, some tributaries upstream of diversions still support small coho populations. During dry years the mainstem often goes dry because of diversions, as do the lower reaches of most 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 biggest 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. Historic logging practices that have left a legacy of altered streambeds include the construction of splash dams. 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 out coho habitat and deprived the fish of essential cover in the form of fallen trees (large woody debris). 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 or otherwise rendering unsuitable a great deal of 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 decline of the SONCC coho indicates 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.

Grazing and agriculture: Grazing and other agricultural 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. See also estuarine alteration.

Mining: As is the case of logging, historic 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 to warm and shallow to support coho during the summer months today. Similar effects can be seen on other SONCC streams. Unfortunately, the rise in the price of gold in recent decades has seen a resurgence of instream mining, mostly though the use of small gasoline-powered vacuum dredges. This activity disturbs fish, turns over stream beds, and reduces water clarity when juvenile coho are most stressed because of natural conditions (e.g., warmer temperatures).

Estuarine alteration: Perhaps the least appreciated crucial habitat for juvenile salmonids, including coho salmon, is the estuary or lagoon at the river mouth. Juvenile coho may rear in an estuary for varying lengths of time and most are resident for at least a few weeks as they adjust to the shift from fresh to salt water. Consequently, estuaries with abundance 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.

Pollution: Many of the streams containing SONCC coho salmon are regarded as impaired under the Clean Water Act usually because of high sediment loads, although high 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. Sediment is often the legacy of past logging, road building, and other activities.

Alien species: Non-native predators are mainly a problem for coho salmon in the Eel River, where the out-migrants have to pass through large stretches of river infested in Sacramento pikeminnow, introduced in the 1980s. The effects of pikeminnow predation 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 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. Instream fisheries are small and only catch-and-release fishing is allowed. Small numbers of fish, however, are retained in the tribal harvest on the Klamath River. Overall, fisheries are having only a minor impact on coho populations today and their closure 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, usually met, of 500,000 volitionally released smolts per year. 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). According to CDFG (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 fish. Curiously, the mixed-origin fish that do spawn in the wild appear to contribute little to wild populations (Bucklin et al. 2007). 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 fish of hatchery origin, in ever-declining numbers.

Integrated effects: 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 poorly-timed 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. 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 fish effects. Streams were largely regarded as convenient ways to float logs to accessible locations (often behind a mill dam) so flash dams (see above) and log drives down the bigger rivers were commonplace. The second wave of damage was the result of post-World War II logging practices that reversed the partial recovery of the streams from past damage. Unrestricted logging using trucks and other heavy equipment caused massive erosion and removed riparian vegetation and woody debris from channels. As a result, stream temperatures increases, pools filled with silt and gravel, stream channels became altered, and water quality declined. SONCC coho streams are still suffering from this double legacy of harmful logging and although there has been impressive recovery of the landscape in many areas under better land management practices, the streams are still suffering and the coho are disappearing from them as a consequence. At the present time, populations are so low that even incidental fishing pressure on wild coho may prevent recovery, even in places where stream habitats are adequate. Existing regulatory mechanisms, such as forest practice rules, water agreements, and stream alteration agreements, have been inadequate to protect SONCC coho. Our relationship with the landscapes containing coho salmon clearly needs to be changed on a large scale if only to prevent extirpation, much less recover some semblance of their historical populations.

Because populations are so low, stream flows are so greatly altered, and watersheds are so damaged, coho salmon are exceptionally vulnerable to rapid climate change. Predicted effects on coho habitat include increases in stream temperatures, increased variability in flows (including reduced summer flows), and changed ocean conditions. 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.

Conservation: The key to stopping the decline of coho salmon is to protect their spawning and rearing streams, to restore damaged habitat, and to improve water quality. For example, if the Shasta River is to be restored as cold water spawning and rearing habitat for a significant population of coho, a key strategy will be increasing summer flows of cold water, as well as improving habitats. This in turn will require (among other things) removing (or developing passage over) Dwinnell Dam, recapturing the flows of Big Springs for fish, keeping livestock away from the river, and improving flows and habitat in tributaries such as the Little Shasta River and Parks Creek by reducing the amount of water removed for irrigation.

Improving conditions for coho salmon is a difficult task because it means modifying logging, farming, and road construction activities in dozens of coastal drainages and implementing habitat restoration plans along hundreds of miles of streams. In many streams it means that major reconstruction 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. CDFG reports (2002, 2004) 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, buying water rights, changing forest practice rules, and other major actions. Other recommendations for the Klamath basin are provided by NMFS (2007). Projects related to SONCC coho salmon are listed in http://swr.nmfs.noaa.gov/recovery/SONCC.htm.

Serious consideration should be given to eliminating or greatly reducing all production hatchery programs, especially those that rely on non-native stocks. This would reduce the effects of interbreeding of hatchery coho with wild coho, and reduce the spread of hatchery diseases to wild fish. Where population augmentation is deemed necessary, small-scale, on-stream hatchery operations using local wild stock could be used as temporary measures (but must be used with extreme caution, with a firm closure dates). At the very least a thorough investigation of the effects of hatchery-reared coho salmon on wild populations should be conducted.

Management actions put forward by CDFG (2004) could go a long ways towards reversing the trends if properly implemented, but that will require hugely increased effort involving increased funding, considerable interagency cooperation, and development of an extensive monitoring program. Monitoring the populations is a necessity; spawning streams should be identified and populations should be sampled annually. This would allow population trends to be followed and provide focus for restoration efforts. The challenges of managing such a diffuse resource as coho salmon are considerable, but if the population declines are not reversed soon, SONCC coho salmon are likely to disappear from California.

Trends:

Short term: For the past 10-20 years, SONCC coho salmon have remained at low populations. Monitoring is inadequate to say that the populations have definitely decreased, but they certainly have not increased significantly. The findings of Bucklin et al. (2007) suggest that most SONCC coho populations are in a state of collapse from which recovery will be difficult.

Long term: 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, 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%. This trend is most likely continuing, so extirpation of wild SONCC coho from California seems likely in 50-100 years or less.

CDFG (2004) takes a more optimistic view of SONCC trends. "Coho salmon are now found in less than 60% of the SONCC coho ESU streams that were historical coho salmon streams. However, these 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)."

Status: 2. Vulnerable to extinction within next 100 years (Table 2). This score is conservative, given the apparent rapid declines of most populations and the probable 95% plus decline from 50-60 years ago. Present trends suggest that most or all populations in small coastal streams will disappear in next 25-50 years without serious intervention. SONCC coho are listed as Threatened by both state and federal governments. The federal status was reaffirmed in 2005.

Metric	Score	Justification
Area occupied	2	Populations mainly in California, some in Oregon
Effective population size	3	Most populations are isolated and function independently and are <100 fish. This score is for the largest populations (Klamath, Eel).
Intervention dependence	3	All populations require 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	Vulnerable in all watersheds
Average	1.8	11/6
Certainty (1-4)	4	Fairly well studied populations

Table 2. Metrics for determining the status of SONCC coho salmon in California, where 1 is a poor value and 5 is excellent.

CENTRAL CALIFORNIA COAST COHO SALMON
Oncorhynchus kisutch

Description: Central California Coast (CCC) coho salmon are morphologically similar to coho salmon in the Southern Oregon-Northern California Coast (SONCC) ESU. Coho in the two ESUs can only be distinguished by genetic means.

Taxonomic Relationships: CCC coho salmon are highly adapted to local environments within the southern edge of the species distribution. Bucklin et al. (2007) showed that each population in each 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 along the Mendocino coast to the Golden Gate generally appearing more closely related. Populations further south did not fit this pattern (Good et al. 2005), presumably because dispersal among these basins is more pervasive than between this set of populations and the ones further north, a problem enhanced by extirpation of populations from streams between Santa Cruz County and Marin County. There has also been some dispersal by humans such as the movement of coho from Scott Creek to Waddell and Gazos Creeks (B. Spence, pers. comm. 2008). Bucklin et al. (2007) confirmed that widespread planting of coho from outside stocks in the past has had minimal influence on the genetics of local populations within this ESU. This genetic and geographic pattern was also observed at the southern edge of the steelhead range (see south-central coastal and southern steelhead descriptions).

Life History: The first comprehensive life history study of coho salmon was done on fish of the CCC coho ESU, 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) and CCDFG (2002) 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. Jerry J. Smith of San Jose State University has continued the life history and monitoring studies of Shapovalov and Taft in recent years (e.g., Smith 2006).

Habitat Requirements: Habitat requirements of CCC coho are basically the same as those SONCC coho, the summary of which is based on Moyle (2002) and CDFG (2002).

Distribution: For broad aspects of coho distribution see the SONCC coho account. CCC coho were historically native to California coastal streams from Punta Gorda down to the San Lorenzo River (Moyle 2002, Spence et al. 2005, Adams et al. 2007), as well as some streams tributary to San Francisco Bay (Leidy et al. 2005). It is also likely that a small run also existed in the Sacramento River (Brown et al. 1994).

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) and CDFG (2002), from which this information comes.

Mendocino County streams: At least 200 streams in Mendocino County once contained coho salmon, including most permanent tributaries to the Ten Mile, Noyo, Big, and Navarro Rivers. Recent surveys indicate that today 62% of them have retained at least small runs. In rivers such as the Navarro, coho are largely confined to small areas near the coast where streams are still cool (e.g., North Fork Navarro and tributaries).

Sonoma County streams: In Sonoma County, CCC coho salmon historically were observed in about 70 streams, most of them tributary to the Russian and Gualala Rivers (Spence et al. 2005). In recent years, coho salmon have been observed in just five of these streams, and in only one (Green Valley Creek, tributary to the Russian River) are they still observed in most years.

Marin County streams: There are historical records of coho salmon from at least 31 small coastal streams in Marin County. Coho have recently been observed in 17 (55%) of these streams, most of these tributaries to Lagunitas and Redwood Creeks.

San Francisco Bay streams and Sacramento River: Leidy et al. (2005) documented historical presence of coho in only four San Francisco Bay streams, although 11 others may have had them at one time. Likewise, Brown et al. (1994) thought there was enough evidence to conclude that there was once a small run of coho salmon up the Sacramento River, perhaps into the McCloud River. In any case, all of these populations are extirpated.

Streams south of San Francisco Bay: Coho salmon were historically found in 17 streams, as far south as the San Lorenzo River, Santa Cruz County, close to the end of coastal redwood forests and the EPA's West Coast Forest Ecoregion (Adams et al. 2007). They may have once been found a bit further south in Aptos and Soquel Creeks, also in the redwood zone, but presumably the populations were extirpated by logging before anyone was really looking for them. Today they are confined to Waddell, Scott, and Gazos Creeks with runs supported in part by a conservation hatchery, although Good et al. (2005) reported some occurrences in 8 of 12 streams surveyed in the 1999-2001 brood cycle. More recently, snorkel surveys conducted in 2006 and 2007 in randomly selected stream reaches (constituting about 13% of the accessible coho habitat in Santa Cruz and San Mateo counties) detected juvenile coho salmon at only two sites: one in Scott Creek and the other in San Vicente Creek (both in 2006). These studies indicate that, with the exception of Scott Creek, coho salmon are extirpated or nearly so from streams south of San Francisco Bay in two of three brood years (B. Spence, NOAA Fisheries, unpublished data).

As the above summaries indicates, CCC coho salmon were more or less continuously distributed in coastal streams from Mendocino County south to the San Lorenzo River in Santa Cruz County, with extensive inland distributions in the larger streams, especially the Navarro, Russian, and, probably, Sacramento Rivers. 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. In recent years, coho have been detected in about 40-48% of 328 coastal and tributary streams that historically held CCC coho salmon, depending on the year and who is doing the calculation (CDFG 2001, Good et al. 2005). Many of the occurrences, however, are single records from years (e.g., 2001) with strong brood classes.

Abundance: Historical abundance of coho in California overall is discussed in the SONCC account. Brown et al. (1994) considered 5,000-7,000 fish to be a realistic assessment of the total number of naturally spawned adults returning to California streams each year in 1987-1991 (20% of which were CCC coho, or ca.1,000 to 1,400 spawners). Presently, there are probably somewhere between 500 and 3,000 wild coho salmon spawning in the CCC region each year, but this number should vary with cohort and with variation in annual survival in both stream and ocean. A significant proportion of these fish are found in just one stream system, Lagunitas, Olema, and San Geronimo Creek in Marin County. From 1997-98 through 2004-05, Ettlinger et al. (2005) recorded between about 175 and 625 coho in the combined streams and their smaller tributaries, with no real trends observed since counts made in Lagunitas Creek in the early 1980s. If it assumed that each redd represents 2-4 spawners, the number of coho ascending these streams each year ranges between 350 and 2500 fish/year, numbers consistent with adult counts made in 2003-04 (949) and 2004-05 (1830). However, redd counts (182) in 2007-08 were the lowest in 12 years, giving rise to some concern about the status of the population (Salmon Protection And Watershed Network, unpublished data, 2008).

In the Noyo River, reasonably good records have been kept since the 1960s, although until the 1990s counts were incomplete (Grass 2008). These numbers indicate that prior to the late 1970s, even the incomplete accounts ranged between 1200 and 5000 spawners. Since 1990, most counts have been <500 fish, with 79 fish in 2005-2006 and 59 in 2006-2007 (Grass 2008).

In the Russian River, the last coho are being reared in the conservation hatchery at Dry Creek, while in Scott and Waddell Creeks returns are enhanced by a conservation hatchery (Smith 2006).

In reaffirming the endangered status of this ESU in 2005, NMFS stated:

> "Coho salmon populations continue to be depressed relative to historical numbers, and strong indications show that breeding groups have been lost from a significant percentage of streams within their historical range. A number of coho populations…appear to be either extinct or nearly so, including those in the Gualala, Garcia, and Russian Rivers, as well as smaller coastal streams in and south of San Francisco Bay (Good et al. 2005, p. 380)."

Almost all of the remaining streams have coho populations of fewer than 100 individuals during strong cohort years. 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 CCC coho populations are nearing extinction, with the possible exception of the population in the Lagunitas Creek drainage. Hard data on numbers, especially in recent years, is surprisingly difficult to come by. However, the actual numbers do not matter much: what information exists indicates that CCC coho salmon live in a tiny fraction of historical habitat with numbers to match. To make maters worse, these fish are mostly in isolated populations that show evidence of genetic and demographic problems that increase the likelihood of extinction (Bucklin et al. 2007).

Factors affecting status: The same factors that affect SONCC coho populations affect CCC coho populations only more so. A major difference, however, is that many of the heavily logged watersheds have not returned to forest, but have been urbanized or 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, the watershed is largely incapable of supporting *any* salmonids, much less coho salmon (Viers et al. in press). 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. The Russian River also has two major dams on it that have drastically altered its flow regime and denied access to upstream areas. Another growing problem is urbanization, which has eliminated populations in the San Francisco Bay region and is increasingly contributing to the loss of CCC habitat in streams elsewhere. It is ironic that the one population that is at least not crashing, Lagunitas Creek, is maintained in part by cold water released from a dam, combined with some watershed protection.

As indicated in the SONCC review, the effects of anthropogenic change on coho are particularly severe because virtually all females are three years old. Therefore, well-timed flood or severe drought, when acting on a severely depleted population, can eliminate one or more year classes from a stream. 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.

In CCC coho streams, the most severe damage was done by a long legacy of logging, starting in the 19th century, that 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 often crib dams were common on the larger streams. Crib dams impounded water upstream so that logs could be floated downstream, or so that water could be released to flush logs that had been dragged into the channel below the dams. Often streams would have multiple crib or splash dams on them and they were frequently left in place 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, NMFS, pers. comm.). 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 have been the likely coho habitat, resulting in early extirpation from the river.

It is hard to overestimate the importance of loss of large woody debris as the result of historical logging practices. 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, absolutely essential for providing refuge during high flow events in winter, because there were fewer opportunities for 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 largely present as the result of 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 and quantity for coho salmon (and other salmonids), as well as to degrade habitat. Thus Opperman et al. (2005) found that in the Russian River watershed, the pervasive large-scale changes land use had resulted in many former spawning areas being too highly imbedded in sediment to allow successful spawning. This is just one demonstration of how many CCC coho streams were never given a chance to recover because of the conversion of watersheds to farms, suburbs, and towns. CCC coho are disappearing rapidly as a consequence.

Conservation: 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 even keeping the ESU from extinction will require special, high energy/cost efforts, some of which are underway.

- Protect the few watersheds that have the potential to support coho in the future, such as Scott and Waddell Creeks and the Garcia, Noyo, and Gualala Rivers. They require not only protection from further degradation, but large-scale restoration efforts.
- Develop and maintain restoration hatcheries where they can be used in conjunction with habitat improvement and evaluation measures. Studies to improve rearing of wild coho by CDFG and NMFS in the Dry Creek Hatchery on the Russian River (Don Clausen, Captive Broodstock Program) should be expanded to increase reintroduction efforts in the watershed. Other efforts underway include the Scott Creek Captive Broodstock Program, and the Scott Creek/Kingfisher Flat Conservation Program, So far, these programs do not appear to have altered the genetics of local populations (NMFS, Federal Register 70 (123): 37176, June 28, 2005.). However, more monitoring is needed of genetic and demographic effects on both source and receiving populations.
- Resolve the complex water allocation issues in the watersheds to make sure adequate water is left in the streams to support coho salmon.
- 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 CCDFG (2002) and NMFS (2006, http://swr.nmfs.noaa.gov/recovery/Coho_NCCC1.htm) 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. Monitoring the populations is a necessity; spawning streams should be identified and populations should be sampled annually.

Trends

Short term: In the past 10 years, coho salmon have remained at low populations, although numbers in 2007-08 seem to have been exceptionally low. There is inadequate monitoring to say the populations have definitely decreased, but they certainly have not increased significantly. 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.

Long term: 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 even then in very small numbers. Additionally, coho salmon appear to be extirpated from, or nearly so, several large watersheds including the Garcia, Gualala, and Russian Rivers, as well as from streams within and south of San Francisco Bay, leading to increased isolation of extant populations. These trends are most likely continuing, so extirpation of wild CCC coho from California seems likely within 50 years or less.

Status: 1. Highly vulnerable to extinction within next 50 years. This score is the result of the precarious state of all populations and the probable 95% plus decline from 50-60 years ago. Present trends suggest that most or all populations in small coastal streams will disappear in next 25-50 years without increased intervention and protection of watersheds. NMFS (Good et al. 2005) and CDFG (2002) 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. The federal status was reaffirmed in 2005.

Metric	Score	Justification
Area occupied	2	Populations only in California
Effective population size	2	All populations are small, isolated, and function independently. Most are <50 in most years.
Intervention dependence	2	All population require intervention to persist and most have intensive management in place or proposed.
Tolerance	1	Coho are among the most sensitive salmonids to environmental conditions and CCC coho face adverse conditions.
Genetic risk	1	See Bucklin et al. (2007)
Climate change	1	At southern end of range so exceptionally vulnerable.
Average	1.5	9/6
Certainty (1-4)	4	Well documented.

Table 1. Metrics for determining the status of CCC coho salmon, where 1 is poor value and 5 is excellent.

PINK SALMON
Oncorhynchus gorbuscha (Walbaum)[6]

Description: Pink salmon are the smallest of the Pacific salmon, usually reaching 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 both caudal lobes and back. The number of gill rakers, which ranges from 16-21 on the lower, first gill arch, is also distinctive (McPhail and Lindsey 1970). The mouth is terminal and there are sharp teeth on both jaws, the 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 rays. There are 147-198 scales along the lateral line. Branchiostegal rays number from 10-15 on either side of the jaw.

Marine-phase fish are steel blue to blue-green dorsally, are white ventrally, and have silver sides. The back and upper parts of the lateral surfaces have large black spots which are also present on the adipose and caudal fin lobes (Scott and Crossman 1973). Spawning males have a pronounced hump immediately behind the head (the reason for their other common name, humpback salmon) and the snout is greatly enlarged and hooked. The body color becomes darker, especially on the head and back. The sides become pale red, with brown to olive-green markings. 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 reproductive pink salmon become deeply embedded. Juveniles in fresh water are small (<40 mm) and lack parr marks.

Taxonomic Relationships: This species was first described in 1792 (see Scott and Crossman 1973, for complete synonymy). Nothing is known about the genetic identities of California fish or how they relate to more northern populations. However, biochemical differences have been observed between pink salmon stocks in different river systems (Hard et al. 1996) and Russian workers also have noted genetic differences between stocks in different geographical areas (Omel'chenko and Vyalova 1990). Hard et al. (1996) indicate that the southernmost populations in Washington are in Puget Sound, and, with one exception, only have spawning runs on odd years. These odd-year fish are regarded by NMFS as a distinct ESU that is in no danger of extinction. It 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 only been recorded in October (Fry 1967, C. Bell, pers. comm.2003). Most pink salmon spawn in the intertidal or lower reaches of streams and river, although upstream migrations of 100-700 km are found in some northern river systems. Spawning occurs in gravelly riffles with water depths between 20-60 cm. The six redds built by females in the lower Russian River were all situated along the stream edges where the substrate was finer (Fry 1967). No redds were found in the middle portion of the riffle where the

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[6] Modified and updated from Moyle et al. 1995. Fish Species of Special Concern in California, 2nd edition. Sacramento, Calif. Dept of Fish and Game.

substrate was composed of coarser gravel. During nest building, the female lies on her side and excavates a depression approximately 90 cm long and 45 cm deep. The female indicates spawning readiness by sinking down into her redd until her anal fin touches the gravel. The male then swims up alongside and both fish quiver and gape as they release gametes. Once egg deposition is completed, the female covers the redd with gravel by displacing substrate from the upstream margin of the redd. Females may spawn with several males; the nest area is typically defended by a large dominant male and several smaller, subordinate males. Likewise, a single male will spawn with several females.

A female usually lays 1,200-1,900 eggs during the spawning period, which lasts for several days. Both males and females die a few days to a few weeks after spawning. Embryos hatch after 4-6 months of incubation, presumably in February and March in California. The alevins emerge from the gravel in April or May, at which time the yolk-sac has been absorbed. The fry are about 35 mm TL and immediately begin to migrate downstream into the estuary. Juvenile migration takes place at night and fish move rapidly downstream, usually reaching the estuary in one night. Once in the estuary they form large schools and remain in the inshore areas for several months before moving out to sea. Most juveniles do not remain in fresh water long enough to feed, although those that hatch from redds further upstream have been known to feed on aquatic insects. At sea, juveniles feed on small crustaceans and other invertebrates. Maturing adults feed mostly on fish, squid, euphausid shrimp, amphipods, and copepods.

Pink salmon wander great distances while in the oceans and tagged fish have been captured 2,700 km (1,700 mi) from where they were tagged (Omel'chenko and Vyalova 1990). However, they generally return to their natal streams for spawning. The discrete two-year life span of pink salmon results in distinctive populations, which form odd- and even-year spawning runs. Some streams may support major runs of both (odd and even) years whereas others may support major runs of one or the other year. Historically, the southernmost pink salmon fisheries in North America landed large numbers only in odd-numbered years, and in California most records of pink salmon are 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 current velocities of 30-140 cm/sec over the redds (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. Given that 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 distance are short. Rearing temperatures are likely to be similar to incubation temperatures (Heard 1991).

Distribution: Spawning pink salmon ascend coastal streams of northern Asia, from Korea through Japan to Siberia (Heard 1991). Along the northwestern Pacific coast of North America they range from the MacKenzie River in the Yukon Territory (Canada) south to California coastal rivers. 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). Pink salmon are apparently absent from Oregon streams.

In California, small numbers have been reported from the San Lorenzo River (Scofield

1916), the Sacramento River and tributaries (US Commission on Fish and Fisheries 1891; Hallock and Fry 1967), the Klamath River (Snyder 1931), the Russian, Garcia, and Ten Mile Rivers (Taft 1938) and Redwood Creek (Sparkman 2005). Occasional fish have also been reported from the Mad River and Prairie Creek, Humboldt County (Taft 1938, Smedley 1952), Lagunitas Creek at the south end of Tomales Bay (B. Cox, CDFG, pers. comm.), and from Mill Creek, Tehama County (Taft 1938). A pink salmon caught in the Mad River also was reported in the popular press (Arcata Union, Sept. 6, 1928; S. Van Kirk, pers. comm.), which stated that this species had been frequently taken in the Mad River by net fishermen many years earlier. Pink salmon have been observed spawning in the Ten Mile and Garcia Rivers at various times (Taft 1938). Occurrence of spawning in some Mendocino County streams was reported by Roedel (1953). In the lower Russian River, Fry (1967) observed at least six pink salmon redds in 1955; pink salmon were apparently present in other years in that period and small numbers were observed in 2003 (Chase et al. 2005). The most consistent occurrences seem to have been in odd years in the lower Garcia River; in 2003, 23 pink salmon redds were documented in an incomplete survey. However, Sparkman (2005) captured small numbers of juvenile pink salmon in outmigrant traps in Redwood Creek in 2000, 2002, 2004, as well as 2005, suggesting spawning was taking place in both even and odd years.

During the 1800s, pink salmon were reported to occur 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 taken in the Sacramento River system; this included 12 fish from Coleman National Fish Hatchery, 4 in Mill Creek and 3 at Nimbus Fish Hatchery on the American River (Hallock and Fry 1967). Recent occurrences of pink salmon have been infrequent. One pink was seen in the American River by T. Mills (CDFG, pers. comm.1995) and 3 more (males) were taken on that river on three separate occasions (R. Ducey, pers. comm.1995). Regardless of the limited sightings in the Central Valley, spawning does occur on occasion in the Sacramento-San Joaquin River system. Thus, seven juvenile pink salmon were captured at the state J.E. Skinner Fish Protective Facility near Tracy in March, 1990 (D. McEwan, CDFG, pers. comm., 1990).

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 they have never been common here and present only in odd years. However, given that pink salmon spawn in the lower reaches of streams in October, when few observers are likely to be present, and 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). Taft (1938) cited reports by CDFG wardens that considerable numbers of pink salmon were running in northern California streams in 1937: "many quite large schools of them" in the Ten Mile River, and "several hundreds" in the Garcia River, "spawning all over from the Red Bridge to the western boundary of the Indian Reservation, a distance of about two miles." They also were observed in the Russian River during that year (Taft 1938). Their occurrence in the Russian River in 1937 and evidence of limited spawning in 1955 (Fry 1967), would indicate that this "run" may have been the southernmost one for the species, except for occasional spawners in the Sacramento River. How regular spawning in the Russian River has been is questionable, although a few have been observed in recent years as well. On the other hand, the Garcia River has had pink salmon recorded from it surprisingly often, including a number of spawning fish in 2003, suggesting that spawning may still be occurring in odd years. Likewise, Redwood Creek and its tributary, Prairie Creek, they have been observed in four different recent years with at least one older record as well. Overall, it seems highly likely that pink salmon were once common enough in California to support small runs in several rivers.

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 there is a viable population in California still, as defined by McElhany et al. (2000). If pink salmon were historically a species that occurred in California mainly as a 'sink' population from sources further north, then its 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, as seems likely, did once have self-sustaining populations in California, 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 in California as the result of logging, gravel mining and other human activities. This also makes them very hard to observe.

Conservation: The first step in a management plan is to determine if reproducing populations exist anywhere in California. The lower reaches of the Ten Mile, Garcia and Russian Rivers, as well as Redwood and Prairier Creeks, should be thoroughly surveyed at the appropriate time of year (mid-September through November) and recent records elsewhere in the state carefully investigated. If viable spawning populations exist, then habitat, flow, and water quality should be protected.

Trends:

Short term: Assuming there are regular spawning populations, their small size and odd year occurrence suggests high vulnerability to extirpation, even in the short run.

Long term: Persistence of pink salmon in California seems unlikely without artificial propagation to enhance whatever populations exist. 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.

Status: 1. Pink salmon are considered by Moyle (2002) and Augerot and Foley (2005) as extirpated from California, except for occasional strays. However, reports of a spawning run in the Garcia River and the presence of juveniles in multiple years in the Redwood Creek drainage suggest that small populations may still exist and have been overlooked. It is highly likely they will disappear completely from California streams in the reasonable future, although it is possible that populations have periodically gone extinct and then become re-established when pink salmon are abundant in more northern waters.

Metric	Score	Justification
1 Area occupied	1	Only confirmed from Garcia River and Redwood creek.
2 Effective pop. Size	2	Numbers very uncertain, so this is a best guess.
3 Intervention dependence	3	Largely unstudied, but some intervention needed if this species is to persist.
4 Tolerance	1	Short life cycle, dependent on 1-2 streams.
5 Genetic risk	1	If a local population, then risk is high
6 Climate change	1	Garcia watershed has been highly impacted by logging; spawning areas unprotected.
Average	1.5	9/6
Certainty (1-4)	2	Very limited documentation

Table 1. Metrics for determining the status of pink salmon, where 1 is poor value and 5 is excellent.

CHUM SALMON
Oncorhynchus keta

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 a strong homing tendency (Salo 1991) which contributes to the genetic isolation of spawners in different streams. No genetic studies on chum salmon are available for California fish, so their relationship to more northern populations is not known. However, populations in Oregon and Washington are considered part of the "loosely defined" Pacific Coast ESU (Johnson et al. 1997, p. 105), therefore California fish presumably also belong to this ESU.

Life History: Because of their economic importance, their life history, wide distribution, and habitat requirements chum salmon have been well studied but mainly 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 partly results in most chum salmon spawning within 200 km of the ocean and some populations spawn in the intertidal reaches of streams. Chums 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 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). Females dig sequential redds which the female guards until she dies. Males, which are sexually active for 10-14 days, spawn with multiple females. Large females can 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 and move into estuaries soon after emerging from the gravel. They may remain in their estuary, however, for several months before moving out into more oceanic waters. Migration of fry is mainly nocturnal, unless turbidities are high.

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).

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 gravels that range from 1-10 cm in diameter but optimal sizes seem 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 sea 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).

Distribution: Chum salmon have been recorded spawning in streams in Korea north along the Arctic coast of Russia, and from the Mackenzie River on the Canadian Arctic coast of North America southward into central California. They have been caught in the ocean as far south as San Diego, but the southernmost freshwater record has been the San Lorenzo River, Santa Cruz County (Moyle 2002). Historically, they were reported to be present in most streams north of San Francisco Bay, although the evidence was anecdotal. At present, they become progressively less common in southern streams within their historic range but they are still present in small numbers in some Oregon streams, as well as in California (Moyle 2002).

In California, chum salmon are commonly taken in the commercial salmon fishery but records of their regular occurrence in fresh water are sporadic. 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 and 2005, juveniles were collected from the lower Napa River during a fish monitoring program (Leidy 2007).

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. A few chum salmon also 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, pers. comm. 1995). 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.

Monitoring of Mill Creek, a tributary to the Smith River estuary, by J. Waldvogel (2006) suggests that chum salmon spawn there 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. Chum salmon are also observed on an irregular basis in other coastal streams, such as Redwood and Lagunitas creeks in Marin County, although they are easy to overlook. When regular surveys of spawning salmon very made on Lagunitas Creek for four years, chum salmon were observed every year, including individuals on redds (Ettlinger et al. 2005).

Abundance: Chum salmon are abundant from Washington on north, with some runs supported by hatchery production (Johnson et al. 1997). In California they are rare and have probably always been uncommon. There is evidence of spawning in the South Fork Trinity. In the period 1985-1990, between 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, USFWS, 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 1980-2002, entering the stream during early to mid-December,when high stream flows, a period when Chinook salmon were also entering (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 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). But subsequent records have been spotty (Moyle 2002) and they are rarely seen in salmon surveys. Curiously, chum salmon juveniles were found in 2006 in the Napa River, indicating successful spawning (Martin 2007).

Overall, it appears chum salmon at least sporadically in streams from San Francisco Bay north to the Oregon border. The evidence suggests, however, that the only California rivers that currently are used by chum salmon for spawning on a regular basis are the South Fork Trinity, Klamath and Smith rivers, although the numbers of fish in each river is small and they may not be present every year. It is highly likely that chum salmon were more widely distributed in the past.

Factors affecting status: The historic rarity of chum salmon in California makes it difficult to identify factors that may 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. If California populations are largely driven by fish ‘straying’ from more northern populations, 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. It is also possible, however, that California streams have maintained small populations of chum salmon continuously but they have largely been overlooked because they tend to spawn close to coast and do not remain long in fresh water as juveniles.

Conservation: Surveys in the South Fork Trinity, Klamath, and Smith rivers should be continued to monitor the status of the few fish spawning there. The exact timing and place of spawning need to be determined. Suitable habitat, flow, and water quality should be maintained in order 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. 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). 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.'

Trends: Chum salmon abundance has always been small, few observers are aware of them, and juveniles are easy to overlook, so there is no real trend data available on chum salmon. It is reasonable to think, however, that they 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.

Status: 1. Johnson et al. (1997, p 164) reported chum salmon as being extinct in California and all populations in Oregon are regarded as "depressed or extinct." We think there is enough evidence to indicate that at least three very small self-sustaining populations (in Smith, Klamath, and Trinity rivers) still exist in the state, which are all threatened with extinction. However, given the paucity of data, the certainty of this status designation is low (Table 1). The alternative, however, is to admit they are extinct in the state as a viable species with California populations depending entirely on fish from elsewhere. In this case, spawning in California streams would take place mainly when populations are high in the ocean. 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.

Metric	Score	Justification
Area occupied	2	If chum salmon are still maintaining populations, there are several (Smith, Trinity, Klamath rivers).
Effective pop. Size	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 and it is likely that without intervention, the species will soon be extirpated.
Tolerance	2	Southern populations of chum salmon seem to have fairly narrow spawning habitat requirements and their young require functioning estuarine habitats for rearing.
Genetic risk	1	California populations are extremely small and vulnerable to inbreeding depression and other genetic problems. This is not an issue if the populations are maintained by 'strays' from northern populations.
Climate change	1	Even small changes in flows or temperatures and/or small changes in ocean conditions could eliminate the populations
Average	1.5	9/12
Certainty (1-4)	1	Information is very limited.

Table 1. Metrics for determining the status of chum salmon, where 1 is poor value and 5 is excellent.

CALIFORNIA GOLDEN TROUT
Oncorhynchus mykiss aguabonita

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 Golden Trout Creek 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 given 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 South Fork Kern River, *S. whitei* from the Little Kern River, and *S. roosevelti* from Golden Trout Creek. 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 (but see discussion in Little Kern golden 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 the low productivity of the high elevation streams of their native range (Knapp and Dudley 1990, Knapp and Matthews 1996). In streams, they are usually 3-4 cm 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, and 21-28 cm 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 growth of golden trout in lakes are suspect because the populations were established from introductions and hybridization with rainbow trout is common.

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. They spawn in gravel riffles in streams. Spawning behavior is typical of other members of the rainbow trout group although they spawn successfully in finer substrates (decomposed granite) more than most other trout (Knapp and Vredenburg 1996). Females produce 300-2,300 eggs, the number depending on body size (Curtis 1934). Embryos hatch within 20 days at an incubation temperature of 14°C. The 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 the 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 (Chronomidae) (T. Armstrong, unpublished data). Although the bright coloration makes them highly visible, there are very few natural predators in the range occupied by this subspecies (Moyle 2002). Their tendency to be more active during the day than most trout also suggests low predation. Thus, the 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 (Needham and Gard 1959). When these trout are removed from the mountainous streams and brought down to low elevation streams, they may lose the 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 unglaciated valleys of the Kern Plateau are broad, flat, and filled with glacial alluvium, which results in wide meadows through which the streams meander. The streams are small, shallow, and have only limited riparian vegetation along the edges. The exposed nature of the streams is largely the result of heavy grazing of livestock on a fragile landscape, which began in the 1860s, causing compaction of soils, collapse of stream banks, and elimination of riparian plant cover (Odion et al. 1988, Knapp and Matthews 1996, Matthews 1996b). The 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).

Distribution: California golden trout are endemic to the South Fork of the Kern River (SFKR), which flows into Isabella Reservoir and to Golden Trout Creek (GTC) (including its tributary, Volcano Creek),which flows into the Kern River (Berg 1987). Initially (1909 and earlier) California golden trout collected from Golden Trout Creek and transported north by pack train, extending their range by some160 km by 1914 (Fisk 1969). They were also translocated into many other waters within and outside California, including the Cottonwood Lakes not far from the headwaters of Golden Trout Creek and headwaters of SFKR, such as Mulkey Creek (Stephens et al. 2004). The Cottonwood Lakes served as a source of golden trout eggs for stocking other waters, beginning in 1917, and are still used for aerial stocking of lakes in Fresno and Tulare Counties (Stephens et al. 2004). As a result of stocking in California, these fish are now found in more than 300 high mountain lakes and 1100 km of streams outside their native range (Fisk 1969). Unfortunately, many, if not most, of these native and 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). Golden trout are also widely distributed in lakes and streams of the Rocky Mountains, but most populations there are also likely hybridized with either rainbow or cutthroat trout. It is possible that a few unhybridized populations still exist from early transplants in the Sierras and elsewhere, but they are likely to have limited genetic diversity due to small numbers used to establish these populations.

Abundance: Within their native range, California golden trout are known to 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 appear particularly in degraded reaches of steam with little cover and food. Presumably, densities were much higher on average before livestock began grazing the drainage. Outside their native range, populations should not be regarded as contributing to golden trout conservation because most (if not all) have hybridized with coastal rainbow trout.

Knapp and Dudley (1990) estimate that golden trout streams typically support 8-52 fish/ 100 m of stream, although a recent estimate for Mulkey Creek, a tributary to the South Fork Kern River, was 472 fish/100m (Carmona-Catot and Weaver 2006). If the Knapp and Dudley figures are accepted as correct then in 1965, when the first major CDFG habitat management plan was issued (CDFG 1965), there would have been 2400-15,600 individuals in Golden Trout Creek (30 km) and 4000-26,000 in the South Fork Kern (50 km). Curiously, the high numbers in the South Fork Kern River are found in habitats degraded by grazing where there are extensive exposed reaches with decomposed granite substrates that are used for spawning (S. Stephens, pers. comm. 2008). The lack of cover in these reaches may also select for smaller fish, which are more numerous.

At present, if unhybridized fish exist only in 5 km of Volcano Creek, then there are only 400-2600 ‘pure’ golden trout left today, a drop of at least 95% from historic numbers. A caveat on this very rough calculation is that it may not be necessary to eliminate all rainbow trout genes from the population through eradication, if management focuses on golden trout phenotypes that show low introgression of rainbow trout genes. If this management strategy was used, the numbers of golden trout would be considerably higher and might include fish outside their native range as well. Nevertheless, because golden trout had already been eliminated from most of lower South Fork Kern River by 1965, where populations would have been most dense, the 95 percent decline figure may still be valid, even if populations with low introgression are counted.

California golden trout in the upper South Fork Kern River and Golden Trout Creek are introgressed with non-native rainbow trout. However, the levels of introgression are different in these two streams. On the South Fork Kern River there is a cline of introgression from the lower Kennedy Meadows area (94%) upstream to the headwaters (8%). All or nearly all trout are introgressed with non-native rainbow trout to some degree. In many reaches of Golden Trout Creek, levels of introgression are low, close to the limits of detection; only one or two fish out of 40 fish seem to be hybridized at low levels so there may be little real concern (Cordes et al. 2006; M. Stephens 2007).

Factors affecting status: The principal threats to California golden trout are (1) hybridization with rainbow trout, (2) competition and predation from alien trout, and (3) degradation of their streams from livestock grazing, which continues (legally) even in the Golden Trout Wilderness Area (Inyo National Forest).

*Hybridization***:** 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), but stocking of hatchery rainbows in the mainstem Kern River and in the South Fork Kern River at Kennedy Meadow continues to support a popular put-and-take sport fishery (even though this violates CDFG golden trout policy). In addition, a golden trout brood stock operation was established in the Cottonwood lakes in 1891, near the SFKR headwaters, and this population, the source of most golden trout transplants to other watersheds, was apparently contaminated with rainbow trout fairly early in its history. In the SFKR, rainbow trout were able to move upstream over the deteriorated Schaeffer Fish Barrier upstream to the Templeton Fish Barrier. 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. 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 on the SFKR and hybridizing with them (Cordes et al. 2006). In the Golden Trout Creek drainage hybridization only affects a small percentage (about 5%) of the trout. Only the population in tiny Volcano Creek has escaped this problem but apparently has relatively low genetic diversity. In the South Fork Kern basin, only a few headwater populations may have escaped hybridization, although even this is not certain (Cordes et al. 2006).

Likewise, most places where golden trout have been planted outside their native range have likely been planted with rainbow trout at one time or another as well or originated from hybrid stocks. Hybridization with rainbow trout is a problem because the hybrid fish are likely to less brightly-colored than the native golden trout. The rainbow trout phenotype eventually becomes dominant, so the fish look like rainbow trout everywhere else they occur. This has been demonstrated well in the lower SFKR where hatchery rainbow trout have been planted annually since the 1930s and the few wild 'golden' trout still left are heavily hybridized, with a rainbow trout appearance. Hybridization can ultimately result not only in the loss of the uniquely colored variety of trout but in the 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 remain phenotypically golden trout.

In 2004, CDFG began planting only triploid (sterile) rainbow trout in the lower SFKR to try to eliminate additional hybridization. There is considerable demand for a hatchery-supported fishery in the basin, but anglers often move fish around, compounding problems for the already-besieged golden trout. It is thus assumed that planting the presumptive sterile fish will provide a fishery without further jeopardizing golden trout. There are problems with this assumption, including the possibility that not all the fish planted as triploids are sterile. Even sterile trout can have negative effects on resident golden trout, through predation, competition for food and space, and spread of disease (next section).

Alien trout: In addition to the threats of triploid rainbow trout, predation and competition from introduced brown trout are a continuous threat. Brown trout were eradicated from defensible upstream habitat in the SFKR in the early 1980s and barriers were constructed to prevent their reinvasion (Ramshaw, Templeton and Schaeffer barriers), although brown trout still dominate nearly 780 km of former golden trout water in the SFKR basin (Stephens et al.2004). In these reaches, they 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 1993, CDFG 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 the trout got there is not known, but it would have been relatively easy for anglers to move fish over the barrier. By the early 1990s, both Templeton and Schaeffer fish barriers had deteriorated and the Schaeffer Barrier did allow upstream fish passage. Both barriers were replaced with substantial concrete structures in 1996 and 2003 respectively. 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 non-native trout.

Livestock grazing: Grazing of livestock is permitted in designated Wilderness Areas, such as the Golden Trout Wilderness Area, so it occurs around Golden Trout Creek and South Fork Kern River where California golden trout reside. While levels of cattle grazing have been reduced in recent years (e.g., two of the four allotments are being rested for up to ten years, S. Stephens et al. 2004), the negative effects of grazing at all levels in the fragile meadow systems of this region have been well documented (Knapp and Matthews 1996, Matthew 1996b). Basically, grazing reduces habitat by reducing the amount of streamside vegetation, collapsing banks, making streams wider and shallower, reducing bank undercutting, polluting the water with feces and urine, increasing temperatures, silting up spawning beds (smothering embryos), and generally making the habitat less complex and suitable for trout. The result is further declines in trout populations.

Other threats: Although California golden trout waters are entirely within Sequoia and Inyo National Forests, they are faced with threats from other kinds of human use, including off-road vehicles, recreational damage by hikers and horse packers, fire suppression activities, and possibly introduced beaver. A particular problem is movement of off-road vehicles through Monache Meadows and the extreme degradation of the SFKR due to multiple causes throughout that area.

Conservation: Ever since it was realized in 1968 that California golden trout in the SFKR were being threatened by alien trout, mainly brown trout, major efforts have been made 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 kill all unwanted fish above or between the 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. Despite these efforts, most populations of California golden trout are hybridized and are under continual threat from brown and rainbow trout invasions. Thus a focus of conservation should be protection of the original gene pools of golden trout in Golden Trout Creek and South Fork Kern River as (1) a source for future fish transplants, (2) stocks that can be genetically compared with introduced populations, and (3) an aesthetic measure.

Major reasons why efforts to protect the golden trout have been inadequate are shortage of funding for fisheries management agencies and perhaps full realization of the threats facing California's state fish. Implementation of the recovery plan for California golden trout could reduce the threat of extinction through management of hybrids, multiple barriers (redundancy in case one fails), improved management of the watersheds, and elimination of non-native trout populations (S. Stephens et al. 2004). The Conservation Strategy (Stephens et al. 2004) has not been fully implemented. However, several key goals of this document have been met, including the replacement of two failing fish barriers and the increase in genetic research to better understand the current status and distribution of the California golden trout in this watershed. An additional downstream barrier, in a remote location, is being planned. Two of the four grazing allotments are being rested for ten years. However, more needs to be done, as indicated in S.

Stephens et al. (1995) and Sims and McGuire (2006); they include (1) repair or replacement of barriers, (2) eradication of all rainbow trout and brown trout populations that threaten California golden trout, (3) greatly improved management of livestock grazing (preferably elimination of grazing altogether), and (4) improved management of recreation to reduce impacts on the trout.

Barrier improvement: Barriers to prevent alien trout from invading golden trout waters are important, if ultimately short-term, management measures. Templeton and Schaefer 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 must be found. D. Christensen and S. J. Stephens suggested (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 is in the early planning stages about 10 km upstream of Kennedy Meadows. Whether such a structure will ever be built in a Wilderness Area is unclear (S. Stephens, pers. comm. 2008).

Eradication of aliens: Eradication of non-native trout continues to be a necessary measure. Aliens must be eliminated as soon as they are detected, anywhere in the watershed, including hybrid fish from headwater lakes. 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. Given the controversial nature of the use of poisons, a thorough risk analysis should be conducted for streams for which their use is contemplated which involves risks entailed if they are *not* used, as well as if they are used.

Use of genetic techniques: Increased use of new genetic techniques is needed to allow for genetics-based management. Thus, the best management approach in the Golden Trout Creek watershed (now that introgressed trout have been removed from headwater lakes) is to simply monitor the levels of introgression every five years for change. No other management action is recommended for this population. The Volcano Creek population needs to be reevaluated to determine if they are genetically bottlenecked. Establishment of refuge populations elsewhere for these trout should be considered. All trout in the SFKR are introgressed with non-native rainbow trout. It appears the golden trout in GTC and the SFKR are slightly different genetically (Stephens 2007) and they should be managed as separate populations (Stephens et al. 2004; Stephens 2007). Efforts should proceed with a new fish barrier at Dutch John Flat, if possible. Once that barrier is in place, then a decision needs to be made as to which California golden trout population on the SFKR best genetically represents this subspecies. Once a decision is made, unwanted populations would have to be systematically eliminated and replaced with the selected California golden trout from the SFKR.

Elimination of grazing: Elimination of livestock grazing in the Golden Trout Wilderness Area is needed because it would result in rapid recovery of riparian areas and stream channels and protection not only of golden trout but of other endemic organisms in the Upper Kern basin. If complete elimination is deemed undesirable, then intense management of grazing to reduce impacts on streams should be instituted, including the use of allotment rotation and more use of cowboys to keep cows away from streams. Monitoring needs to occur to document that grazing practices are in compliance with appropriate Forest Service guidelines.

Recreation management: Improvement of recreation management is needed, which basically means better enforcement of existing laws and better education of the public. One step would be to manage only for catch-and-release fishing for wild trout in the entire upper Kern basin above Isabella Reservoir, as a way of reducing transport of fish above barriers and emphasizing the importance of maintaining native trout populations. The stocking of hatchery trout, including triploid rainbow trout, in the SFKR should be phased out, in combination with a major re-education program for anglers. Another step in recreation management is to allow low-impact recreation only (e.g. eliminate off-road vehicles from areas where they are currently permitted).

Integrated management: Annual monitoring of the native populations, now accomplished by CDFG (Carmona-Catot and Weaver 2006), should continue in order to determine population status and to look for presence of non-native trout. Two kinds of refuges should 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 that do not meet these criteria should be converted to one or the other type of refuge as soon as possible. This type of very intense management requires rapid, annual genetic assessments of refuge populations.

For additional more specific measures, see Stephens et al. (2004) and Sims and McGuire (2006).

A major boost for golden trout conservation has been the establishment of the Edison Trust Fund in 1996 by the Federal Energy Regulatory Commission as part of the relicensing of the Southern California Edison Company Kern River No. 3 Hydroelectric Project. The Trust Fund should produce about $200,000 per year to be used for implementation of the Upper Kern Basin Fishery Management Plan (Stephens et al. 1995), restoration of Kern River rainbow trout, and other improvements to fisheries in the upper Kern basin. Release of the funding was delayed for 10 years because of a lawsuit by rafting groups who also wanted a piece of the funding pie but they were finally denied. The most immediate benefit for California golden trout has been funding to study the genetics of all populations, to guide management.

Trends

Short term: The native populations in Golden Trout Creek and South Fork Kern River watersheds are mostly hybridized with rainbow trout, although the extent of hybridization in many populations is small. Genetically 'pure' populations exist in only a few kilometers of streams and this is likely to continue for the short term (<5 yrs). Elimination of introgressed trout populations in the headwater lakes of Golden Trout Creek will eliminate the infusion of new rainbow trout genes into this population. However, the general trend in recent years seems to be downward, for unhybridized golden trout.

Long term: Populations in the native watersheds have persisted only because of cooperative interventions by fish managers in the California Department of Fish and Game, US Fish and Wildlife Service, and US Forest Service. The native populations suffered major declines during the 19th and first half of the 20th Century from overfishing and heavy grazing. Invading brown trout have displaced California golden trout, including hybrids, from all reaches below artificial barriers, so the golden trout are now confined to a few kilometers of stream in the Golden Trout Creek watershed and in the South Fork Kern watershed. Improvement in this condition will require active management all aspects of golden trout habitat, as well as reducing the effects of hybridization (or learning to live with low levels of it).

Within the restricted reaches, numbers of golden trout, including hybrids, have undoubtedly increased since the days of heavy harvest and grazing, but these numbers are presumably less than historic highs because of continued grazing and other human impacts. All introduced populations of California golden should be regarded as heavily hybridized unless otherwise demonstrated.

Status: 2. High likelihood of extinction in 50-100 years, or sooner (Table 1). California golden trout is currently considered to be a Species of Special Concern by CDFG, Species of Concern by USFWS, and a Sensitive Species by USDA Forest Service. A petition to USFWS to list it as endangered was submitted by Trout Unlimited in 2000 (Behnke 2002) but the trout has not yet been listed. The USFWS determined in a 90-day finding that the proposal deserves additional consideration. The listing proposal is currently undergoing a year-long (in 2008) review to determine if listing is warranted. While much of South Fork Kern River and Golden Trout Creek watersheds are managed for golden trout, with a few exceptions the populations have become introgressed to some degree with rainbow trout (Cordes et al. 2002; S. Stephens et al. 2004, M. Stephens 2007). Until recently, the California golden trout was perceived as being in no danger of extinction because it had been widely introduced throughout the Sierras and the Rocky Mountains. However, not only are introduced populations on a different evolutionary trajectory from the native populations (most are in lakes) but they have largely hybridized with rainbow trout, and can no longer be considered part of a conservation strategy (unless undoubted non-hybridized populations are located). Meanwhile, even the lightly hybridized native populations can only be maintained through constant intervention such as building and repairing of barriers and eradication of non-native trout and golden-rainbow hybrids (Behnke 2002).

Metric	Score	Justification
Area occupied	1	"Pure" California golden trout confined to a few small tributaries.
Effective pop. Size	2	Tributary populations show signs of genetic bottlenecking but probably still contain 100-1000 adults, although effective population size could be smaller.
Intervention dependence	3	Persistence requires maintenance of barriers and continued vigilant management.
Tolerance	2	Require conditions present in relatively undisturbed small alpine streams
Genetic risk	1	Hybridization with rainbow trout is a constant high risk
Climate change	3	Risk declines with better watershed management.
Average	2	12/6
Certainty (1-4)	4	Well documented

Table 1. Metrics for determining the status of California golden trout, where 1 is poor value and 5 is excellent.

LITTLE KERN GOLDEN TROUT
Oncorhynchus mykiss whitei

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 caeca (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 Little Kern golden trout look more similar to California golden trout than 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 identical 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, while Konno (1986) showed the fish have relatively small home ranges.

Habitat Requirements: Little Kern golden trout have the same habitat requirements as California golden trout in the neighboring South Fork Kern River and Golden Trout Creek. Basically they are adapted for living in small, meandering meadow streams and the higher gradient tributaries that feed them. Myrick and Cech (2003) found that these trout are physiologically adapted to optimal temperatures of 10-19°C, although they no doubt encounter higher temperatures in their streams at times during summer months. They co-occur with Sacramento suckers, also native, in some areas.

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 1984; Behnke 2002). By 1973 their range had shrunk to five headwater streams in the basin (Wet Meadows Creek, Deadman Creek, Soda Spring Creek, Willow and Sheep creeks, and Fish Creek) plus an introduced population (originating from Rifle Creek) in Coyote Creek, a tributary to the Kern River nearby (Ellis and Bryant 1920; Christenson 1984). The Upper Coyote Creek population subsequently was found to be a population 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 creek. Starting in 1975, systematic efforts were made by DFG and other agencies to restore Little Kern golden trout to its historic 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. This effort resulted in their apparent restoration to about 51 km of stream plus introduction into three headwater lakes by 1998. However, subsequent genetic studies indicated that many of the re-established populations have hybridized with rainbow trout (M. Stephens 2007) and the extent of unhybridized fish is uncertain. Recent genetic studies have identified unhybridized Little Kern golden trout populations in Upper Soda Spring Creek, Trout Meadow Creek, Clicks Creek, Burnt Corral Creek, Tamarack Creek, Deadman Creek, Wet Meadows Creek, Fish Creek and Coyote Creek, which were most of the original refuges. All of these streams except for Coyote Creek are within the native drainage (M. Stephens 2007). Overall, it appears that Little Kern golden trout currently occupy about 31% of their historic habitat but the most secure populations are above barriers in a few small headwater streams (<10% of historic habitat).

Abundance: When Little Kern golden trout were at their minimum range (16 km of stream), their population was estimated at 4500 fish (Christenson 1978). If it assumed they currently persist in 50 km of small streams, with 300 fish age 1+ and older per km (500/mi; Christensen 1984), the total numbers probably hover around 15,000 such fish. Quite likely, the numbers are considerably less than that, especially during low-flow years. If only unhybridized fish are counted, then the number is only those confined to the 20 km or so of refuge streams, perhaps 5-6,000 juvenile fish. The estimated number of spawning Little Kern golden trout within each refuge is unknown and may be small, so thus may limit long term persistence of some of these populations.

Factors affecting status: Little Kern golden trout are confined to the headwaters of the Little Kern River in small tributary streams which are isolated from one another. All are on public land managed by Sequoia National Forest, with upper Soda Spring Creek in land managed by Sequoia National Park. The primary threat to these remaining populations is introgression with hatchery rainbow trout or competition from brown trout that might be moved from illegally from the Kern River. At the present time brown trout appear to be gone from the basin and there is no stocking of hatchery trout (S. Stephens, 2008, pers. comm.). The reason hybridization with rainbow trout is a concern is that the rainbow trout phenotype may come to dominate the population, so even hybrids look more like rainbow trout than golden trout.

Additional problems include habitat loss from the regions long history of grazing, logging, and roads, as well as stochastic events such as floods, drought, and fire. Such events potentially increase local population extinction risks (Moyle 2002). For a full discussion of broader problems, see the California golden trout account.

Conservation: One of the three main goals of a multi-agency management plan for the upper Kern River basin is restoration of native trouts to a point where they can be delisted (S. Stephens et al. 1995). Problems addressed in the plan include planting of non-native trout (including hatchery rainbow trout), grazing in riparian areas, and heavy recreational use of the basin, including angling. Since the trout was listed, several kilometers of stream and seven headwater lakes have been treated with piscicides to eradicate hybrid Little Kern golden trout x rainbow trout as well as brook trout. However, a major problem facing managers is that fish available for restoration programs are either introgressed (even if lightly) with rainbow trout or come from small isolated populations with limited genetic diversity (M. Stephens, pers. comm. 2007). For a full discussion of problems and solutions, see California golden trout.

Trends:

Short term: The interagency recovery efforts have expanded the range of this subspecies back into some its original streams, but there is considerable uncertainty about the extent to which these efforts have been reversed by hybridization with rainbow trout or by planting hatchery-reared Little Kern golden trout that might have become ‘contaminated’ by rainbow genes in the Kern River Planting Base, even though the Kern River golden trout were raised in a separate building. If both ‘pure’ and lightly hybridized trout are counted, numbers have probably been stable for the past five years.

Long term: Little Kern golden trout were once very abundant and widespread in the Little Kern Basin and were subject to intensive fisheries as a consequence. Since the 19th century, however, overexploitation combined with habitat degradation from grazing and, most importantly, hybridization with rainbow trout has reduced their populations to those that occupy less than a third of historic habitat. Unless the hybridization problem is solved, long term decline to extinction as ‘pure’ Little Kern golden trout is likely.

Status: 2. The Little Kern golden trout has high probability of disappearing as a distinct entity in the next 50-100 years despite major efforts to protect it (Table 1). This possibility has long been recognized and serious management efforts began in 1975. The Little Kern River was included as part of the Golden Trout Wilderness Area in 1977. The subspecies was listed as threatened by USFWS in 1978 and a management plan was completed by DFG in the same year (Christenson 1978); it was revised in 1984 and again in 1995. Based on recent genetic information, the management plan probably should be revised again. Critical Habitat has been designated in the Little Kern River, main channel and all streams tributary to the Little Kern River above the barrier falls located on the Little Kern River about 2 km downstream of the mouth of Trout Meadows Creek, Tulare County. However, little has apparently changed within the critical habitat since its designation, except perhaps to prevent further degradation from timber harvest, cattle grazing, and other factors.

Metric	Score	Justification
Area occupied	1	Secure populations are above barriers in a few small headwater streams.
Effective pop. size	3	Existing populations fairly dense
Intervention dependence	3	Barriers must be maintained and non-native trout removed using piscicides when needed.
Tolerance	2	Very sensitive to changes in habitat that raise temperatures and degrade water quality.
Genetic risk	2	Hybridization with rainbow trout a constant high risk
Climate change	3	Risk declines with better grazing and other management practices
Average	2.3	14/6
Certainty (1-4)	4	Golden trout biology and conservation is well documented.

Table 1. Metrics for determining the status of Little Kern golden trout, where 1 is poor value and 5 is excellent.

KERN RIVER RAINBOW TROUT
Oncorhynchus mykiss gilberti

Description: This subspecies is similar to coastal rainbow trout but its coloration is brighter, with a slight tinge of gold; it has heavy 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 trout (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. D. S. Jordan's 1894 designation of this fish as a distinctive subspecies of rainbow trout 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. 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. gilberti* 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). This analysis indicates that Kern River rainbow trout represent a distinct lineage that is intermediate between coastal rainbow trout and Little Kern golden trout, although there is also some evidence of recent hybridization with coastal rainbows, presumably of hatchery origin. It is also possible that the mixed nature of the genome was the result of planting of the other two golden trout subspecies into Kern River rainbow trout waters (Bagley and Gall 1998).

Life History: No life history studies have been done on this subspecies, but its life history is no doubt 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 25cm TL are rare today (S. Stephens et al. 1995),

Habitat Requirements: Little information is available on Kern River rainbow trout, but in general the habitat requirements should be similar to other rainbow trout, with some modifications to reflect the distinctive environment of the upper Kern River (Moyle 2002).

Distribution: This subspecies is endemic to the Kern River and tributaries, 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 Onyx (S. Stephens et al. 1995). It has been extirpated from the Kern River at least from the Johnsondale Bridge (ca. 16 km above Isabella Reservoir) on downstream. Today, remnant populations live in the Kern River above Durrwood Creek, in Upper Ninemile, Rattlesnake and Osa Creeks, and possibly upper Peppermint Creek, and others (S. Stephens et al. 1995). Bagley and Gall (1998), using a variety of genetic techniques, determined that several populations, mostly located in the middle section of the Kern River drainage appeared to be unhybridized Kern River rainbow trout: Rattlesnake Cr. (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 in the middle of the historic range lacked apparent influence from California golden trout (either anthropogenic or natural) that was seen in the upper sections of the Kern and also lacked apparent rainbow trout hybridization seen in the lower sections. While Behnke (2002) doubts 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. 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 Kaweah-Kern River and Chagoopa Creek, which have maintained their genetic identity (M. Stephens 2007).

Abundance: 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 (600-1400 trout per mile) of all sizes (Stephens et al.1995). There is no data on current abundance but if it assumed they currently persist in 20 km of small streams, with 400-900 trout per km, the total numbers would be 8,000-18,000 fish total. If 10% of these fish were capable of reproduction, then the effective population size would probably be less than 1000 fish. These estimates are highly questionable, given natural variation in numbers, smallness of sample sizes on which they are based, and uncertainties about the actual distribution of Kern River rainbow trout, but they do suggest that absolute numbers in the wild are low and vulnerable to reduction by natural and human-caused events.

Factors affecting status: Kern River rainbow trout are confined to 4-6 small streams, each isolated from one another, plus some sections of the Kern River. The entirety of their habitat is on public land, including Sequoia National Park. The primary threats to the remaining populations are identical to those facing other endemic trout of the southern Sierra but center on interactions with non-native trout: (1) hybridization with hatchery rainbow trout, which are still planted in the upper Kern Basin, (2) hybridization with golden trout planted or moving into their waters, and (3) 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 (and even likely), 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 reducing population persistence (Moyle 2002). For a full discussion of broader problems, see the Little Kern golden trout and California golden trout accounts in this report.

Conservation: A multi-agency management plan for the upper Kern River basin has the following as one of its three major goals " 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). Problems addressed in the plan include planting of non-native trout (including hatchery rainbow trout), grazing in riparian areas, and heavy recreational use of the basin, including angling There is now in place the Edison Trust Fund (see California golden trout for details) that should provide at least $200,000 each year to implement the management plan and improve fisheries in the upper Kern Basin. Current (2008) funding provides for developing a conservation hatchery for Kern River rainbow trout, for increasing patrols of wardens in areas where the trout are fished, and for funding studies on genetics (http://www.kernriverflyfishing.com/)

For a full discussion of conservation problems and solutions, see the Little Kern and California golden trout accounts in this report.

Trends:

Short term: Trends are not known, but they do not seem to have changed in last 10-15 years (since 1995). However, status of Kern River rainbow trout could change rapidly considering the limited number of local populations.

Long term: Kern River rainbow trout were once very abundant and widespread in the upper Kern Basin, and were subject to intensive fisheries as a consequence. Since the 19th century overexploitation combined with habitat degradation from grazing and, most importantly, hybridization with other trout, have reduced its populations to a small fraction of historic numbers, probably less than 5%.

Status: 2. The Kern River rainbow trout has a high probability of disappearing as a distinct entity in the next 50-100 years, if not sooner. It is listed as a Special Concern (formerly Category 2) species by USFWS, indicating that it is a candidate for listing as a threatened species but that there is inadequate information to make the determination. It is also a California Species of Special Concern (Moyle et al. 1996). In fact, they were once thought to have been extirpated through introgression with nonnative rainbow trout (Gerstung 1980, Moyle 2002). Today, there is adequate information to justify its listing as a threatened species, in order to improve the likelihood that management actions will be taken to keep it from going extinct. However, additional genetic work is needed to better define its status and distribution. Basically, Kern River rainbow trout are confined to a handful of streams that are subject independently and collectively to natural and human-caused trauma, such as landslides and fire, even through most are in protected areas. The biggest single threat continues to be invasions of non-native rainbow trout, brown trout, and brook trout into their remaining streams, either through natural invasions or angler-assisted introductions. Protection of the remaining populations therefore requires constant vigilance and the ability to react quickly to counter new threats. Fortunately, the Kern River Trust Fund provides at least some secure funding for management.

Metric	Score	Justification
Area occupied	1	Found only in 4-6 small tributaries and short reaches of the Kern River
Effective population size	3	Much uncertainty about size of unhybridized populations
Intervention dependence	2	Barriers must be maintained, planting of hatchery fish managed, grazing managed, and other continuous activities
Tolerance	3	Presumably fairly tolerant, as are most rainbow trout but not tested
Genetic risk	1	Hybridization with rainbow and other golden trout a constant high risk.
Climate change	3	Risk declines with better land management
Average	2.2	13/6
Certainty (1-4)	3	This is least studied of the three Kern River trouts.

Table 1. Metrics for determining the status of Kern River rainbow trout, where 1 is poor value and 5 is excellent.

McCLOUD RIVER REDBAND TROUT

Oncorhynchus mykiss stonei

Description: The following description is based on the Sheepheaven Creek population (Hoopaugh 1974, Gold 1977) that seems to have a narrower range of characters than is found in populations throughout the range of the subspecies. Behnke (1992), however, considers this population to best represent the subspecies because it is unlikely to have had any history of hybridization with introduced rainbow trout. Overall body shape of this redband trout is similar to the "typical" trout shape 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 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, of the redband trout have basibranchial teeth, a characteristic more typically associated with cutthroat trout.

Taxonomic Relationships: Distinct "redband trout" from the lower McCloud River were first recognized in 1885 by Deputy US Fish Commissioner Livingston Stone who was responsible for a fish hatchery located on the river, although the fish he recorded were most likely resident coastal rainbow trout. The redband trout we recognize today are varieties of rainbow trout that resulted from invasions of headwater systems thousands of years ago, followed by isolation. 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). A complicating factor is that many populations have hybridized with the closely related coastal rainbow trout, which have been widely planted in historic redband trout streams. Behnke (1992, 2002) considers redband trout in the western USA to consist of a number of distinct lineages, each independently derived from early invasions of rainbow trout into headwater systems, which then became isolated through geologic events. Behnke (2002) indicates that the 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 the upper McCloud River watershed apparently retains unhybridized redbands and these fish are now the exclusive possessors of the subspecies epithet (Behnke 2002). The population in Sheepheaven Creek, described above, is so distinctive that Behnke suggests 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 M. Stephens (2007) using DNA (ALFP technique) support the conclusion that the Sheepheaven Creek form is distinct. However, further studies are needed on the relationship of the Sheepheaven population to other populations in the upper McCloud Basin before a formal designation is made and a subspecies name other than *stonei* is used.

Life History: Available information suggests that the life history of McCloud River redband trout is similar to that of other rainbow trout, 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.

Habitat Requirements: Habitat requirements for the McCloud River redband are derived from the conditions of Sheepheaven Creek (Hoopaugh 1974; Moyle 2002) and the McCloud River, based on descriptions in the Draft Redband Trout Conservation Agreement (Draft RTCA 2007), which summarizes information from unpublished habitat surveys. Sheepheaven Creek is a small, spring-fed stream at an elevation of 1,433 m. Water temperature in summer typically reaches 15°C and the flow drops to 0.03 m^3 sec^{-1} (1 cfs). The stream flows for about 2 km from the source and then disappears into the stream bed. However, during times of drought flow drops to a trickle and the stream becomes intermittent; as a consequence summer temperatures of the water can exceed 22°C. 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 presumptive redband trout are generally small, dominated by riffles and runs with under-cut banks. Where present, 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 6-10°C. Fry rear in shallow water on stream edges for the first weeks after emergence.

Distribution: McCloud River redband trout are confined to small creeks that are tributary to the upper McCloud River (Table 1). Historically, they were apparently present in the mainstem 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 Sheepheaven Creek were transplanted into Swamp Creek in 1972 and 1974 and into Trout Creek in 1977 (Draft RTCA 2007). They are now established in both streams. According to a 1996 DFG survey, putative redband trout exist in streams with a total length of about 67 km. During most years, about half of the stream km are dry by late summer, so total permanent habitat for the trout is presumably about 25 km, less in dry years. Potential habitat, including the upper McCloud River, is about 98 km, or about 50 km in dry years (Draft RTCA 2007).

Stream	Summer Flow class	Redband status	Isolation	Comments
Sheepheaven	1	2	3	Key population
Trout	2	2?	3	Introduced from Sheepheaven
Swamp	1	2	3	Introduced from Sheepheaven
Edson	1	2	3	
Tate	1	1	2	
Moosehead	1	1	2	
Raccoon	1	1	2	
Blue heron	1	1	2	
Bull	1	1	2	
Dry	1	1	2	
Upper McCloud	3	0	1	Dominated by Non-native trout

Table 1. Redband trout streams in the upper McCloud River watershed. Summer flow class (1 =<1 cfs, 2 =1-5 cfs, and 3 =>5 cfs in late summer in most years). Redband status (2 = 'pure' population, 1 = likely 'pure' population but status needs clarification, 0 = all redbands present likely hybridized). Isolation (3 = no passable connections with other streams, 2 = connections present in wet years in lower reaches, and 1 = no barriers to non-native trout).

Abundance: Redband trout creeks were surveyed a number of times in 1975-1992 (Table 2, from draft RTCA 2007). Numbers of fish estimated were highly variable and depended on the stream and habitat sampled; the numbers ranged from 75 to 1100 per km. The 1987-1992 drought resulted in severe reductions in populations in Sheepheaven, Edson, and Moosehead Creeks; populations in other waters fared better (E. Gerstung, CDFG, pers. comm. 1995). If it is assumed that 100 fish greater than 50 mm TL per km is a reasonable average for dry years, then the minimum total population for the 10 streams would be about 2500 fish, although in wet years populations could easily reach many times that number.

Factors affecting status: Long-term survival of populations of redband trout in small creeks such as Sheepheaven Creek poses problems because the streams may become largely dry during drought years, a process accelerated by poor watershed management, including grazing of livestock in riparian areas. Fortunately, interest in conservation of McCloud River redbands has resulted in a reversal in downward trends in populations and habitat quality. The factors, past and present, that have threatened McCloud River redband trout populations are (1) alien trout, (2) hybridization, (3) logging, (4) grazing, and (5) harvest. The redband streams can be regarded as exceptionally vulnerable to these factors because they are naturally fragile in the face of drought, flood, and other factors.

Alien trout: Rainbow, brown (*Salmo trutta*), and brook trout (*Salvelinius 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. Generally where alien trout are present, redband trout are absent. The exact causes of their disappearance have not been documented in the McCloud, but presumably it is a combination of predation on young (brown trout), competition for space (all species), disease introductions (all species), and hybridization (rainbow trout, next section). Fortuitously, a number of the redband trout streams were too small or isolated to be subject to introductions although some (e.g. Trout Creek) were nevertheless invaded at one time or another.

Hybridization: Hybridization between introduced coastal rainbow trout and redband trout in some respects is a natural event: both are native to California and both can do well in small streams. The concern over hybridization is that once it occurs, the rainbow trout phenotype and genotypes tend to dominate, resulting in a loss of the distinctive, brightly-colored redband trout phenotypes and of their contribution to the rainbow/redband genetic complex (i.e., biocomplexity).

Logging: The region is which the McCloud River redband trout live contains a checkerboard of private and public ownership, with most of the public lands being part of Shasta-Trinity National Forest. This resulted in extensive logging on both public and private lands. According to the draft RTCA (2007):

> "Small sawmills were operating in the upper McCloud River watershed starting in the late 1800s. At the turn of the century, railroads facilitated expansion of the sawmill capacity by allowing access to timber on steeper slopes, untapped by the previous horse/oxen era. Railroad-style logging predominated through World War II when truck and tractor operations replaced Shay locomotives and steam donkeys in the woods….
>
> 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, both as a legacy of the past and through continued logging. Impacts that continue include culverts blocking instream movement, removal of water to wet logging 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 streams containing redband trout (draft RTCA 2007).

Grazing: Grazing by cattle and sheep has likely 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, reduces trout habitat by eliminating streamside vegetation, collapsing banks, making streams wider and shallower, reducing bank undercutting, polluting the water with feces and urine, increasing temperatures, 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 find ways to keep cattle away from streams has no doubt led to considerable improvement in the condition of small streams in the McCloud River watershed and improved habitat for redband trout.

Harvest: It is likely that harvest was never a major problem in the small streams of McCloud basin but redband trout populations are small enough so even occasional harvest by anglers or scientific collectors could reduce populations (draft RTCA 2007).

Natural factors: The fact that existing redband trout streams are so small means that they are exceptionally vulnerable to natural factors such as floods, drought, and fire. However, the persistence of distinctive trout in tiny Sheepheaven Creek is a tribute both to the fish and to the springs that keep the creek alive, even during severe drought. Presumably most of the other 10 streams have similar 'safe' water sources. It is worth noting, however, that spring flows can be eliminated by volcanic activity (all streams sit on the side of Mt. Shasta, a volcano) and by loss of flow caused by climate change. In the short run, neither of these factors is likely to be important but in the longer run both may be.

Conservation: Conservation of McCloud River redband trout is active and ongoing, thanks to the leadership of California Trout, Shasta-Trinity National Forest, and the California Department of Fish and Game. The forging of a new Redband Trout Conservation Agreement (2007) is the latest step towards protecting these fish and their habitats. In the past, most management attention focused on the Sheepheaven Creek population, because it is so distinctive. Because this population existed only on private land and was threatened with extinction due to logging and grazing, some fish were transplanted to Trout and Swamp Creeks in 1977. Today, the conservation interests encompass all the populations and private and public landowners actively cooperate. Thus on private lands, considerable effort has been made to improve roads in ways that do less harm 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 the 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 that agreement. The recommendations are not in order of importance.

1. Establish a McCloud Redband Refugium: A portion of the upper McCloud River basin should be managed for the protection and enhancement of McCloud redband populations and their habitat. The refugium should include the main stem McCloud River and its tributaries above the confluence with Bundoora Spring Creek. While this area contains all the streams known to contain likely redband trout at the present time, reaches of other perennial tributaries not known to contain redband trout should nevertheless be included in the refugium as future restoration sites. Streams that have potential for expanding the range of redband trout should be evaluated and management plans that include eradication of non-native trout should be developed. In particular, the upper McCloud River should be evaluated as a refuge during periods of reduced stream flow caused by climate change.

2. Maintain and enhance existing habitats: These redband trout survive in remarkably small and fragile habitats, so continued work is needed to improve their ability to support redband trout and to reduce the probability that human activity will reduce their ability to support the fish. Of particular concern are grazing and logging practices, but other factors such as fire protection, angling and off-road vehicle also have been taken into consideration. While management plans and agreements are in place to protect the streams, continued vigilance is required to avoid long-term loss of habitat. The ongoing project to improve conditions in Trout Creek is a good example of the kind of work that needs to be done in the basin (C. Knight, California Trout, pers. comm. 2007).

3. Protect the genetic integrity of existing populations: The present populations are highly vulnerable to loss of genetic integrity (and phenotypic distinctiveness) due to hybridization with introduced rainbow trout. 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.

4. *Eliminate all planting of hatchery fish in streams of the upper McCloud Basin, including the McCloud River:* Stocking of hatchery rainbow trout still takes place in the upper McCloud River near public access points. In preparation for the ultimate eradication of non-native fish from the upper basin, such stocking should be halted except perhaps for that of sterile fish and perhaps fish in isolated fishing ponds (e.g., Lakin Pond).

5. *Develop and enforce angling regulations appropriate for protection of redband trout:* Basically, angling should be discouraged in most redband trout streams, although some could be managed for anglers participating in the Heritage Trout Program.

6. *Complete genetic evaluations of all populations:* This will provide a better basis for a subspecies description, if that is justified, which can help in formal protection procedures (i.e., if listing under the Endangered Species Act is required).

7. *Establish a regular 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).

Trends:

Short term: It is likely that habitat conditions and consequently populations of McCloud River redband trout have improved considerably in the past 10 years, although data is largely lacking. This is the predicted response to the many ongoing habitat restoration and protections efforts that have taken place. Presumably, the improvement in habitats, including protection of springs, has reduced population fluctuations and made the fish populations more resistant to drought.

Long term: It will take effort to make sure that the present apparent improvement in redband trout populations is not just a temporary phenomenon. A particular threat is climate change and potential reduction in stream flows in 25-50 years (once the full effects of global warming hit Mt Shasta). Until then, it is likely that redband populations will continue to increase as long as active management continues.

Status: 2. Because of the level of interest and management, there seems to be no *immediate* risk of extinction but the populations are small and exist in small isolated habitats, so status could change quickly (in 5-10 years). In longer time frames, extinction probability will increase as climate becomes warmer and droughts more frequent.

Metric	Score	Justification
Area occupied	2	Isolation of at least four populations provides some security, although the Sheepheaven Creek population may be distinct from other populations
Effective pop. Size	3	Minimum total population today is probably more than 1,000 adults, although individual populations presumably have effective sizes of 100-500 fish in drought years.
Intervention dependence	3	Continual monitoring and habitat protection and improvement are required.
Tolerance	3	It is likely they are fairly tolerant of high temperatures as are other redband trout but water quality their small streams can become too extreme if not carefully managed.
Genetic risk	2	Hybridization risk with rainbow trout high. Small populations during drought can create genetic problems
Climate change	1	Vulnerable in all streams because of small size.
Average	2.3	14/6
Certainty (1-4)	3	Information mainly for Sheepheaven Creek population

Table 2. Metrics for determining the status of McCloud redband trout, where 1 is poor value and 5 is excellent.

GOOSE LAKE REDBAND TROUT
Oncorhynchus mykiss ssp.

Description: Goose Lake redband trout are similar in appearance to other rainbow/redband trout. They have a yellowish to orange body color with a brick-red lateral stripe. 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 scales above the lateral line and 139 scales in the lateral series. See Behnke (2002) for color plates of both lake and stream forms of Goose Lake redband trout.

Taxonomic Relationships: Redband trout are inland forms of rainbow trout (Behnke 1992, 2002) and the Goose Lake trout belongs in the group of redband trout that Behnke (2002) calls "redband trout of the northern Great Basin." The Goose Lake form is most similar to redband trout of two adjacent basins: the Warner Basin, Oregon and Nevada, and the Chewaucan Basin, Oregon (Behnke 2002). This conclusion was based on the lower vertebral counts and higher gill-raker counts of redband trout in the basins and distinct genetic markers (Behnke 2002). The Goose Lake redband trout has not been assigned a subspecific name but Behnke (2002) suggests that the Goose Lake trout, along with various redband trout populations in isolated Oregon basins, should be placed together in *O. mykiss newberrii*. Berg (1987), using electrophoretic techniques, indicated that Goose Lake redband trout were distinctive enough genetically to warrant subspecies status although more recent work using DNA (ALFP technique) indicates a close relationship with Warner Valley redband trout (M. Stephens 2007). However, fish from Davis Creek, which flows into lower Goose Lake and/or the upper Pit River group genetically with putative Sacramento redband trout. No genetic differences between the lake and stream forms in the Goose Lake drainage have been documented. The USFWS has 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 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. Regardless of its ultimate taxonomic designation, the Goose Lake redband trout is clearly a distinct evolutionary unit confined to the Goose Lake basin and upper Pit River.

Life History: There are two life history strategies present in the Goose Lake redband trout: a lake strategy and a headwater strategy. The lake strategy fish live in Goose Lake where they grow to large size and spawn in tributary streams. The headwater strategy fish remain small and 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 forms 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 appeared in the streams again. The best explanation for this is that the new fish came from headwater populations. In the small cold streams of the Warner Mountains above the lake, scattered populations of resident trout have managed to persist, completing their entire life cycle in the streams. Most of these populations are above apparent barriers to fish coming in from the lake. Nevertheless, they seem to be identical to lake fish, even if they look quite different because of small size and color patterns reflecting responses to a stream environment. Presumably, small numbers of headwater redbands always moved 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 reached the lake and a few years later, they matured and spawned, renewing the cycle.

In California, the lake-dwelling form spawns in Lassen and Willow Creeks. If sufficient flows are available, they spawn primarily in Cold Creek, a small tributary of Lassen Creek, and 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 occurred in Willow and Lassen Creeks following snow melt and rain in the spring, usually during late March or in April. Spawning fish are rather pale looking, presumably from a life in murky water. Adults return to the lake following spawning. Young trout apparently spend one or more years in the stream before moving down into Goose Lake. In the lake, the trout presumably feed on Goose Lake tui chub, fairy 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 (files, CDFG).

The life history of the stream-dwelling form has not been studied, but it is presumably similar to that other redband and rainbow trout that live in small, high-elevation streams. Surveys by CDFG (J. Weaver, 1999 files; Hendricks 1995) indicate that headwater streams have 4-5 length classes of trout, with a maximum size is around 24 cm TL. It appears that fish in their third summer are 9-12 cm TL. Spawning was observed in May 14-15 in 2007, though spawning time is highly dependent on the water year and runoff (K. Ramey, CDFG, file report).

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. Goose Lake redbands nevertheless survive the warm temperatures, high alkalinities, and high turbidity that exist in Goose Lake in summer. Presumably, a major factor contributing to their survival is the extraordinarily high abundance of fish, fairy shrimp, and other food in the lake (P. Moyle and R. White, unpublished observations).

Most spawning areas are located in high-elevation sections of tributary streams and are up to 40-50 km from the lake. Prior to spawning, adults must have access from the lake to spawning areas. The spawning sites are reaches with clean gravels and suitable riparian cover for maintenance of cool water temperatures. Goose Lake redbands have been observed to spawn in lower reaches of Willow and Lassen Creeks when access to upstream areas is blocked (P. Chappell, pers. comm. 1995), but most spawning areas are upstream of the Highway 395 crossing.

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 creeks (1-2 cfs flows in late summer), became progressively warmer from headwaters to mouth, so that its headwater streams 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 (< 1 cfs flow in summer, often dry in lowermost reaches) in both headwaters and lower reaches could reach 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. The typical streams in the Warner Mountains are riffle dominated with undercut banks and pools in meadow areas that house most of the larger fish. Dense overhanging vegetation, especially willows, provide essential cover.

Distribution: Goose Lake redband trout are endemic to Goose Lake and its major tributaries. In California, Lassen and Willow Creeks are their most important streams although they are also present in smaller streams (Pine, Cottonwood, Davis, 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, Parker, and East Creeks, tributaries of the upper Pit River in California, contained trout genetically similar to Goose Lake redband. Similar results for upper Pit River redbands were found by M. Stephens (2007). In addition, two populations in the Warner Mountains above the Surprise Valley seem to be Goose Lake redbands, perhaps as the result of introductions (M. Stephens 2007).

Abundance: According to local history, in the 19th century the trout were once abundant enough in the lake so 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 again during the 1976-1977 drought, recovered during the wet early 1980s, and dropped precipitously during the 1986-1992 drought.

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, pers. comm.1995). The only large run documented in recent years in Lassen Creek was in 1988 when several hundred spawners were present (J. Williams, unpubl. data), which suggests that there were fewer than 1,000 adults in Goose Lake. In 1989, in the middle of the 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. B. Reid (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.

The stream form of the 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, a number which is presumably low compared to historic numbers, before the streams were degraded by grazing and other activities. Survey for California streams made in 1988 and 1999, show 600-1600 trout per km in Lassen Creek, which suggests that densities/numbers in California and Oregon streams are roughly comparable.

Factors affecting status: Goose Lake redband trout populations have been affected by many factors, but habitat degradation and diversions have been the biggest problem. 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. All problems are exacerbated, however, during periods of severe drought.

Habitat modification: 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 the streams, but since 1995 most of these problems have been eliminated or reduced. In addition, all streams have been degraded from human activities in the drainage (livestock grazing, road-building, logging, etc.). The 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. Roads are also a problem on some streams, especially where culverts may be barriers to fish movement or where the road-cuts are a source of silt. Some streams have multiple problems with poor water quality as the result of road building, channelization, and waste materials from uranium mines.

Diversions: Much of the critical stream habitat for Goose Lake redband trout is on private land and at times the water on which they depend upon is diverted to irrigate fields. On some streams, the small dams creating the diversions are barriers to fish movement (ODFW 2005). Diversions may be the biggest problem in dry years because they have the potential to dry up stream reaches that are refuges for trout and other fish when the lake is dry.

Overexploitation: When lake-dwelling fish are moving upstream to spawn, they are extremely vulnerable to angling or 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, although the lower reaches of the Goose Lake streams are now closed to angling during the spawning period but may still face some difficulty passing culverts and water diversion systems. In 1992, all headwater streams were closed to angling until it can be demonstrated they can sustain fisheries.

Introduced species: Brook, brown, and rainbow trout have been introduced into streams of the Goose Lake drainage and brown trout are known to persist in California in Davis and Pine Creeks (Hendricks 1995, S. Purdy, unpublished data, 2006). Brook trout are still present in at least one Oregon stream (ODFW 2005). California has not stocked any rainbow trout in the drainage since 1980, when electrophoretic studies indicated that the native trout 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 rainbows (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.

The potential for future introductions to disrupt native trout populations through disease, hybridization, predation, or competition remains. Numerous attempts have also been made to introduce warm-water fishes, including striped bass, into the lake, but they have been largely unsuccessful because of the lake's extreme environment. This does not preclude the possibility that at some time fish or invertebrate species could be introduced that would disrupt the lake ecosystem as it exists today.

Drought: Goose Lake dried up in the 1420s, in the 1630s, 1926 (with low lake levels from 1925 to 1939), and 1992. Thus, the key to the survival of the Goose Lake trout (and other fishes) is presumably conditions in the lower reaches of the streams, as well as conditions in the headwaters. During the dry periods, the lake dwelling trout persisted either (1) by maintaining populations in the lower reaches of the tributary streams, which assumes the streams had year-round flows, or (2) by repeated recolonization from the resident populations in the headwaters, which assumes that fish from headwater populations are capable of adopting the lake life history strategy.

Conservation: After a long period of neglect, there has been considerable interest in conserving populations of this unusual trout and those of other endemic fishes in the Goose Lake Basin. During the 1987-1994 drought, a proposal was developed to list the Goose Lake fish fauna as threatened under the federal ESA. In response, in 1991, the Goose Lake Fishes Working Group was formed, made up of representatives from both California and Oregon of private landowners, state and federal agencies, nongovernmental 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 this strategy was to conserve all native fishes in Goose Lake by reducing threats, stabilizing population numbers, and maintaining the ecosystem. The Conservation Strategy identified factors in each stream that were affecting fish and provided a list of actions since 1958 that were implemented to benefit potential problems. Since publication [of the conservation strategy] in 1996, a number of additional projects have been completed or long-term projects begun. These include 2 culvert improvements, 11 diversion or passage projects, 10 fencing projects, 16 habitat improvement projects, 11 fish surveys, and road improvement project to reduce sedimentation."

In the lower reaches of most streams, major actions taken 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 Game has removed natural and artificial migration barriers. Headcut control, bank stabilization, fencing of streams, planting of riparian vegetation, changed 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 the streams, 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 fish, especially during periods of severe drought.

Some of the management actions that are needed include:

1. Identification and modification of barriers to fish movement, especially diversion dams.
2. Identification, protection, and improvement of reaches of stream that are critical for spawning and rearing of lake strategy trout and for their survival through periods of drought. Currently identified as important for management are Cold Creek (tributary to Lassen Creek) and Buck

Creek (tributary to Willow Creek). At present, a diversion structure often diverts the flows of lower Buck Creek.
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 fish in other streams.
4. Improved management of headwater areas to protect streams from livestock grazing and other problems in addition to managing and protecting coldwater resources, particularly under predicted climate change scenarios.
5. Through education and management, ban the presence of non-native fish in the Goose Lake basin, including eradicating existing populations where possible. From the perspective of the trout, the abundant tui chub population and lake invertebrates have been an excellent food resource which presumably contributes to the large size attained by lake-dwelling trout. Introductions of alien fishes or invertebrates that could alter the forage base or add another predator should be banned, including the planting of hatchery trout in Oregon. Management to provide a sport fishery should focus on improving conditions for redband trout rather than on stocking non-native predatory fishes.
6. Because of the small size of spawning streams and the large size of adult trout, spawning redband trout are susceptible to poaching. Therefore, regular checking by wardens and others should be done each year to prevent poaching as adults mass in pools and in shallow spawning areas.
7. The Goose Lake Fishes Conservation Strategy should be fully implemented and revisited periodically to make sure it is up to date. The continued involvement of private landowners and public agencies is crucial for this effort, as is the continued involvement of University of California Cooperative Extension, which has provided coordination and scientific studies to support the conservation efforts.

Trends

Short term: Since 1995, conditions for Goose Lake redband trout in California have steadily improved and runs of lake fish have re-established themselves. Presumably headwater populations have increased as well.

Long term: ODFW (2002) indicated that most of the redband trout streams are impaired to a greater or lesser degree, as the result of the accumulation of 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. Streams in California suffer from similar problems although the largest stream, Lassen Creek, seems to be in better condition than most. Thus overall the carrying capacity of Goose Lake streams is presumably a fraction of their historic carrying capacity.

Status: 3. No immediate extinction risk (Table 1). Although the risk metrics suggest a higher rating (4) might be appropriate, the lower score has been given because (a) the 19 extant populations in California and Oregon are largely isolated from each other, (b) most stream populations are small, and (c) during drought periods (which will likely increase over the coming century) the lake population disappears and stream populations shrink.

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, and (d)

ODFW, Vulnerable or At Risk Species. 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) for the following reasons:

"…the Great Basin experienced a drought from 1987 to 1992, with 1994 also being a very dry year. The drought caused Goose Lake …to go dry in 1992. This second drought eliminated the lake habitat and, consequently the lacustrine redband trout that made spawning runs up connected creeks. This drought also undoubtedly reduced the available stream habitat. However… the numbers of redband trout… appear to have rebounded…An analysis of historic and current distributions based on area concluded that Great Basin redband trout currently occupy 59 percent of their historic distribution."

The USFWS analysis also cites the many successful restoration projects in the Goose Lake Basin as further reason for finding that listing was not needed.

Metric	Score	Justification
Area occupied	4	Present in six creeks in California and 13 in Oregon
Effective pop. Size	4	Lake spawners are <1000 but headwater populations presumably contain more fish, especially in Oregon
Intervention dependence	4	Long-term decline reversed by human actions, which must be continued if the fish are going to persist in numbers
Tolerance	4	Indirect evidence suggests they are more tolerant than most salmonids of adverse water quality.
Genetic risk	3	Genetic risks are currently low although hybridization with introduced rainbow trout may have occurred in the past.
Climate change	2	Because it mainly occurs in small streams that are now largely isolated from one another, these trout are very susceptible to major declines as the result of prolonged drought.
Average	3.5	21/6
Certainty (1-4)	2	Mostly 'grey' reports and expert opinion

Table 1. Metrics for determining the status of Goose Lake redband trout, where 1 is poor value and 5 is excellent.

EAGLE LAKE RAINBOW TROUT
Oncorhynchus mykiss aquilarum

Description: This subspecies is similar to other rainbow trout in gross morphology, but differs slightly in meristic counts (Table 1). It possesses 58 chromosomes rather than the 60 present in most other rainbow trout (Busack et al. 1980).

Table 1. Means (+/- one standard deviation) of meristic characteristics of Eagle Lake rainbow trout, compared to means of other western trout, modified from Busack et al. (1980).

Character	Eagle Lake rainbow trout	Rainbow trout	Cutthroat trout	Redband* trout
Lateral series	138.3 +/- 1.47	135	166	162
Scale rows above lateral line	27.4 +/- 0.28	25	37	33
Gill rakers	19.2 +/- 0.25	19	24	16
Pyloric caeca	57.0 +/- 2.5	55	48	36
Branchiostegal rays	10.9 +/- 0.11	12	11	10
Pectoral rays	14.3 +/- 0.14	15	14	13
Pelvic rays	10.0 +/- 0.06	10	9	9
Vertebrae	62.0 +/- 0.23	64	62	61

*McCloud River redband trout (Sheepheaven Creek).

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 Eagle Lake rainbow trout were derived from hybridization between native Lahontan cutthroat trout and introduced rainbow trout, Miller (1950) later retracted the hybridization theory. Needham and Gard (1959) then suggested that the Eagle Lake rainbow trout was 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 the Eagle Lake rainbow trout was derived either from immigration *or* unrecorded introduction of a rainbow trout with 58 chromosomes. Given the distinctive morphology, ecology, and physiology of this form, it is clear that the Eagle Lake rainbow trout is derived from a natural invasion from the Sacramento drainage. Behnke (2002) speculated that Lahontan cutthroat trout were the original inhabitants of Eagle Lake but 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 2002). Recent genetic studies (ALFP DNA techniques) suggest that the closest relatives of Eagle Lake rainbow trout are rainbow trout from the Feather River (M. Stephens 2007).

Life History: Eagle Lake rainbow trout are late maturing (at 2-3 years) and were historically long-lived, up to 11 years (McAfee 1966). Trout older than 5 years are rare in the lake (McAfee 1966), although individuals as old as 8-9 yrs have been caught (DFG, unpublished data). Originally, mature trout moved up Pine Creek (and probably the much smaller Papoose and Merrill creeks, which feed the southern basin of Eagle Lake) to spawn in response to high flows in March, April, or May. In Pine Creek, located along the western shore of Pine Creek, the main spawning areas were presumably the gravel-bottomed spring-fed creeks in the Bogard and Stephens meadow systems, such as Bogard Spring Creek, about 45 km from the lake. It is likely that the trout spent at least their first year of life in these creeks, reaching 15-20 cm FL before migrating to the lake, although it is possible some spent two years as well. They then grew to about 40 cm in their second year (first year in the lake), 45 cm in the third, 54-55 cm in the fourth, and 60 cm in the fifth year (McAfee 1966). Mature females produce 2,500- 3,000 eggs.

The life history of these fish is now different from the original pattern because Pine Creek has become difficult to access for spawning (see below). As the fish move up Pine Creek in the spring, a weir blocks their way and they are trapped by CDFG. The eggs and milt are stripped from the fish for artificial spawning. The embryos are then taken to Crystal Lake and Darrah Springs hatcheries where they are reared for 14- 18 months. The fish are marked and planted in Eagle Lake at 30-40 cm FL (CDFG, unpubl. data). 180,000-200,000 fish are planted in the lake each year, about half in the lake at the mouth of Pine Creek. These marked fish are then trapped and used for spawning when they return to Pine Creek. The marks are used to eliminate sibling crosses (reduce inbreeding) and to select for longer lived fish to compensate for longevity reductions that may have been caused by past hatchery practices (R. L. Elliott, CDFG, pers. comm. 1998). A captive, domestic strain of Eagle Lake rainbow trout is maintained at the Mt Shasta Hatchery, which has been planted widely in reservoirs of the state and used as a source for brood stock in other hatcheries in California, as well as elsewhere in the western USA. Eagle Lake rainbow trout are prized because of their delayed maturity, rapid growth, and longevity.

The diet of the trout 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*, and on benthic invertebrates, especially leeches and amphipods. By August, most of the trout switch to feeding on young-of-year tui chubs (King 1963, Moyle 2002, Moyle, unpubl. data).

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 (pH 8-9) lake. The lake consists of three basins, two of them averaging 5-6 m deep, the third averaging 10-20 m with a maximum depth of about 30 m. The shallow basins are uniform limnologically, 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.

Eagle Lake rainbow trout are stream spawners. They formerly migrated over 45 km upstream to spawn in the gravelly upper reaches of Pine Creek and its tributaries. Juveniles then spent their first year (perhaps two) in the stream before moving into the lake during high run-off periods that reconnected headwaters to the lake. During the summer, upper Pine Creek is a spring-fed trout stream, flowing at .03-0.14 cm/s through meadows and open forest, with modest gradients. The meadow streams have deep pools and glides with deeply undercut banks, providing lots of cover for trout. The Pine Creek watershed is described in detail by Pustejovsky

(2007). Unfortunately, the trout present today in the creek are almost entirely alien brook trout, in high densities.

Distribution: Eagle Lake rainbow trout are endemic to Eagle Lake, Lassen County, 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 Pine Creek Spawning Station. The trout have also been exported to other states and to Canada. It is unlikely that naturally reproducing populations of genetically pure Eagle Lake trout are present in any of these planted waters.

Abundance: Naturally spawned Eagle Lake rainbow trout 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 in the meadows, and the first construction of railroad grades across the meadows and streams, all of which altered the stream channel. Although commercial fishing for trout was banned in California in 1917, the Eagle Lake rainbow trout populations remained low, presumably because of the poor condition of Pine Creek (and probably Papoose and Merrill creeks as well) and the establishment of predatory largemouth bass and brown bullheads in the lake. During the 1930s, lake levels dropped as the result of diversion of water through the Bly Tunnel and prolonged drought, presumably reduced access of spawning trout to Pine Creek. The high alkalinities brought on by dropping lake levels apparently also eliminated bass from the lake, although bullheads persisted into the 1970s. Even with the return of wetter conditions, the trout populations showed little sign of recovery. In 1950, six trout were captured from Pine Creek and about 2,000 fertilized eggs were taken to Crystal Springs Hatchery. The 600 trout that resulted were used for brood stock (Purdy 1988). 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 a negligible amount of natural spawning (Pustejovsky 2007, Moyle, unpublished data).

At the present time, about 150,000-200,000 trout are planted in the lake each year, all first generation fish derived from adults captured at the weir at the mouth of Pine Creek; these planted fish support a major sport fishery for "trophy" trout. Hundreds of trout are trapped each year and roughly 2 million fertilized eggs per year are taken for hatchery rearing. There is little or no evidence of natural reproduction contributing to the lake population; the fish captured by anglers usually show signs of a year or more in a hatchery environment, mainly white snout tips and damaged or missing fins.

Factors affecting status: The factors affecting Eagle Lake rainbow trout fall into five categories: (1) habitat change, (2) hatchery rearing, (3) exploitation, (4) disease, and (5) introduced species.

Habitat change: Historically, the greatest single factor causing the near-extinction of Eagle Lake rainbow trout has been the poor condition of the Pine Creek watershed. The watershed was severely altered as the combined result of logging, grazing, and railroad and road building. Besides deforesting large chunks of the watershed and creating erosion-prone roads, 19th and early 20th century logging activity in the region resulted in a railroad being built across the Pine Creek drainage, restricting flow of the creek at one point and channelizing the streambed. This situation worsened when a highway (State 44) was built parallel to the railroad and forced the stream through several culverts. The combination of culverts and channelized stream created a nearly-impassible velocity barrier for the trout. Grazing livestock have been (and continue to be) a major problem because livestock concentrate around the stream (Pustejovsky 2007). In the lower reaches of the stream (Pine Creek Valley, etc.) most of the riparian vegetation is gone and the wet meadows have been so compacted that they have been largely converted into dry flats dominated by sagebrush. As the result of all these activities acting on the stream for nearly 100 years, the lower creek cut down into the former meadow 1-2 m and became more intermittent in flow during summer, with flows diminishing rapidly in the spring. As a consequence, the stream (especially the key spawning and rearing areas around Stephens Meadow) has been nearly inaccessible to spawning adults and contains less habitat for juvenile fish. As noted below, many of these problems have been addressed by a multiagency Coordinated Resource Management (CRMP) group and the habitat has been steadily improving (Pustejovsky 2007).

Even the lake habitat for the trout is not completely secure. The Bly Tunnel continues to be a threat to the lake. Although it was blocked off, it still discharges, through an eight inch pipe in the plug, 0.34 cms of Eagle Lake water into Willow Creek for downstream water right holders. While some of the water coming from the tunnel may be spring water, most of it is Eagle Lake water because it is chemically nearly identical to Eagle Lake water (Moyle et al. 1991). This is important because in the long run the lake is less likely to become severely alkaline in a prolonged drought if it has more water in it to start with.

Hatchery rearing: Eagle Lake rainbow trout are completely dependent on hatchery production for survival (Moyle 2002). If CDFG had not begun trapping these fish in the 1950s, they would now be extinct. Prior to this, they presumably persisted only because occasional wet years permitted access to upstream spawning areas through degraded stream channels and because the fish were exceptionally long-lived. The danger in the present program is that fish are being selected for survival in the early life history stages in a hatchery environment, rather than in the wild, and perhaps for early spawning (as has happened in steelhead, Araki et al. 2007). 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. Survival is further threatened by disease in hatcheries, loss of adaptation for life in the wild, loss of life history diversity, and inbreeding. National Marine Fisheries Service guidelines indicate that a salmonid population dependent on hatchery production cannot be regarded as viable in the long term (McElhaney et al. 2000). Fortunately, the present management program for Eagle Lake rainbow trout is aimed at establishing a self-sustaining wild population again (K. Vandersall, USFS, pers. comm., 2006, Pustejovsky 2007), although hatchery production is regarded as being a perpetual necessity in order to sustain the trophy fishery (P. Chappell, CDFG, pers. comm. 1998) and currently has a higher priority than re-establishment of a wild population.

Exploitation: As indicated above, in the 19th century, Eagle Lake rainbow trout were once heavily exploited by a commercial fishery, which probably contributed to its initial decline. Since the 1950s, however, demand for the trout in the lake sport fishery has been the principal reason its population has been maintained by hatchery production. If a run becomes re-established in Pine Creek, the trout fishery in the creek will have to managed in ways that do not effect recruitment to the lake.

Disease: A continual, if remote, threat to the survival of Eagle Lake rainbow trout is exotic diseases, either in the hatcheries in which they are reared or by introduction into the lake by hatchery-reared fish.

Alien species: Many different species have been introduced into Eagle Lake in the past, but none have persisted because of the lake's alkalinity. However, because of Eagle Lake's large size and accessibility, it is likely that other species will be introduced illegally and eventually one may succeed, altering the ecology of the lake. 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 (see above). The only alien species in the drainage now is brook trout, which is abundant in Pine Creek. Predation and competition by brook trout in Pine Creek may prevent reestablishment of Eagle Lake rainbow trout, so a program to eliminate this species from the watershed is needed. The high densities of brook trout in upper Pine Creek (Thompson et al. 2007), however, also demonstrate why Eagle Lake rainbow trout could maintain large populations in the lake with natural spawning. The high densities and growth rates of brook trout would presumably be duplicated by rainbow trout, if the brook trout were absent.

Conservation: Because of the interest in Eagle Lake rainbow trout, major efforts have been made in recent years to fix the passage problems in Pine Creek, through the CRMP process. As a result large sections of the creek have been fenced to keep out livestock, off-creek watering stations have been provided, an impassible culvert under Highway 44 has been replaced with a passable one, and a structure to divert water in Bogard Meadows has been removed (and the meadow fenced). However, the meadows along the lower creek are still heavily grazed by cattle and the creek below highway 44 is generally dry by May or June. There is nevertheless some evidence that Eagle Lake rainbow trout, at least during wet years, can make it up to the spawning areas and spawn successfully. In the 1980s, a few juvenile rainbow trout were found below Stephens Meadow; suggesting adults made it over the weir and migrated upstream (Moyle, unpublished data). In 1999-2005, biologists from DFG, USFS and U C Davis placed radio transmitters in a small number of adult fish which were released above the mouth weir (L. Thompson, UC Davis, pers. comm.). In 1999, one of these fish apparently made it into the Pine Creek headwaters, as its transmitter was recovered in Bogard Springs Creek, a tributary to Pine Creek above the highway (T. Pustejovsky, pers. comm.). In 2002-2006, Paul Chappell of DFG released a few ripe trout from the fish trap at the mouth of Pine creek above highway 40.

In September 2006, a crew from UC Davis, DFG, and USFS sampled Pine Creek to look for Eagle Lake rainbow trout (Moyle, unpublished data). They found evidence that the trout had spawned successfully in the creek in the past two years, because small numbers of young 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 or varying sizes. Presumably the rainbow trout were YOY. The rainbow trout tended to be in faster water than the brook trout, in reaches with deep overhanging cover. The UC Davis crew also found 3-4 similar sized rainbows in Pine Creek below the Bogard Spring Creek confluence, as well as a couple of rainbow trout in the 145 mm range living in a creek filled with brook trout of all sizes, speckled dace, Lahontan redside, and Tahoe sucker. Curiously, several of the 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 of the spawners were found alive in a culvert about 5 km below the highway, in a largely dry stream (no flow), along with a trout that was 142 mm SL.

In August, 2007, the entire length of Bogard Spring Creek was electrofished to remove brook trout to see if spawning success of transplanted adult rainbow trout could be increased (Thompson et al. 2007). Nearly 5,000 brook trout were removed from the 3 km long creek (ca. 1625/km), which is especially remarkable considering the creek is less than 1 meter wide for all of its length and mostly less than 40 cm deep. During the removal program, 170 yearling and two 1+ Eagle Lake rainbow trout were captured and returned to the creek. This evidence strongly indicates that a wild spawning population of Eagle Lake rainbow trout can be reestablished through translocation, although restoration may require trapping and trucking fish in both directions in some years.

Given that Eagle Lake rainbow trout have gone through more than 55 years of selection for reproduction and survival under hatchery conditions for a significant part of their life cycle, it is important to start reversing that process as soon as possible. This has long been recognized, resulting in formation of the CRMP group in 1987, followed by many projects on the creek to improve flow and remove passage barriers (Pustejovsky 2007). Some elements of a conservation strategy for the trout should include:

1. Continued improvements to Pine Creek with the goal to ultimately turn it back into a perennial stream for more of its length, following the recommendations in Pustejovsky (2007).
2. Release of early-spawning adults into the creek above the weir, to maximize potential for natural migration and spawning.
3. Establishing an annual trapping and trucking operation for both adults and out-migrating juveniles until there are signs the population is self-sustaining and habitat has improved.
4. Developing Bogard Spring Creek into an experimental spawning stream by constructing a weir/trap at the lower end and then eradicating brook trout through electrofishing and other means.
5. Conducting annual monitoring of fish populations in the creek to determine spawning success.
6. Continuing to monitor habitat improvement and livestock use in the watershed.
7. Development of an eradication strategy for brook trout using either piscicides or other means. A first step is to conduct a thorough investigation of the aquatic insect fauna of the creek to determine potential impacts of piscicides on the insects.
8. Acquisition of the water rights to Eagle Lake water being diverted through Bly Tunnel and shut off the flows completely, at least during times of drought when lake levels are dropping.

Trends:

Short term: The population appears to be stable because it is maintained entirely by hatchery production. The hatchery program maintains a sport fishery for the trout in the lake and keeps them as important player in the Eagle Lake ecosystem.

*Long term***:** Two major factors affect the long-term persistence of Eagle Lake rainbow trout as a wild fish. The first, as discussed above, is the complete dependence of this fish on hatcheries for persistence. While the effects of hatchery rearing on trout populations are sometimes overstated, there is also ample evidence that it does have an impact on the genetics and behavior of fish released into the wild (e.g., Waples 1999, Araki et al. 2007), affecting their ability to persist on their own. The second is climate change, which could further affect the hydrology of Pine Creek, making it less suitable for trout migration and spawning.

Status: 2. Eagle Lake rainbow trout are likely to go extinct as even a potential wild fish in the next 50 years if present trends continue, absent successful actions to restore a naturally spawning population (Table 2). Extinction will occur because continued hatchery selection is likely to select against the ability of the fish to maintain a natural life history or because of elimination of hatchery stock through a disease epidemic. The Eagle Lake rainbow trout is not formally listed by either state or federal governments. A petition for listing it as a threatened species was rejected by the USFWS in 1994 (Federal Register 60 (151):401: 49-40150, August 7, 1994). A similar petition was rejected by the State Fish and Game Commission in 2004. The Eagle Lake rainbow trout is currently regarded as a Species of Special Concern and a Heritage Trout Species by the California Department of Fish and Game and an R5 Sensitive Species by USFS. The status score awarded here stems from the fact it is effectively extinct as a wild fish today and that continued reliance on hatchery production for its existence may assure that it will be incapable of becoming re-established.

Metric	Score	Justification
1A Area occupied (1-5)	1	Only one watershed
2 Effective pop. Size (1-5)	4	Includes hatchery fish; if only wild fish included the score would be 1.
3 Intervention dependence (1-5)	2	Persistence depends on trapping wild fish for hatchery spawning and rearing.
4 Tolerance (1-5)	4	One of most tolerant, long-lived kinds of trout
5 Genetic risk (1-5)	3	Hatchery rearing presumably has changed genetics; accidental hybridization in hatcheries possible
6 Climate change (1-5)	1	Reduced stream flows or increased alkalinity of lake could endanger fish further.
Average	2.5	15/6 Score would be 12/6, average 2, if only wild fish considered in calculation.
Certainty (1-4)	3	Well documented although limited peer review literature

Table 2. Metrics for determining the status of Eagle Lake rainbow trout, where 1 is poor value and 5 is excellent.

LAHONTAN CUTTHROAT TROUT
Oncorhynchus clarki henshawi

Description: Coloration of Lahontan cutthroat trout is variable but the back usually ranges from a greenish-bronze color to dark olive, with the sides yellowish often with a tinge of pink along the lateral line and on the cheeks. They are marked with large rounded black spots, which are fairly evenly distributed over the back, sides, and caudal peduncle. They possess 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). Gill rakers are 21-28, averaging 23-25 and there are 40-70 pyloric caeca, generally 50-60. Teeth are present and well developed on the upper and lower jaws, head and shaft of the vomer, palatines, tongue, and basibranchial bones. Scales are generally smaller than those of rainbow trout with 150-180 in the lateral line. Parr possess 8-10 narrow parr marks along the lateral line that are narrower than the spaces in between them (Behnke 1992, 2002, Moyle 2002).

Taxonomic relationships: Lahontan cutthroat trout (LCT) are divided into three distinct population segments based on geographic distribution, ecology, behavior and genetics (Behnke 1992, 2002): 1) the Western Lahontan basin segment comprised of fish in the Truckee, Carson and Walker river basins (California and Nevada); 2) the Northwestern Lahontan basin segment comprised of fish in the Quinn River, Black Rock Desert, and Coyote Lake basins (Oregon and Nevada), and 3) the Humboldt River basin (Nevada) segment. These populations are formally recognized by the USFWS as Distinct Population Segments for management (Trotter 2008). Lahontan cutthroat trout are one of the most genetically distinct cutthroat trout subspecies, reflecting long isolation. *Because only the western Lahontan segment occurs in California, this account deals only with it*. However, populations in Nevada and Oregon have similar life histories, trends, and status, so much of what is in this account applies to them as well.

Life History: Lahontan cutthroat trout 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. They are or were found in large, low gradient rivers such as the Humboldt River, moderate gradient streams such as the Carson and Walker Rivers and small, headwater tributary streams such as Donner and Prosser Creeks. LCT generally inhabit well vegetated cold-water streams with plenty of available cover (USFWS 1995). A variety of riffle-run-pool habitat provides both cover and food. 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.

Both riverine and lacustrine LCT are obligate stream spawners. Spawning takes place in streams from April to July depending on stream flow, water temperature and elevation, However, autumn spawning runs have been reported from some populations (USFWS 1995). Spawning migrations are observed at water temperatures between 5 and 16° C (USFWS 1995). Female LCT reach reproductive maturity at age 3 to 4 years while males mature at 2 to 3 years of age. 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 1995). LCT generally live 4 to 9 years with stream-dwelling fish having shorter life spans than lake-dwellers. There is evidence that Independence Lake LCT may live as long as 13 years (William Somer, CDFG, pers. comm.).

Lacustrine females have higher fecundity rates than do riverine females based on weight, length 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. Courtship and spawning in LCT involves pairing-up, digging a redd in gravel substrate and defending the redd against intruders. Spawning fish in particular develop bright red coloration on the underside of the mandible and on the operculum. Coloration is more intense in males which also show changes in shape of the lower jaw during spawning. Eggs hatch after 4 to 6 weeks depending on water temperature and fry emerge from the gravel after 13 to 23 days (USFWS 1995). Fry can spend up to two years in their natal stream before migrating to the lake environment, but most move into the lake at the end of their first summer (Trotter 2008). Growth varies with water temperature and food availability. Faster growth occurs in larger warmer waters, particularly when forage fish are an available food source. Sigler et al. (1983) measured Pyramid Lake LCT and found 217, 291, 362, and 431 mm mean fork lengths in fish aged 1, 2, 3, and 4 years old respectively. In smaller, colder water bodies, growth is slower and longevity is lessened. 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.

Lacustrine trout are capable of making extensive migrations to spawning areas. Trotter (2008) indicates that some trout from Pyramid Lake in Nevada ascended the Truckee River to spawn in tributaries to Lake Tahoe.

Habitat Requirements: LCT are very adaptable and can tolerate a wide variety of habitats and temperatures. The Lahontan basin ranges from tiny alpine headwater streams, to large valley-bottom rivers with lakes ranging from oligotrophic alpine to terminal alkaline basins (USFWS 1995). In streams, substrate composition, cover, geomorphology, and water quality are important components in LCT distribution. LCT require sufficient flows and gravel substrate to dig redds in potential spawning habitat. 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. LCT are noteworthy for their ability to survive in desert streams where water temperatures may exceed 27° C for short periods and can fluctuate 14-20° per day. They can survive prolonged exposure at 23-25° C but cease to grow when temperatures exceed 22-23° C (Dickerson and Vinyard 1999, Moyle 2002). However, ideal summer temperatures for growth and development average 13° C ± 4° C (Hickman and Raleigh 1982). LCT prefer streams with well vegetated and stable stream banks, greater than 50 percent of the stream area providing cover, and pools with close proximity to cover as well as riffle-run complexes for spawning and cover (USFWS 1995).

Lacustrine LCT are adapted to a wide variety of lake habitats but have a considerably higher tolerance for alkalinity and total dissolved solids than most freshwater fish. They can withstand alkalinity levels as high as 3,000 mg/L and dissolved solids as high as 10,000 mg/L (Koch et al. 1979). Optimal lacustrine habitat should have an average mid-summer epilimnion temperature of less than 22° C and a mid-epilimnion pH of 6.5 to 8.5. According to Koch et al. (1979), LCT require ≥ 8 mg/L dissolved oxygen (DO) in the epilimnion. However, there are anecdotal accounts that LCT are capable of tolerating lower DO levels. According to William Somer (CDFG, pers. comm.), fall algal die-off at Heenan Lake triggered low dissolved oxygen conditions throughout the lake which resulted in a major fish kill, although several thousand LCT survived the event. Much of the lake had almost no measurable DO, so presumably the

LCT survived by finding oxygenated springs or getting oxygen near the surface (William Somer, CDFG, pers. comm.).

Distribution: Lahontan cutthroat trout are native to the greater Lahontan basin in eastern California, southern Oregon and northern Nevada (Trotter 2008). In California, they were historically found only in the Carson, Walker, Truckee, and Susan River drainages on the east side of the Sierras. In the early 19th century, Lahontan cutthroat trout were abundant and widespread in this range. 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. In the Truckee Basin, LCT from Pyramid Lake apparently 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 contain LCT (Table 1); they had disappeared from the Susan River drainage by about 1900 (Trotter 2008). LCT have also been planted and become established in a few creeks outside their historic range including west slope drainages near the Truckee basin. USFWS (1995) offers the following lakes and streams in California as supporting current or recently existing populations:

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 Independence Creek Pole Creek*** Upper Truckee River*	Murphy Creek** Slinkard Creek** Bodie Creek* Mill Creek** Wolf Creek** Silver Creek ** By-day Creek *
*Reintroduced populations of Independence Lake strain, actively managed by CDFG	*Reintroduced population above barriers after chemical treatment ***Native population was introgressed by brook and rainbow trout, stream was chemically treated and reintroduced with LCT from Macklin Creek in 1977	*Historic population maintained above a barrier **Historic populations once extirpated but reintroduced

Table 1. List of known populations of Lahontan cutthroat trout in their native range in California; from USFWS (1995).

Yuba River Drainage	Stanislaus River Drainage	Mokelumne River Drainage	San Joaquin River Drainage	Owens River Drainage
Macklin Creek East Fork Creek Unnamed tributary to East Fork Creek	Disaster Creek	Marshall Canyon Creek Milk Ranch Creek	West Fork Portuguese Creek Cow Creek	O'Harrel Creek

Table 2. Known populations of LCT outside their historic native range; from USFWS (1995).

Water Stocked with Yearlings	Waters Stocked with Fingerlings	
Heenan Lake Red Lake Martis Creek Reservoir Kirmen Lake Sagehen Creek June Lake Truckee River Crowley Lake	Angora Lake Upper Blue Lake Upper Crowley Lake Echo Lake Upper Francis Lake Lane Lake Meadow Lake Penner Lake Roosevelt Lake Round Top Lake Showers Lake Twin Lake	Birch Lake Coldstream Creek Pond Echo Lake Lower Eileen Lake Kirmen Lake Martis Creek Reservoir McCleod Lake Red Lake Round Lake Scotts Lake Tamarack Lake

Table 3. Lakes and streams stocked with hatchery LCT in recent years by CDFG (William Somer, CDFG, pers. comm. 2007). Stocking with LCT does not mean populations are established in these waters.

Abundance: There are only 17 lakes and streams that are known to still contain LCT within their historical range in California (Table 1). In addition, there are introduced populations of LCT in nine creeks outside of their native range (Table 2). While population estimates are lacking, all populations most likely contain less than 200 adult fish, given habitat availability.

A major source for planting fish into lakes and streams in Calfornia (Table 3) is Heenan Lake, which was stocked in 1935 with LCT from Blue Lakes, Alpine County (in the headwaters of the Mokelumne River). LCT were introduced into the originally fishless Blue Lakes during 1864, presumably from the West Carson River at Hope Valley. During 1873, rainbow trout from the North Mokelumne River were stocked in Blue Lake. A. J. Calhoun of CDFG noted the presence of rainbow-cutthroat hybrids in 1940 in Blue Lake. Because of this hybridization problem, a decision was made by DFG to shift away from using Heenan Lake strain fish to using LCT from Independence Lake (LCT-I). A plant of 5,000 marked yearlings collected from spawners in Independence Lake was made into Heenan Lake during 1975. This plant began a phasing out of the original earlier strain. Since 1980 only LCT-I were used in the hatchery program (William Somer, CDFG, pers. comm.). As of 1999, ten populations of LCT had been established throughout their native range; however, all but one of them suffer from geographical isolation and small population sizes. While LCT have often persisted in isolation throughout their history, it has never been to the degree that currently exists and extant populations are not self-sufficient (i.e., most are maintained by stocking or managed for non-native species control). Avoidance of genetic drift and stochastic events are significant challenges in current LCT management and, while there are a number of populations, improved connectivity among them would provide a more robust defense against such problems in the long term.

While definitive population estimates are lacking, USFWS's (1995) estimate that LCT persist in 11% of their original stream habitat and in a mere 0.4% of their original lake habitat, indicates extremely reduced populations. Wild self-sustaining populations in headwater streams of California likely total only a few hundred fish age 1+ and older. Recreational fisheries are maintained in many waters by planting broodstock, yearling, and fingerling LCT (Table 3).

Factors Affecting Status: Major factors affecting LCT habitat and abundance are: (1) introductions of non-native trout, 2) overexploitation, 3) logging, 4) dams and diversions, 5) grazing, 6) mining, 7) loss of genetic diversity, and 8) disease.

Non-native fish introductions: Lahontan cutthroat trout were the only salmonid historically found in the Eastern Sierras with the exception of Eagle Lake rainbow trout and Paiute cutthroat trout. Introductions of non-native trout species (rainbow, brown and brook trout), made to improve fisheries, added species that are intense predators on and competitors with LCT. Brown trout are voracious predators on juvenile LCT and they are fall spawners, which gives juvenile brown trout and advantage over LCT, which spawn in spring and so are smaller than the alien trout when they emerge. Brook trout are also fall spawners but their biggest advantage over LCT is the fact that they occur in much higher densities than other trout and effectively outcompete LCT for habitat and resources. Rainbow trout can hybridize with LCT and are therefore a threat to the already compromised genetic diversity of the cutthroat. When the two species co-occur, the rainbow trout phenotype eventually dominates.

Lake trout have apparently contributed to the demise of LCT from Lake Tahoe and Fallen Leaf Lake through predation and competition, and perhaps through disease. Vander Zanden et al. (2003) indicate that the food webs of Lake Tahoe are now so altered, thanks to introduced species, that re-establishing cutthroat trout in the lake may not be possible. More recent introductions of centrarchids and other non-native fish and invertebrates will hamper LCT restoration efforts even further (William Somer, CDFG, pers. comm.).

Overexploitation: Heavy commercial fishing in lacustrine populations in the 19th and early 20th centuries was a major factor contributing to the extirpation of LCT from Pyramid Lake and Lake Tahoe (Trotter 2008). 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. By the 1940s LCT were extinct in Pyramid Lake and hatcheries were then required to support the popular sport fishery there. Those populations continue to require complete hatchery support.

Logging: The watersheds containing LCT were heavily logged in the 19th century to provide timber for mines in Nevada and for railroad ties, denuding large areas of vegetation and increasing silt loads in the rivers (Trotter 2008). In many streams, water 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 then flooding the river destroyed habitat and depleted the fish populations (USFWS 1995). From the 1860s through the 1890s, sawmills along the Truckee River discharged large amounts of sawdust and wood chips into the river.

Following that tradition, industrial and sewage wastes were directly dumped into the river until the 1930s (USFWS 1995).

Dams and diversions: Dams are present on most major cutthroat streams, fragmenting habitats, creating barriers to migration, and creating large areas in reservoirs and regulated rivers that are unsuitable as habitat for LCT. In addition, diversions decrease the flows of many streams. Agricultural water diversions (mainly from Derby Dam, built in 1905) in the lower basin effectively disconnected Pyramid Lake from the Truckee River for the better part of the 20th century, resulting in lake levels dropping nearly 24 m and alkalinities greatly increasing. As the result of federal listing of the cutthroat and 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. The Walker Lake population persisted until Bridgeport Dam was built upstream in 1924, followed by the final blow of Weber Dam in 1933, effectively cutting the population off from its spawning habitat. The present population of Lahontan cutthroat trout in Walker Lake is supported entirely by the planting of hatchery fish, the progeny of the last 39 trout that attempted to spawn below the dam in 1949 (Trotter 2008).

Grazing: Heavy grazing by livestock throughout LCT range, especially of cattle in riparian zones, has degraded habitat of LCT streams, through trampling of banks and riparian vegetation, leading to erosion, incision and siltation of the stream. Further, the loss of riparian vegetation and cover has resulted in higher water temperatures and reduced cover, leaving fish more vulnerable to predators. As much as 70% of LCT habitat occurs on BLM and National Forest lands where heavy grazing has historically been permitted. Public lands are far less heavily grazed today, but active grazing occurs throughout LCT range. Numerous studies point to the negative impacts of cattle in riparian areas and how reduction in this impact can result in significant increases in trout production and biomass (Chaney et al. 1990, USFWS 1995).

Mining: The effects of historic mining on fish populations in generally underappreciated in California, in part because the most egregious effects took place during the Gold Rush era of 19th century when placer mining turned over the bottoms of streams and diverted water from them and when hardrock mining dumped debris and sediment into streams, as well as toxic drain water from the mines. All these activities took place in the eastern Sierras, affecting Lahontan cutthroat streams, but the impacts are largely unrecorded (Trotter 2008).

Loss of Genetic Diversity: When LCT populations crashed in the early part of the 20th century, many genes were lost because only limited stocks were selected for hatcheries or survived in isolated streams. Because LCT historically inhabited many isolated subbasins, there were presumably many genetically distinct populations with local adaptations. USFWS (1995) thus recommended that, as much as possible, genetic stocks should be maintained in their basins of origin. The full morphological and genetic differentiation in remaining LCT stocks is not fully understood but must be protected as much as possible. Lacustrine LCT are the most at risk because there are only two small naturally reproducing populations left within their native range, only one in California. The two populations (in Summit and Independence lakes) are distinctive genetically. Native populations from Pyramid, Walker Lake and Lake Tahoe are now extinct (USFWS 1995) although hatchery strains of Pyramid and Walker Lake fish still persist.

The Lahontan Basin was formerly a network of interconnected streams and rivers which allowed for genetic exchange between separate but interconnected sub-populations of LCT, collectively called metapopulations. The Basin has changed so dramatically as the result of human use dams, 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 on non-native trout together effectively eliminate the possibility for recovering historic self-sustaining natural meta-populations throughout most of their range.

Disease: A number of parasites and pathogens have potentially adverse effects on LCT and their recovery. 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 historic range of LCT (Jon Stead, UC Davis, pers. comm. 2007). While the bacterium is widespread in wild brook, brown and rainbow trout, these fishes do not show any sign of being diseased such as decreased fitness or condition (Jon Stead, UC Davis pers. comm. 2007). The Lahontan National Fish Hatchery in Nevada was forced to euthanize 400,000 fish after an outbreak of furunculosis, caused by the bacteria *Aeromonas salmonicida,* in the winter of 1999/2000. An additional 200,000 were treated with antibiotics and eventually released into Pyramid Lake for the sports fishery there. Other concerns include whirling disease, and the bacterial gill diseases *Ichthyophthirius multifilis* and Costia (*Ichtyobodo necatrix* and *Ichtyobodo pyriformis*). *Nucleospora salmonis* (microsporidia), which causes a leukemia-like condition with anemia, swollen kidneys, and spleen, has infected LCT in CDFG production facilities and has reduced the ability of DFG to plant these fish into waters of the state (William Somer, CDFG, pers. comm. 2007).

Conservation: Lahontan cutthroat trout (LCT) were listed as federally endangered in 1970 and were subsequently relisted as federally threatened in 1975 to allow regulated angling and to facilitate management activities (USFWS 1995). There have been considerable efforts at restoring populations of LCT in their native range. Hatchery propagation of LCT has been ongoing since around 1939 and continues. The Lahontan National Fish Hatchery releases approximately 500,000 fish per year. CDFG and USFWS have spent considerable resources in maintaining genetic diversity and have begun reintroducing LCT in numerous locations. Habitat alteration, abundant alien trout, and the loss of interconnected metapopulations has left the USFWS in the unenviable position of trying to recover a species with very little habitat available for new populations. Thus, persistence will require innovative management, habitat restoration, and elimination of competing species of trout from streams.

Independence Lake, in the Truckee River drainage, is the only lacustrine population where LCT have continuously survived and reproduced independently. Gary Scoppettone (USGS, pers. comm. 2007) has ongoing monitoring and assessment studies in the lake and its inlet. LCT have somehow managed to persist in Independence Lake despite stocking of brook, rainbow and brown trout, as well as kokanee salmon, since around 1931 (William Somer, pers. comm. 2007). Currently the lake is managed for LCT only and all other alien fishes are actively suppressed. LCT from Independence Lake are managed as a brood stock in Heenan Lake and this population has provided fish for hatchery programs and ongoing restoration attempts.

Efforts to protect the endangered cui-ui have resulted in increased flows in the Truckee River thus raising the lake level of Pyramid Lake, Nevada, reducing its alkalinity, and providing access to the Truckee River for spawning, benefiting 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, although it is not known to what degree of, if any, spawning is successful (Gary Scoppettone, USGS, pers. comm. 2007).

In the Upper Truckee River basin, CDFG has worked to restore LCT since the 1980s, with an emphasis on brook trout eradication. During 1988 through 1990 CDFG treated the headwaters with a piscicide (rotenone) and continued treatments for 3 years. LCT were then successfully restocked in the streams. Unfortunately, brook trout were rediscovered in 1995, a suspected illegal reintroduction. Eradication was unsuccessful the second time and now CDFG efforts are aimed at containment and suppression. Ongoing efforts to control brook trout by electrofishing have been successful at keeping populations low enough to not adversely impact reintroduced LCT, but that method is extremely labor-intensive. Electrofishing has be done each year to keep brook trout under control and preserve that small headwater reach for LCT. CDFG is on the verge of eradicating brook trout from the stream with a total catch of five brook trout in 2005 and only two in 2006 (William Somer, CDFG., pers. comm., 2007). Somer also estimates there to be approximately 2000 adult LCT occupying 4.5 miles of habitat in Meiss Meadows on the Upper Truckee watershed. Currently CDFG plants thousands of LCT yearlings and fingerlings in lakes throughout the Truckee, Carson, and Walker basins; however, many of these lakes also contain alien trout or do not have the habitat to support reproducing populations, so these populations are not expected to persist and reproduce naturally (Table 3, William Somer, pers. comm. 2007).

LCT face tremendous odds in the recovery process and despite the combined work of agencies and other researchers, it is unlikely that they will be delisted any time in the near future. Criteria for delisting LCT in the USFWS recovery plan include maintenance of adequate population sizes, protection of existing habitat, and monitoring and protection of existing populations. The USFWS is currently undergoing a 5-year review of LCT status that will ultimately lead to a revised recovery plan, but not necessarily to recovery, given the obstacles. Habitat fragmentation and alien trout invasion has occurred throughout LCT range and the resources, political will and time required to reverse those impacts make it almost impossible to do. All of the reintroduced populations exist because LCT were placed in fishless waters above barriers or because nonnative fish were removed using piscicides. CDFG restoration goals are to protect and expand existing wild populations of LCT, using existing populations as sources for reintroduction. Current management is focused primarily on maintaining genetic diversity and reintroducing LCT in streams and lakes with good potential for success. The lack of interconnected watersheds to support metapopulations, however, will ensure that these fish will not persist without significant support from managers. It is likely that with the continued efforts of CDFG, FWS 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 true, self-sustaining populations.

For additional comments and history of Lahontan cutthroat trout conservation and management see Trotter (2008).

Trends:

Short term: The LCT population appears to be fairly stable due to considerable hatchery production, but hatchery fish exist primarily to provide recreational fishing opportunities and are only rarely used in recovery efforts. Hatcheries produce hundreds of thousands of fish per year, but natural reproduction is limited to only a handful of small streams and lakes (Table 1 and 2), and several of them are outside the fish's historical range. Independence Lake contains the only continuously existing lacustrine population of LCT, but that population is small due to pressures from introduced fishes.

Long term: Persistence of LCT in the wild in the next century will require continued intense management and wild populations are likely to remain small and scattered. The same hurdles faced for the short-term conservation of LCT also exist in the long term, including genetic issues, alien trout, and habitat fragmentation and alteration. Additionally, climate change may adversely impact LCT by increasing stream temperatures and causing lower or no flows in some small streams. Hatcheries will likely maintain sport fisheries for LCT so the fish will probably persist in the basin as long as hatcheries are funded.

Status: 2. Lahontan cutthroat trout in California (essentially the Western Lahontan Basin DPS) have a high likelihood of disappearing in the next 50 years, except as populations sustained by hatchery production. They are currently listed as a threatened species under the federal Endangered Species Act (Federal Register Vol. 40, p. 29864, 1975). They are not formally listed by the state of California, but are managed as a heritage trout species. Wild populations of Lahontan cutthroat trout in California are small and isolated and require continuous management to prevent extinction. While the populations present in Nevada and Oregon reduce the probability of extinction of the subspecies throughout its range (but not the DPS), these populations suffer from similar impacts as those in California. Continued planting of hatchery fish may sustain the species and support fisheries as long as the California hatcheries are maintained, but such populations have evolutionary trajectories distinct from wild populations.

Metric	Score	Justification
Area occupied	3	Occupies multiple watersheds in California, but no connectivity
Effective population size	3	Wild populations have <1000 fish each
Intervention dependence	2	Hatchery program using wild brood stock required for persistence
Tolerance	5	LCT are fairly long-lived and demonstrate broad physiological tolerances. They are also iteroparous.
Genetic risk	1	Hybridization risk and loss of genetic variation is well documented
Climate change	1	LCT are vulnerable to climate change in all watersheds inhabited
Average	2.5	15/6
Certainty (1-4)	4	Reports concerning this risk level are found in peer reviewed literature

Table 4. Metrics for determining the status of wild Lahontan cutthroat trout in which 1 is a poor value and 5 is excellent.

PAIUTE CUTTHROAT TROUT
Oncorhynchus clarki seleniris

Description: Paiute cutthroat trout (PCT) and Lahontan cutthroat trout (LCT, *O. c. henshawi*) are morphometrically and meristically identical. However, while LCT are heavily spotted (particularly below the lateral line) and are bronze to olive in coloration, PCT are virtually spotless, have iridescent copper, green, or purplish-pink body coloration, and retain their parr marks into adulthood (Moyle 2002, USFWS 2004). Although originally described by Snyder (1933) as being without spots on the body, most Paiute cutthroat trout have 1-5 small spots, with a few having up to 9 spots (USFWS 2004). They possess the characteristic cutthroat red 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, Nielsen and Sage 2002, USFWS 2004).

Taxonomic Relationships: The Paiute cutthroat trout is closely related to Lahontan cutthroat trout of the Carson River, from which it has been isolated for approximately 8,000-10,000 years (Behnke 2002). Snyder (1933, 1934) 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 clarki seleniris* (USFWS 2004). Investigations of genetic structure of populations of the Lahontan group of cutthroat trout (Lahontan cutthroat trout, Paiute cutthroat trout, and Humboldt cutthroat trout), using microsatellite DNA testing, detect no unique alleles in Paiute cutthroat, but do show that these fish went through a severe genetic bottleneck (Israel et al 2002, Nielsen and Sage 2002).

Life History: Paiute cutthroat trout are so similar to LCT that we can assume that their life history is similar to that of LCT in small, cold headwater streams. None of the PCT populations occur in areas that have the extremes in temperature observed in some of the LCT habitat and it is unknown if they have the capacity to survive the levels of alkalinity, turbidity and temperature that LCT can withstand. Descriptions of LCT life history are presented in the account in this report and in Moyle (2002) and Behnke (2002). There are no known naturally occurring lake populations of Paiute cutthroat trout, although several attempts have been made in the last century to establish them in lakes outside their historical range, with limited success.

Surprisingly little research has 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 D. Wong of the California Department of Fish and Game (USFWS 2004). PCT life expectancy is quite low and few survive beyond 3-4 years of age in the wild (in streams), which gives them just 2 years of potential spawning activity (Wong 1975, USFWS 2004). Sexual maturity is reached at 2 years of age and peak spawning activity takes place during the months of June and July (USFWS 2004). Mature fish are 15-25 cm TL. Females use their tails to dig redds in clean gravel substrate in which they bury the fertilized eggs. The embryos hatch in approximately 6-8 weeks, and spend an additional 2-3 weeks in the gravel as alevins before emerging as fry. The juvenile fish rear in backwaters, shoals and small tributaries until they reach approximately 50 mm TL (Wong 1975, USFWS 2004). Adult fish establish dominance hierarchies and defend their established territories from intruders. The larger fish dominate the more desirable pool habitats and smaller fish are relegated to riffle and run territories (USFWS 2004). They require pools for overwintering habitat and are vulnerable to ice scour (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 are dependent on water temperature, stream size, and food availability. Few PCT reach lengths over 25 cm, and the largest recorded PCT in Silver King Creek is 34 cm (USFWS 2004). The largest PCT was caught in a lake at 46 FL cm (weight, 1.1 kg) although such fish apparently do not reproduce (USFWS 2004, Behnke 2002).

Habitat Requirements: The only studies of PCT habitat requirements and preferences are of introduced populations in the North Fork of Cottonwood Creek (Wong 1975, USFWS 2004). PCT seem to have similar requirements to other alpine stream trout: 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).

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 of the Carson River. PCT historical distribution is exceedingly limited. They are thought to have existed in only 14.7 kilometers of habitat from the base of Llewellyn Falls downstream to Silver King Canyon and 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 suffered introductions of rainbow trout (*O. mykiss*) as well as of Lahontan cutthroat trout and golden trout (*O. m. aguabonita*). A paucity of records and conflicting recollections of the Silver King Basin's early settlers has made the early history and distribution of this fish difficult to grasp, but to the best of our knowledge, the first transfer of fish out of their historical range took place in 1912 by Joe Jaunsaras, a Basque herdsman working for Virgil Connell, an early grazing permittee in the basin (USFWS 2004). According to Connell (in Ryan and Nicola 1976), the unspotted Paiute trout increased in numbers above the falls ". . . until in 1924 the stream was so well-stocked, that fishing above the falls was better than below." Connell also 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." A conflicting view of the story comes from Joe Jaunsaras' brother, John Jaunsaras, who reported to Ashley (1970) that the first transfer was a failure and in 1924, he and another man carried 75 5-gallon buckets of trout upstream around the falls. Either story may be true, but it is highly likely that by 1924, PCT below the 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 was in the basin. His conjecture was that French-Canadian loggers who worked the area in the 1860s had brought the fish up from lower in the basin (Ashley 1970). No records of the fauna of the two creeks exist before then and there are falls near the mouth of Corral Creek that were presumably a historical barrier to fish. Vestal (1947) made the first documented collections of PCT there in 1947 and thought the streams to be "…formerly barren of fish life," and he attributed the fish's presence there to be a result of herdsmen in the basin who ". . . reportedly planted Piute (sic) trout a few at a time in buckets from Upper Fish Valley"(USFWS 2004).

Over the course of the last century, many transfers were made outside the Silver King Basin. The first transfer was to Leland Lakes in 1937 and failed, probably due to the presence of other salmonids. Next a collection of about 400 fish was taken to the North Fork of Cottonwood Creek, a high elevation spring-fed creek in the White Mountains, in Mono Co., California. That population persists to this day and has been a source population for reintroduction of genetically pure PCT. Introductions continued at McGee Creek (1956) and Delaney Creek (1966), but both were unsuccessful. Other failed introductions occurred in Bull Lake in 1957 and Heenan Lake (already a hatchery for LCT) in 1983. The only self-sustaining lacustrine population of PCT was located in Bircham Lake in Inyo County (planted in 1957), but by the early 1980s, D. Wong found this population to be highly introgressed with rainbow trout (USFWS 2004). Of the 10 known introductions of PCT, there are reproducing populations established only in Cottonwood Creek (Mono Co.), Cabin Creek (Mono Co.), Stairway Creek (Madera Co.), and at the outflow of Sharktooth Lake (Fresno Co.).

Table 3. Known introductions of Paiute cutthroat trout in California.

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	

Abundance: USFWS (2004) estimated that Paiute cutthroat trout occupy a minimum of 33.2 km of stream habitat in five widely separated drainages. No PCT currently occupy the historic habitat below Llewellyn Falls, although there are several tributary creeks in the Silver King basin that now contain transplanted populations of PCT (USFWS 2004). Within the Silver King drainage, PCT are thought to occupy a total of 18.6 km of stream with a core habitat of 12.9 km (USFWS 2004). CDFG population assessments found approximately 1,020 adult fish in six streams in the Silver King Drainage (USFWS 2004). While most populations are stable, they remain heavily fragmented and have no chance of interbreeding without human intervention, thus reducing the effective population size and seriously limiting the genetic viability of the species.

According to the USFWS Recovery Plan (2004), there are approximately nine streams and lakes that currently hold pure Paiute cutthroat trout. The results of a CDFG 2001 population survey in the Silver King drainage above Llewellyn Falls estimated ~424 fish in the reach, an average number over the years that indicates the population is either stable or growing (USFWS 2004). Four Mile, Fly Valley, and Corral Creeks have all had numerous population surveys and those populations appear to have long term stability (despite some fluctuations) with an effective population of between 400 and 700 fish (USFWS 2004). The out-of-basin streams with PCT populations (N.F. Cottonwood Creek, Cabin Creek, Stairway Creek, and Sharktooth Creek) have been surveyed by either visual assessment or fly fishing (to prevent injury or mortality from electrofishing), so the estimates are population minimums rather than a true population counts. They appear to all be stable with ~1-4 km of habitat available in each stream. PCT were originally planted in Sharktooth Lake, but now are found only in its outlet creek. All other lacustrine introductions have failed.

Factors affecting status: PCT are relatively stable in their numbers, but they have been extirpated from their historic native range and persist only where introduced (15 km of stream in Toiyabe National Forest, 8 km in Sierra and Inyo NF). The biggest threats to the persistence of PCT include 1) alien trout, 2) loss of genetic diversity, and 3) habitat loss.

Alien trout: Alien trout are the principal threat to PCT. They impact PCT through competition for resources and habitat, predation, and hybridization. The introduction of non-native rainbow, golden and Lahontan cutthroat trout into the historic range of the PCT below Llewellyn Falls has resulted in the extirpation of PCT from their historic range. PCT readily hybridize with rainbow trout and Lahontan cutthroat trout, resulting in loss of genetic integrity and phenotypic distinctiveness.

Loss of genetic diversity: A study by Cordes et al (2004) found PCT to be the most genetically limited and narrowly distributed native trout in California. Genetic distances among the current populations of PCT show that there are three genetic groups within PCT, but Cordes et al (2004) surmise that this differentiation is due to founders effects and genetic drift, which are more the result of stocking histories than natural variation within the subspecies. Most of the transfers consisted of small numbers of fish and the creeks with similar genetic strains had similar stocking histories (Cordes et al 2004). Nielsen and Sage (2002) found no distinctive alleles differentiating PCT from LCT, but did see that PCT had gone through a major genetic bottleneck. It is likely that the first isolation of PCT when it diverged from LCT represented a genetic bottleneck and that subsequent stocking and culling of stocks to eliminate hybridization has further amplified the situation. Additionally, there is no population that currently possesses all of the alleles known to PCT, so further transfers to maintain what is left of genetic diversity may be required. Loss of genetic diversity due to small populations and lack of metapopulation connectivity combined with introgression represents the largest threat to PCT.

Habitat loss: At this time (2008), the 23 km of stream habitat in which PCT persist are in reasonably good condition, with limited or no grazing affecting the stream banks and light human use. However, this could change if grazing allotments are renewed or if a catastrophic fire swept through one or more of the basins. Thus the limited habitat by itself represents a reason for careful management because conditions can change so quickly.

Conservation: The populations of Paiute cutthroat trout are small but reasonably stable because they are entirely located in streams in remote national forest lands (Toiyabe, Sierra, and Inyo national forests). The habitat closest to their native range is all in the Carson-Iceberg Wilderness as well. However, additional protection is needed by expanding their range back into their historic habitats.

Part of the management actions listed in the 2004 USFWS Recovery Plan is removal of non-native trout in the waters between Llewellyn Falls and Silver King Canyon. This was to be done using the piscicide, rotenone, which has resulted in years of lawsuits and no progress in restoring PCT below Llewellyn Falls. Opposition to the poisoning of the creek comes in part from anglers who value the healthy fishery of wild (though non-native) trout that thrives in the excellent habitat below the falls. There is also concern that endemic invertebrates may occur in that reach and rotenone is toxic to them as well as fish. Additionally, the area is habitat for two species of amphibians that are candidates for listing under the Endangered Species Act, the mountain yellow-legged frog (*Rana mucosa*), and the Yosemite toad (*Bufo canorus*); there is fear that the rotenone treatments could harm them, though the USFWS plans include removal of any amphibians found in the reach prior to treating the creek (USFWS 2004). Recent reviews of toad sampling indicate that most toads in the Silver King basin are western toads (*Bufo boreas)*, although a few hybrids with Yosemite toads may exist. No pure Yosemite toads have been found in the Silver King Creek basin (William Somer, DFG, pers. comm.). It is unlikely that the candidate species amphibians occur in high densities below Llewellyn Falls because non-native trout predation is one of the main factors in their decline throughout their range. The current distribution of PCT and mountain yellow-legged frogs overlap in Silver King Creek basin. The two organisms co-evolved in the basin, so it is likely they can coexist. The two species also co-occur in all four of the out of basin populations of PCT (N.F. Cottonwood Creek, Cabin Creek, Sharktooth Creek, and Stairway Creek). The Recovery Plan includes treatment of Tamarack Lake (in the upper Silver King drainage) to rid it of trout for the benefit of the two amphibian species (USFWS 2004).

PCT have had a complicated stocking history in the last 150 years and have been subject to a variety of management actions (Table 2).The many unauthorized transfers both of PCT and the non-native trout that threaten them have been both a scourge and a savior. By 1924, the PCT in their native reach (below Llewellyn Falls) were already introgressed with LCT, rainbow trout, and golden trout. In 1949, another unauthorized transfer introduced rainbow trout above the falls. If it had not been for the 1946 stocking in Cottonwood Creek and introduced populations within the Silver King Basin in Fly Valley and Four Mile Creeks, PCT could well have been lost. Introductions, hybridization, and culling have occurred repeatedly throughout the 20th century. Current PCT populations are fairly stable, but the lack of genetic diversity of existing stocks, extirpation from their native range, lack of connectivity among populations, and small effective population sizes continue to hamper recovery efforts. The 2004 PCT Recovery Plan lists reintroduction of PCT to their native range below Llewellyn Falls and eradication of non-native salmonids there as one of the criteria for delisting the species. Efforts on the part of CDFG, US Fish and Wildlife Service, and US Forest Service to eradicate alien trout in lower Silver King Creek with piscicides have been blocked by litigation for several years, but USFWS is in the process of completing an environmental scoping document to move forward with a chemical treatment. If treatments are successful, then the process of restocking PCT in lower Silver King Creek can begin. Further conservation plans by CDFG and USFWS include monitoring and maintaining all existing populations of PCT and their habitat as well as continued protection of all existing populations from alien trout incursions. The fact that the historic PCT habitat is publicly owned and suffers from very limited degradation gives the species a good chance of recovery. However, the small geographic range and limited genetic diversity of PCT make it vulnerable to inbreeding depression, stochastic events, and illegal introductions of alien trout.

Restoring PCT to their historic habitat will more than double the number of fish in the Silver King basin, provide greater connectivity of habitat, restore the PCT as the principal aquatic predator, and help to isolate PCT in the basin from the threat of non-native trout introductions. In short, PCT have a good chance at recovery if their range can be expanded, and their habitat protected, especially from invasions of alien trout.

Table 4. History of Paiute cutthroat trout from mid-1800s to the present. SKC=Silver King Creek, USKC=upper Silver King Creek, COY=Coyote Canyon Creek, COR=Corral Canyon Creek, FVC=Fish Valley Creek, NFC=North Fork Cotton Creek, FMC=Four Mile Creek, CC=Cabin Creek. From Cordes et al 2004.

Pre-1860s Historical distribution of PCT in SKC from below Llewellyn Falls downstream to Silver King Canyon Gorge	1860s Fishless COR and COY believed to be stocked with PCT from SKC below Llewellyn Falls	1860s to 1912 Fishless FMC either stocked w/PCT or colonized from 1912 introduction above Llewellyn Falls	1912 PCT stocked into fishless upper SKC above Llewellyn Falls	1924 Hybrid RT/PCT and LCT/PCT noted in SKC below Llewellyn Falls	1946 NFC stocked w/PCT from USKC, COR, and COY
1947 Fishless FVC stocked w/PCT from COR and COY	1949 Unauthorized introduction of RT into USKC	1963 Hybrid RT/PCT found in COR and COY	1964 Unsuccessful chemical treatments of USKC, COR, and COY. Hybrids found in NFC below a barrier	1968 CC stocked w/PCT from NFC	1970 Unsuccessful chemical treatment of NFC
1972 SC stocked w/PCT derived from FMC	1976 Unsuccessful chemical treatments of USKC, COR, COY, and NFC	1976 RT/PCT hybrids found in USKC and NFC but not FMC	1977 Successful chemical treatment of COR, unsuccessful in COY	1978 COR stocked w/PCT from FVC	1980-83 Successful chemical treatment of NFC. Restocked w/ NFC from above barrier
1984 CC population deemed not hybridized based on allozymes	1987-89 Successful chemical treatment of COY. Restocked w/PCT from FVC	1991 COR, COY and FMC deemed not hybridized based on allozymes	1991-93 Successful chemical treatment of USKC	1994-98 USKC restocked w/PCT from FVC and COY	2004 No RT genes found in any of the PCT populations sampled by Cordes et al. 2004

Trends:

Short term: PCT populations have remained fairly stable since 1998. They currently inhabit more miles of stream than they did historically, but the populations are heavily fragmented and cannot interbreed. Non-native trout exist below Llewellyn falls and 56 years of restoration and conservation efforts could be unraveled by a single illegal introduction of non-native fish into current PCT habitat.

Long term: Though PCT have a far more limited distribution than LCT, their habitat is in reasonably good condition and is all on public land (National Forest) which simplifies recovery efforts considerably. Their dependence on humans for reproduction is limited, although continued transfers will be required in order to maximize genetic diversity in the populations. Monitoring and removal of any alien trout must continue indefinitely to protect genetic integrity of the remaining fish. Climate change may pose a threat to PCT populations, but the alpine setting of their native habitat could potentially buffer them from the effects of warming or loss of snowpack. The out-of-basin populations at Cottonwood and Cabin Creeks may be more at risk because of the arid nature of the White Mountains.

Status: 2. PCT have a high likelihood of extinction in their native range within the next 50 years without continued intense monitoring and management. All populations are small and isolated, so therefore subject to illegal introductions of alien trout and local natural and man-made disasters. PCT were listed as endangered under the Endangered Species Preservation Act of 1966 on March 11, 1967. However, they were subsequently downgraded to threatened under the Endangered Species Act of 1973 (on July 16, 1975) to facilitate management activities and to allow limited recreational fishing (USFWS 2004). The 2004 USFWS Recovery Plan's goal is to restore PCT to their native range in Silver King Creek and continue to monitor and protect all existing populations. The status determination (Table 3) is for the effective wild populations, including introduced populations.

Metric	Score	Justification
Area occupied	2	Occupies several watersheds but connectivity is non-existent
Effective population Size	3	The largest effective population may be around 1000 but most are smaller.
Intervention dependence	3	Human assistance required to maintain genetic diversity and protect its limited habitats.
Tolerance	2	Actual physiological tolerances not known but adapted for small cold-water headwater streams, which suggests limited tolerance.
Genetic risk	1	Past hybridization has reduced current population size and genetic diversity
Climate change	3	Vulnerable because streams very small and some may become dry during droughts.
Average	2.3	14/3
Certainty (1-4)	4	PCT well documented in the peer-reviewed literature and in agency reports.

Table 3. Metrics for determining the status of Paiute cutthroat trout, where 1 is poor value and 5 is excellent.

COASTAL CUTTHROAT TROUT
Oncorhynchus clarki clarki (Richardson)

Description: Coastal cutthroat trout appear similar to rainbow trout (*O. mykiss*) but can be distinguished by heavier spotting, particularly below the lateral line, as well as by spots on paired and anal fins. Background coloration can be extremely variable: spots become nearly invisible and the fish takes on the silver coloration common to other anadromous salmonids when in salt water. The body color typically has a dark coppery or brassy sheen when mature fish are found in fresh water (Behnke 1992, Moyle 2002). Cutthroat trout tend to be more slender-bodied than rainbow trout and possess characteristic red to orange to yellow slashes under the mandibles although the slashes are seldom 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 on the basibranchial bones. 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 the 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 are difficult to distinguish from rainbow trout parr. 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 individuals from landlocked populations to exceed 30 cm FL.

Taxonomic Relationships: Behnke (1992, 1997) proposed that rainbow and cutthroat trout both evolved from a common trout ancestor somewhere in what is now the Columbia/Snake River basin near the beginning of the Pleistocene epoch, approximately 2 million years ago. He indicates that approximately 1 million years ago, cutthroat diverged again into two major groups, the coastal type (*O. c. clarki*) and the westslope type (*O. c. lewisi*). The coastal type is characterized by 68 or 70 chromosomes and the westslope types are characterized by 66 chromosomes (Behnke 1992, 1997). The coastal group has remained essentially intact and colonized coastal rivers from northern California to Prince William Sound in Alaska. The westslope type diverged again with a 64 chromosome type that has been isolated into two "major" subspecies, the Lahontan cutthroat trout (*O. c. henshawi*) found in the western Great Basin and the Yellowstone cutthroat (*O. c. bouvieri*) found in the Snake River basin (Johnson et al. 1999). Behnke lists 14 extant subspecies of cutthroat trout which are broken into four major groups: coastal, westslope, Lahontan and Yellowstone cutthroat trout. These four groups appear to have diverged over 500,000 years ago, while the remaining 10 "minor" subspecies are more recent evolutionary divergences from Lahontan and Yellowstone ancestors (Behnke 1992, 1997, Johnson et al. 1999). Within the coastal cutthroats, there is an interior group and a coast range group, with no gene flow between them (Johnson et al. 1999). California's populations are at the southern end of the coast range group.

Life History: Coastal cutthroat trout possess a variable life history strategy ranging from fully anadromous to resident (DeWitt 1954; Pauley et al. 1989, Moyle 2002). This plasticity is among the most extreme in Pacific salmonids and variations in anadromy and potadromy are found both between and within populations. Offspring of resident fish can be anadromous and vice versa. The Smith River in California has both anadromous populations and resident populations isolated in small streams (e.g., Jones Creek). Migratory cutthroat trout generally do so when two to three years old, although they can enter sea water as late as their fifth year. When multiple forms coexist, temporal and spatial segregation presumably influence the genetic structure of the population and may lead to genetic differentiation between sympatric ecotypes within a watershed. Environmental conditions that affect growth rate, such as food availability, water quality, and temperature markedly influence the migratory behavior and residency time of coastal cutthroat trout (Hindar et al. 1991, Northcote 1992, Johnson et al. 1999). 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 more rigidly anadromous (but competitively dominant) salmonids; this flexibility may release cutthroat trout from competition and predation pressures at certain times of year.

Coastal cutthroat trout have ecological requirements analogous to those of resident rainbow trout and steelhead and when the two species co-occur, cutthroat trout occupy smaller tributary streams while the competitively dominant steelhead occupy larger tributaries and rivers. As a consequence, cutthroat trout tend to spawn and rear higher in streams than steelhead. Age at first spawning ranges from 2 to 4 years depending on migratory strategy and environmental conditions (Trotter 1991). Their life spans are 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). Resident fish generally reach sexual maturity between the ages of 2 and 3 years whereas anadromous fish rarely spawn before age 4 (Johnson et al. 1999). Sexually mature trout can demonstrate precise homing capabilities in their migrations to their natal streams. In northern California, coastal cutthroat trout migrate upstream to spawn after the first significant rain, beginning in August. Peak spawning occurs in December in the larger streams and January to February in smaller streams (Johnson et al. 1999). In California, ripe or nearly ripe females have been caught from September to April, however, indicating a prolonged spawning period.

Females dig redds 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). Females excavate redds in clean gravel with their tails. 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 by pushing upstream gravel over it with her tail. Each female digs a series of redds and 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.

Eggs hatch after 6-7 weeks of incubation depending on temperature. Alevin emerge as fry between March and June, with peak emergence during mid-April then spend the summer in backwaters and the stream margins (Johnson et al. 1999). Juveniles remain in the upper watershed until approximately 1 year in age at which point they may move about extensively through the watershed. Once this age is reached, it is difficult to determine the difference between sea-bound smolts and silvery parr moving back up into the watershed (Johnson 1999). Smolts or adults entering the salt water environment generally remain close to the shore and do not venture more than about 7 km from the edge of the coast (Johnson et al. 1999).

Adults feed on benthic macroinvertebrates, terrestrial insects in drift, and small fish, while juveniles feed primarily on zooplankton, macroinvertebrates, and microcrustaceans. (Romero et al. 2005, Wilzbach 1985). White and Harvey (2007) found that cutthroat trout of all sizes in small creeks fed mainly on aquatic insects in low numbers, but that earthworms washed in by winter storms may be bioenergetically most important for overwintering survival. In the marine environment, cutthroat trout feed on various crustaceans and fishes, including Pacific sand lance (*Ammodytes hexapterus*), salmonids, herring and sculpins. Marine predators include Pacific hake (*Merluccius productus*), spiny dogfish (*Squalus acanthias*), harbor seals (*Phoca vitulina*) and adult salmon (Pauley et al. 1989). Freshwater predators include the typical array of herons, mergansers, kingfishers, otters, snakes, and piscivorous fishes.

Habitat Requirements: Coastal cutthroat trout require cool, clean water with plenty of cover and deep pools for holding in summer. They prefer small, low gradient coastal streams and estuarine habitats. Optimal stream temperatures are less than 18° C. They require high dissolved oxygen and will avoid areas with less than 5 mg/L DO in the summer months. Feeding and movement of adults are impaired at turbidities of greater than 35 ppm. Embryo survival is greatly reduced at turbidities greater than 103 ppm and dissolved oxygen levels <6.9 mg/l. 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 (Pauley et al. 1989). Adults overwintering in streams, rather than estuaries, prefer pools with fallen logs or undercut banks but will also utilize boulders, depth, and turbulence as alternative forms of cover if woody debris is not available (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) Fish using large woody debris as cover are less affected by winter high flow events than those without such cover (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. Cutthroat preferentially use riffles and the tails of pools for spawning with velocities of 0.3-0.9 m/sec, though they have been observed spawning in velocities as low as 0.01-0.03 in small streams in Oregon (Pauley et al. 1989). Spawning has been recorded at temperatures of 6-17° C, with preferred temperatures of 9-12° C (Pauley et al. 1989, Moyle 2002).

Distribution: Coastal cutthroat trout are distributed from the Seward River in Southern Alaska to Salt Creek, a tributary to the Eel River estuary in Humboldt County, California. There are additional reports of small populations of cutthroat in Fortuna area tributaries and possibly in the lower Van Duzen River tributaries (Tom Wesleloh, pers. comm. 2008.). The interior range of the subspecies in Washington, Oregon, and California is bounded by the 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 only about 8 km wide at the mouth of the Eel River and 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).

In California, coastal cutthroat trout are at the southern edge of their range and have been observed in 182 named streams (approximately 71% of the 252 named streams within their range in California) and an additional 45 streams are likely support populations (Gerstung 1997). Self-sustaining populations apparently occur in many coastal basins including Humboldt Bay tributaries, Little River, and Redwood Creek (Gerstung 1997). The principal large interior basins where coastal cutthroat trout are the Smith, Mad and lower Klamath Rivers. Cutthroat trout also rear in approximately 1875 ha of habitat in five coastal lagoons and ponds—Big, Stone, and Espa Lagoons, and the Lake Earl-Talawa complex (Gerstung 1997). However, the largest populations are currently in the Smith River, and to a lesser extent, the lower Klamath River and tributaries (Gale and Randolph 2000). Gerstung (1997) indicates the lower Mad River as another area of high cutthroat occupancy, but more recent assessments indicate that it contains only a small population (T. Weseloh, pers. comm. 2008). Thus, as Gerstung (1997) noted, almost 46% of California coastal cutthroat trout populations occupy habitats in the Smith and Klamath River drainages.

Historical coastal cutthroat trout distribution may have once extended farther south to the Russian River in Sonoma County. 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.

Abundance: There are a limited number of long-term data sets readily available to evaluate population trends in coastal cutthroat trout and those data sets that do exist are primarily related to adult fish in Oregon and Washington. Data is spotty, scattered, and typically unpublished. There is no agency systematically keeping track of ongoing surveys. Records suggest that coastal cutthroat trout were more abundant historically and, in some locations, supported robust fisheries (Gerstung 1997). Current coastal cutthroat trout abundance is thought to generally be low in most waters, particularly where juvenile steelhead are present (Johnson 1999, Griswold 2006). Effective population size in California streams is difficult to determine, but Gerstung (1997) estimates that there are likely less than 5,000 spawners each year in all of California. The largest population apparently exists in the Smith River, where a local watershed monitoring group, the Smith River Alliance (SRA) conducts snorkel surveys for salmon and trout. For example, SRA surveyed various reaches of the South and Middle Forks of the Smith River in 2005 (totaling 34 miles) and the South, Middle and North Forks, totaling 47 miles surveyed in 2006 (Reedy 2005, 2006). The SRA 2006 surveys observed a total of 922 CCT in 2005 and 1361 adult CC T in 2006 (Reedy 2005, 2006). Previous population and trend data collections from the Smith River have been intermittent and represent only a small portion of the CCT range with inconsistent locations and methods over the years (Table 2). The Yurok Tribe has conducted anadromous salmonid surveys on the lower Klamath River and many of its tributaries and found cutthroat widely distributed in medium to high densities in nearly all of the lower Klamath tributaries downstream of Mettah Creek (Gale and Randolph 2000). Data covering a wider geographic area from Johnson et al. (1999) suggest that populations are generally low, if persistent, but with insufficient data for long-term trend analysis (Figure 1).

Fish Category	Species	Size Range (inches)	South Fork	Middle Fork
Cutthroat, large	*O. c. clarki*	*12 – 20"*	336	231
Cutthroat, medium	*O..c. clarki*	*10 – 12"*	242	130
Cutthroat, small	*O. c. clarki*	*7 – 10"*	174	96
Resident rainbow	**O. mykiss**	*10 – 12"*	43	45
Steelhead	*O. mykiss*	*16 – 28"*	11	14
Half-pounder	*O. mykiss*	*12 -- 16"*	10	7
Chinook salmon	*O. tshawytscha*	*18 – 42"*	11	0
Smallscale sucker	*C. rimiculus*	*8 – 20"*	4	23

Table 2. Results of 2006 snorkel survey on the South and Middle Forks of the Smith River conducted by the Smith River Alliance.

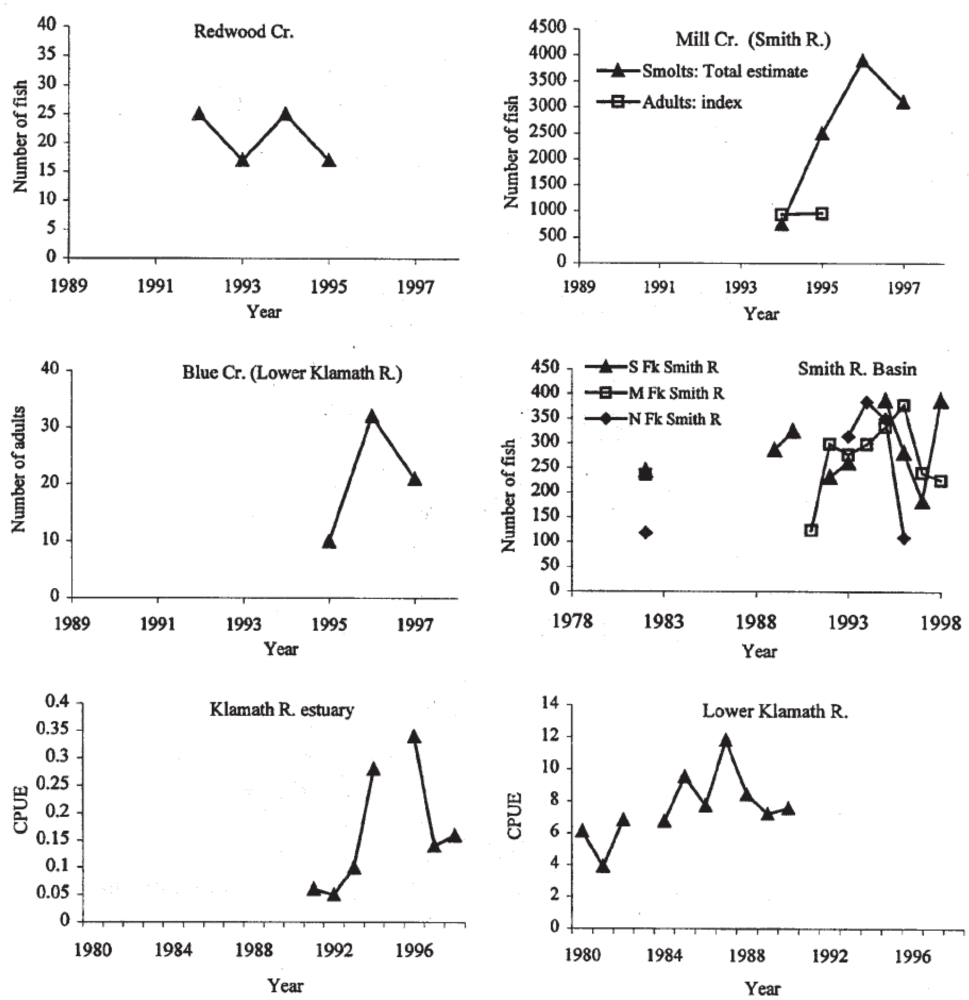

Figure 1. Coastal cutthroat trout abundance trends from Johnson et al. (1999). Data includes both snorkel surveys and electrofishing efforts.

Factors Affecting Status: Major factors affecting the status of coastal cutthroat trout include 1) habitat degradation, 2) dams and diversions 3) overexploitation, 4) interactions with hatchery salmonids ,and 5) hybridization with steelhead..

Habitat Degradation: According to Gregory and Bisson (1997), degraded habitat is associated with more than 90% of documented extinctions or declines of Pacific salmon stocks. Coastal cutthroat trout stocks are no exception to the rule. Major anthropogenic land-use activities, including agriculture, forestry, urban and industrial development, road construction, and mining, have resulted in the alteration and loss of cutthroat trout habitat and a subsequent loss in production (Johnson et al. 1999). Fish passage issues from loss of over-wintering habitat, changes in geomorphic processes and channel geometry, channelization and simplification of habitat in estuaries, the loss of large wood in channels, and road impacts on small headwater streams are all associated with habitat degradation in cutthroat trout range. Logging and associated road cuts have caused tremendous impacts to their habitat with massive landslides and erosion stemming from excessive tree removal on the steep, unstable soils found in the coastal mountains. Small streams (e.g., those favored by cutthroat trout) are inherently more susceptible to such impacts and have therefore been disproportionately damaged by land use practices. Johnson et al. (1999) cite numerous studies showing the importance of riparian vegetation to fish production and notes that in California, approximately 89% of the state's riparian forest has been lost with associated declines in aquatic habitat. Heavy erosion results in stream sedimentation and can elevate turbidity levels to intolerable levels as well as burying spawning gravel, altering rearing habitat and filling pools. Urbanization plays an important role in reducing cutthroat trout habitat in urban streams in the Humboldt Bay region and around Crescent City (T. Weseloh, pers. comm. 2008) but most impacts to stream habitat stem from agriculture, forestry practices, dams, and aggregate mining. Agricultural practices that impact cutthroat trout the most are likely water diversions and the associated dike building, damming, culverts and runoff. These factors result in degraded water quality, increased temperature, loss of in-stream flows, and loss of estuarine rearing areas (Johnson et al. 1999). Unfortunately, there are few studies that document such impacts specifically for coastal cutthroat trout.

Dams and Diversions: Dams and diversions have impacted flows on a number of coastal rivers, most conspicuously the Klamath and Mad Rivers within coastal cutthroat trout range. The impact of these dams on cutthroat trout is not known but altered flow regimes are unlikely to have had a positive effect. Likewise, the effects of small diversions, common in coastal streams, are not known.

Overexploitation: Gerstung (1997) indicates that historical runs of coastal cutthroat trout were quite large and that, in some areas, substantial commercial and sport fisheries existed for them. 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 of legal and illegal fishing are in fact unknown.

Interactions with hatchery salmonids: Coastal cutthroat trout are competitively subordinate to all other species of salmonid (Johnson et al. 1999) and hatchery production of steelhead in particular may deeply affect their numbers through predation and competition (Johnson et al. 1999). Some cutthroat trout are raised at the Humboldt State University and Mad River hatcheries and are planted in lagoons the support the fishery. Their interactions with wild fish are not known (but assumed to be minimal).

Hybridization with steelhead: Steelhead and cutthroat trout naturally co-occur and hybrids occur naturally, with no obvious impacts on cutthroat trout populations (Neillands 2001). However, habitat disturbance and other factors may increase rates of hybridization, with unknown consequences, but presumably to the detriment of the rarer cutthroat trout.

Conservation: The biggest single conservation need for cutthroat trout is more and better information so appropriate measures can be taken. The NMFS team that wrote the 1999 status review of coastal cutthroat trout in Washington, Oregon, and California grappled with the difficulty of assessing a species that was so data poor, concluding that "there is insufficient evidence to demonstrate that coastal cutthroat trout are at significant risk of extinction," as well as "there is insufficient evidence to demonstrate that coastal cutthroat trout are *not* at significant risk of extinction" (Johnson et al. 1999). Petition for listing coastal cutthroat trout under the ESA was therefore denied. In 2005, a symposium on coastal cutthroat trout was held in Port Townsend, Washington, followed by another in 2006 with the goal of "developing a consistent framework to help guide and prioritize conservation, management, research, and restoration of coastal cutthroat trout throughout their native range". This group was formalized (November 2006) as the Coastal Cutthroat Trout Executive Committee (Griswold 2006). Nearly a decade after the 1999 status report, the CCT Executive Committee found the state of coastal cutthroat trout research and monitoring remained virtually unchanged. The committee took up the task of determining the extent of current knowledge and identified data gaps and priorities for monitoring, assessment and restoration (Table 3).

In California, research and monitoring of cutthroat trout is taking place by the California Department of Fish and Game, Humboldt State University, the Yurok Tribe and other agencies and groups, but there appears to be little coordination of efforts or consistency of sampling across years. There has been no statewide assessment since Gerstung (1997).

Griswold (2006) notes "It should be recognized that a voluntary effort that tackles difficult scientific and monitoring issues for a non-listed non-commercial sub-species requires considerable leadership and good will from Federal and State agencies." Though doubtlessly true, it is presumably part of the mission of those agencies to monitor trends in a potentially declining species such as coastal cutthroat trout, regardless of their listing status and commercial value. There is certainly a chance that increased monitoring will determine that listing is indeed warranted. The development of this multi-agency group is a step in the right direction for cutthroat trout conservation but will not mean much if significant resources by state and federal agencies are not put into monitoring. This is especially true in California where cutthroat trout are at the southern end of their range and therefore exceptionally vulnerable to climate change. There are several non-governmental organizations that have started to monitor coastal cutthroat trout populations and the increased attention can only help with the conservation and restoration of this species, dubbed the "problem child" of West Coast salmonid species by the Oregon Fish and Game Commission as far back as 1946 (Griswold 2006).

Data Gap	Alaska	British Columbia	Washington	Oregon	California	Average
Incidence anadromous vs. other forms	4	4	5	5	5	4.6
Life history and ecology	3	2	5	5	5	4.0
Age specific survival	4	5	4	3	4	4.0
Smolt yields	4	5	4	2	4	3.8
Spawning and fecundity	5	2	3	4	5	3.8
Juvenile rearing habitat	3	4	4	3	3	3.4
Migratory patterns and adult habitats	3	3	3	4	4	3.4
Stream and habitat type	4	4	2	3	2	3.0
Isolated resident populations	3	3	3	2	3	2.8

Table 2. Data gaps identified for coastal cutthroat trout and their habitats ranked by priority of need to increase information (5 = high, 3 = moderate, and 1= low) by participants in the Coastal Cutthroat Trout Science Workshop from Griswold (2006).

Presumably, the many measures, both local and regional, taken (or proposed) to protect steelhead and salmon populations will also benefit coastal cutthroat trout, but even this is not known for sure. Particularly important, however, is continued management of the Smith River as a free-flowing, wild river that is a refuge for all salmonids, because cutthroat trout seem particularly abundant in this river. Important recent conservation events have been acquisition and protection of much of the Goose Creek and Mill Creek watersheds. Other targeted restoration efforts include Lake Earl (Jordan Creek, Stone Lagoon, and a few small creeks (Tom Weseloh, pers. comm.).

Trends

Short term: There is little data on the status and trends in recent years throughout coastal cutthrout range and the most thorough surveys published are eithedr 10 years or more old (e.g., Gerstung 1997) or only represent a few years of data (e.g., Reedy 2005, 2006).The surveys of the Yurok Tribe on lower Klamath River tributaries (Gale and Randolph 2000), however, are a good example of regional surveys that are needed on a regular basis; this particular set of surveys indicates cutthroat trout are still found where they would be expected. Nevertheless, quantitative measures of historical abundance are lacking; therefore it is difficult to say with any certainty whether populations are in decline, increasing, or stable (Johnson et al. 1999, Griswold 2006). Decline is the most likely scenario, however, because there have been changes to estuaries and watersheds and loss of structure and flows in cutthroat streams throughout its range in California. Fortunately, there is increasing protection for their streams (e.g. Smith River, streams in Redwood National Park) in part to protect coho salmon.

Long term: It seems likely that populations have been considerably depleted over the last 50 years because existing numbers suggest that the overall population in California is exceedingly low (Johnson et al. 1999). Climate change may also alter hydrology and increase water temperature in California streams to make more effective watershed conservation imperative for their persistence in the state. Developing the long term management strategies for coastal cutthroat trout is heavily dependent on improved monitoring and assessment. It appears that state, federal, and tribal agencies are now beginning to tackle the issue of monitoring status and trends of cutthroat trout or at least know they *should* be monitoring them.

Status: 3. Coastal cutthroat trout are apparently in no immediate risk of extinction but there is also high degree of uncertainty about their status in California (Table 2). Coastal cutthroat trout apparently persist in many streams on the northern California coast but in fact most populations are rarely monitored. They are listed by the California Department of Fish and Game as a Species of Special Concern and as a Sensitive Species in California by the U.S. Forest Service. While some fish from the Humboldt State and Mad River hatcheries are planted in lagoons (mainly Freshwater Lagoon) all other populations are entirely dependent on natural reproduction. This makes them unique among the more abundant North Coast salmonids, so they are therefore presumably a good indicator of condition of north coast streams in their range. Nevertheless, coastal cutthroat trout are a non-commercial, non-listed, widely distributed, and somewhat cryptic salmonid that supports only a minor sport fishery. Therefore, they will be neglected unless there is strong public and agency interest in protecting them and their habitats. There is a particular value in monitoring their populations to look at the effects of climate change on north coast rivers because of their low exploitation rates, wide distribution, and preference for smaller streams.

Metric	Score	Justification
1B Area occupied	5	Found in most watersheds from Eel River north.
2 Effective population size	3	This would be a '5' if we assumed all populations are genetically interconnected. Most appear to be small and isolated.
3 Intervention dependence	3	Persistence requires improved management of heavily logged watersheds.
4 Tolerance	3	Moderately tolerant of conditions in California streams
5 Genetic risk	4	Little information on genetics available; hybridization with steelhead may be a problem in some streams.
6 Climate change	2	Because most populations are in small streams, there is considerable range-wide vulnerability to climate change.
Average	3.3	20/6
Certainty (1-4)	2	Information is scattered and not systematically compiled.

Table 3. Metrics for determining the status of coastal cutthroat trout, where 1 is poor value and 5 is excellent.

BULL TROUT
Salvelinus confluentus

This account is derived from Moyle (2002) and sections in quotes are taken directly from Moyle (2002).

Description: The bull trout has fine scales (110 or more in the lateral series) and pelvic, pectoral and anal fins with white leading edges. Live fish are olive green in color with tiny yellowish spots on the back and small red spots on the sides. The body and fins lack black spots, although there are usually a few yellow spots at the base of the tail. 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. 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, 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 were once considered to be a variety of Dolly Varden charr (*S. malma*), a largely anadromous coastal species but studies by Cavender (1978), Hass and McPhail (1991), and others have eliminated doubts about their distinctiveness and species status. Museum specimens of California bull trout are distinct morphologically from other populations, but probably not sufficiently so to label them a subspecies.

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, p. 298-299. "In terms of basic life history, bull trout can be adfluvial (adults in lakes, spawning and rearing in streams), fluvial (all stages in streams, but adults migrate up tributaries for spawning), or resident (no separation of life history stages)…. Most resident populations occur in small streams, and it is possible that many, if not all, of these populations are remnants of populations that were once fluvial (e.g., populations in Klamath basin tributaries in Oregon)... In the McCloud River the population was apparently fluvial, 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. Fish gradually become more important in the diet as they grow larger. Bull trout more than 25 cm TL feed primarily on fish, including juvenile trout and salmon, sculpins, and their own young. Frogs, snakes, mice, and ducklings have also been found in their stomachs. Bull trout typically lie in wait underneath a log or ledge and then dash out to grab passing fish. …High bull trout densities are often associated with concentrations of small fish, often from migratory populations. Chinook salmon that once spawned in the McCloud River were presumably once a major source of food for local bull trout, both as loose eggs and as juveniles that reared in the river year round."

"Bull trout grow slowly but have long life spans (up to 20 years), and so 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 thereafter in resident populations and fastest in adfluvial populations, members of which may reach 40–45 cm TL in 5–6 years. The largest bull trout on record, from Lake Pend Oreille, Idaho, 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; a second display fish at the hatchery reached a similar size. 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 from fluvial and adfluvial populations 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. Movements toward spawning grounds can begin in July or August, but spawning does not begin until water temperatures have dropped below 9–10°C in late summer or fall, apparently in 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 her size, lays 1,000–12,000 eggs…"

"Embryos are buried at a depth of 10–20 cm and hatch in 100–145 days. After hatching 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 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…"

Habitat requirements: According to Moyle (2002, p. 298) "…the defining characteristic of streams containing bull trout is exceptionally cold, clear water, often originating from springs. They are rarely found in streams that have maximum temperatures greater than 18°C, and optimum temperatures appear to be 12–14°C for adults and juveniles and 4–6°C for embryo incubation. The McCloud River prior to the construction of McCloud Dam 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 conditions for spawning and rearing in the reach below Lower Falls, deep pools in the lower river for adults, and abundant food in the form of juvenile Chinook salmon…."

"Adult bull trout in rivers prefer to live on the bottom in deep pools; they are also associated with pools in smaller streams. Adfluvial populations thrive in large coldwater lakes and reservoirs (e.g., Flathead Lake and Hungry Horse Reservoir, Montana). In California, bull trout were unable to maintain populations in either McCloud or Shasta Reservoir, the two to which they had access. Juvenile trout (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 only about 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 streams of the upper Sacramento and Pit Rivers, but records are lacking. According to Moyle (2002, p. 298): "This was the southernmost population of the species. Today the southernmost populations are found in the Jarbridge River, Nevada, and small streams in the upper Klamath Basin, Oregon. The northernmost populations appear to be in the headwaters of the Yukon River, British Columbia. The easternmost populations are found in Columbia River tributaries in Alberta and Montana. In between these points 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 disjunct populations in their present range indicates a wider distribution in the Pleistocene period, under wetter and cooler conditions."

Abundance: Bull trout are now extinct in California.

Factors affecting status: According to Moyle (2002, p 299-300) the factors that resulted in the extirpation of bull trout from California are as follows.

"*Depletion of salmon*: In the 19th century the McCloud River supported at least two runs of chinook salmon, a run of steelhead, and a small run of coho salmon. Juveniles of these fish as well as the annual influx of energy from salmon carcasses quite likely supported fairly large bull trout populations. The 19th-century Sacramento River fishery combined with sediments from hydraulic mining severely depleted salmon runs coming into the McCloud. The Baird Hatchery, established on the lower river in 1874 to take eggs from chinook salmon in order to help restore depleted runs, may, ironically, have contributed to the further decline of McCloud River salmon because the weir next to the hatchery blocked much of the run at times. In the early 20th century the runs recovered somewhat, but not to former levels. Then in 1942 Shasta Dam closed and blocked access for all salmon. 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 1910 or so. They are present in small tributaries that juvenile bull trout may once have used for rearing. Brook trout will hybridize with bull trout, and this hybridization is a major cause of the decline of resident populations in Oregon and elsewhere. However, 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 do not seem to have been especially abundant until after the creation of Shasta Reservoir in the 1940s. The reservoir allowed a substantial migratory population of large fish 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.

"*Shasta Dam and Reservoir*: When Shasta Dam closed in 1942, it blocked access of major salmon runs, provided better habitat for migratory brown trout, and flooded about 26 km of the lower McCloud River, about a quarter of the bull trout's habitat. Although fluvial bull trout elsewhere have become adfluvial following the construction of reservoirs, this did not happen with Shasta Reservoir. Small numbers of bull trout appeared in the reservoir fishery, but runs from the reservoir never developed. Presumably the reservoir was just too warm for the growth and survival of bull trout (Rode 1990)"

"*McCloud Dam and Reservoir*: McCloud Dam, completed in 1965 and blocking the river about 45 km upstream from Shasta Reservoir, was the final blow to bull trout. First, it flooded 8 km of prime habitat for bull trout. Second, it probably severed the connection between juvenile and adult habitats by blocking adult migrations to upstream areas. Third, it 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). Once the dam was in place, the long-lived bull trout hung on for 10–12 years before dying out completely."

Conservation: CDFG has a plan for restoring bull trout, mainly by 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 reoperated (to produce colder water downstream), 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 may not be able to support a self-sustaining population of bull trout, especially in the face of competition from brown trout.

Trends: Bull trout are extinct in California. The last known bull trout caught in California was captured by UC Davis graduate student Jamie Sturgess in 1975, by hook and line. It was tagged and released. They 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 gone by the late 1970s.

Status: 0. Bull trout are extinct in California and are listed by the USFWS in 1999 as Threatened throughout the rest of their range in the USA.

MOUNTAIN WHITEFISH[7]
Prosopium williamsoni

Description: Mountain whitefish are silvery, coarse-scaled (74-90 on lateral line) salmonids, with a large adipose fin, a small ventral mouth, a short dorsal fin (12–13 rays) and a slender, cylindrical body. Gill rakers are short (19–26 on the first gill arch) with small teeth. 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. The tail is forked. The body is silvery and olive green to dusky on the back, and scales on the back are often outlined in dark pigment. Breeding males develop distinct tubercles on the head and sides. Juveniles are pencil-thin and silvery with 7–11 dark, oval parr marks.

Taxonomic Relationships: Mountain whitefish are regarded as one species throughout their extraordinarily wide range. A thorough genetic analysis will probably reveal a number of distinct population segments within their range. The Lahontan population in California and Nevada is the one most isolated from other populations and therefore is likely to be recognized eventually 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). Small juveniles feed on small chironomid midge, blackfly, and mayfly larvae but their diet becomes more diverse with 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. However, they will feed during the day on drifting invertebrates, including terrestrial insects (Moyle 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 seems to be one measuring 51 cm FL and weighing 2.9 kg from Lake Tahoe." 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 at water temperatures of 1–11°C (usually 2–6°C)…. Spawning is preceded in streams by upstream or downstream movements to suitable spawning areas, possibly as the result of homing to historical spawning grounds….. Movement is often associated with a fairly rapid drop in water temperature. From lakes, whitefish migrate into tributaries to spawn, but some spawning may take place in shallow waters as well… Whitefish do not dig redds but scatter eggs over gravel and rocks, where they sink into interstices. The eggs are not adhesive. Little is known about spawning behavior, but they seem to spawn at dusk or at night, in groups of more than 20 fish. They become mature in

[7] Most information in this account is from Moyle (2002).

their second through fourth year, although the exact timing depends on sex and size. Each female produces an average of 5,000 eggs, but fecundity varies with size, from 770 to over 24,000... The embryos hatch in 6–10 weeks (or longer, depending on temperatures) 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 gradually move into deeper and faster water, usually in areas with rock or boulder bottoms. 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)."

Habitat Requirements: Mountain whitefish in California inhabit clear, cold rivers and lakes at elevations of 1,400–2,300 m. Generally they prefer waters with summer temperatures <21°C. In streams, they are generally associated with large pools <1 m deep or deep runs. In lakes, they generally live close to the bottom in fairly deep water, although they will move into shallows during spawning season. 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.

Distribution: Mountain whitefish are found throughout the 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 Truckee, Carson, and Walker River drainages on the east side of the Sierra Nevada. Their range includes both lakes (e.g., Tahoe) and streams. Curiously, they are absent from Susan River and from Eagle Lake.

Abundance: According to Moyle (2002), "Mountain whitefish are still common in their limited California range, but their populations are fragmented. There is no question that they are less abundant than they were in the 19th century, when they were harvested in large numbers by Native Americans and then commercially harvested in Lake Tahoe. There are still runs in tributaries to Lake Tahoe, but they are relatively small and poorly documented. Whitefish apparently were already reduced in numbers by the 1950s. They still seem to be fairly common in low-gradient reaches of the Truckee, East Fork Carson, East and West Walker, and Little Walker Rivers. Small populations are still found in Little Truckee River, Independence Lake, and some small streams, such as Wolf and Markleeville Creeks, tributaries to the East Carson River. Their populations in Sierra Nevada rivers and tributaries have been fragmented by dams and reservoirs, and whitefish are generally scarce in reservoirs. A severe decline in the abundance of whitefish in Sagehen and Prosser Creeks followed the construction of Stampede and Prosser Reservoirs, respectively." These observations all 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 so that they can support recreational fisheries. At present, both California and Nevada allow 10 whitefish per day to be taken by anglers.

Factors affecting status: Mountain whitefish are little studied in California so factors affecting their abundance and distribution are poorly documented. The keys to understanding their apparent decline, however, are habitat-related: (1) they live primarily in the larger streams of the northeastern Sierras and associated lakes, (2) they do not seem to do well in most reservoirs, and (3) they require high water quality for persistence. Essentially, they live in the waters most likely to be impacted by human actions, especially by dams and diversions. 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. Mebane et al. (2003), however, noted that mountain whitefish were somewhat more tolerant of adverse water quality (high temperature, low dissolved oxygen) than other salmonids and therefore likely more resilient in response to environmental change.

Whitefish coexist in many areas with alien brown, brook, and rainbow trout, but it is possible that these trout may limit whitefish populations by preying on their fry, which have been recorded as an item in brook trout diets. Over-exploitation in past presumably depleted whitefish numbers but this no longer seems to be an issue, in part because few anglers target them, despite high catch limits and high edibility.

Conservation: Mountain whitefish are treated as a low-value game fish in California and elsewhere and seem to be able to handle whatever harvest exists today. But they should be treated as a more valued member of the fish communities of the eastern Sierras, as the one native salmonid that is still persisting in some numbers. 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. Thus the best thing that can be done for mountain whitefish is to maintain flows in the rivers at high enough levels in summer so that temperatures remain below 21° C at all times, preferably cooler and to otherwise keep water quality high.

It is clear that mountain whitefish in California would benefit from a thorough study of their biology, including systematics, genetics, distribution, abundance, and habitat requirements of all life stages. Results from such a study would improve management options for these fish.

Trends:

Short term: There is no evidence in that whitefish populations have declined significantly in last 5-10 years but no one is really monitoring their populations either.

Long term: Present numbers of whitefish in most of their habitats are presumably a small fraction of their historic numbers, when they apparently were one of the most abundant fish in the rivers and lakes of the eastern Sierra. However, all evidence for this is anecdotal.

Status: 4. Mountain whitefish are locally abundant in many areas although their distribution is presumably more restricted in California than it was historically (Moyle 2002). Their status may be poorer than indicated because of poor knowledge of their actual numbers (Table 1).

Metric	Score	Justification
Area occupied	4	Present in three watersheds and widely distributed outside state, assuming all mountain whitefish are the same taxon.
Effective pop. Size	5	Numbers appear to be large in the Truckee River and other streams.
Intervention dependence	5	They persist on their own, despite being ignored.
Tolerance	5	Whitefish are more physiologically tolerant than most salmonids, live at least 5 years, and are iteroparous.
Genetic risk	4	Genetics have not been studied but most populations are isolated from other large populations.
Climate change	3	Whitefish will be vulnerable to decreased flows, warmer temperatures and increased diversions that are likely to result from climate change.
Average	4.3	26/6
Certainty (1-4)	2	Most reports are anecdotal although there is some grey literature.

Table 1. Metrics for determining the status of mountain whitefish, where 1 is poor value and 5 is excellent.

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